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KEYNOTES

Origin Green, measuring and improving sustainability performance

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[Origin Green](#) is an independently verified, national sustainability programme launched in 2012 that operates at both farm and food manufacturing level in Ireland. Developed and implemented by Bord Bia – Irish Food Board, the programme aims to provide a structure for farmers and food manufacturers to demonstrate their current sustainability performance while developing and implementing plans for ongoing improvements. Results to date are highlighted in Bord Bia's first Origin Green [Sustainability Report](#), which was published in November 2015.

At farm level, Bord Bia has broadened the scope of its existing Quality Assurance programmes to incorporate sustainability issues such as greenhouse gas emissions, water, energy, biodiversity and animal welfare. Since the commencement of the programme more than 120,000 beef and dairy farm assessments have been completed with each farmer receiving an individual feedback report. In partnership with Teagasc, Bord Bia has developed a Carbon Navigator software tool, which helps farmers engage with practical measures that can improve on-farm profitability and enhance environmental performance. This tool is currently being rolled out to more than 30,000 farms through the CAP Rural Development Programme.

At food manufacturing level, companies can become a verified member of Origin Green by developing a multiannual sustainability plan with clear, measurable targets across three key areas: raw material sourcing, resource efficiency and social sustainability. Members are required to submit an annual progress reports to outline their performance towards reaching their targets. Plans and annual progress reports are verified by an independent third party. To date 202 companies, accounting for 90% of Ireland's food and drink exports have become verified members of the programme with a further 318 companies registering to take part.

The programme is currently being extended to include retailers and foodservice companies, with targets being set in relation to sourcing, food waste, packaging, health and nutrition, operations and social sustainability. This helps ensure that all parts of the supply chain up the point where consumers buy/consume the product are part of Origin Green.

The future focus for Origin Green is to help lower our environmental footprint, prioritise health and wellbeing, incorporate the complete supply chain on the Irish market, collaborate internationally to help further develop Origin Green and engage the Irish public with the programme.

How life cycle assessment (LCA) supports decision making at Nestlé

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ABSTRACT

Life cycle assessment and simplified eco-design tools have been very widely used at Nestlé over the past years. Introducing a systematic eco-design process in the innovation of new products has resulted in better decision making, but has also improved the understanding of sustainability issues among internal stakeholders and external partners in the supply chain. By sharing the approach as well as the underlying life cycle inventory data, the entire industry sector can improve and the goal of a sustainable food system can be achieved faster.

Keywords: food processing, simplified eco-design tool, supply chain engagement, water scarcity.

1. Introduction

For a company to be successful over the long term and to create value for shareholders, it must create value for society. For this purpose, Nestlé has embraced the Creating Shared Value (CSV) approach (Porter & Kramer, 2011). CSV in Research and Development means improving the sustainability performance of the next generation of products, by optimizing the product innovation and development process. To this end, Nestlé has implemented the concept of Sustainability-by-Design, where the sustainability performance of product development projects is evaluated at the different project stage-gates.

Nestlé uses life cycle assessment (LCA) to implement the environmental dimension of sustainability at the earliest stage of the product development cycle. The environmental performance of new products are systematically assessed using a simplified eco-design tool called EcodEX (Selerant, 2016), complemented where appropriate with critically reviewed LCA studies. Over the past years, more than 18'000 simplified eco-design assessments have been performed by the users of the tool. The tool users typically are product developers, process engineers, and packaging managers across the entire global research and development organization. This keynote illustrates in three concrete examples how the scientific relevance of the tool is currently being improved, how Sustainability-by-Design is supporting decision making in a business context, and how we share the tool and data developed with partners outside our company.

2. Regionalization of freshwater consumption impacts

In its original version, EcodEX measured water consumption at inventory level, based on the amount of cubic-meters consumed. It is, however, well known that the impacts of water consumption vary widely at a global scale, and are strongly driven by the availability of water resources at a given watershed. Therefore, EcodEX has further been developed to improve the way freshwater consumption is taken into account.

To this end, a new functionality has been implemented: key elements in the EcodEX life cycle inventory (LCI) database have been regionalized to better quantify the water scarcity impacts when products are developed with ingredients from different areas of the world. For example, rice in the United States is grown in a few areas only – many of them having a similar climate / water scarcity (USDA 2016). Therefore, by replacing water inventories with the average water scarcity in the areas in which rice is produced in the US, a relevant water scarcity value can be provided for rice in the US. This value will be very different from the water scarcity value of other crops or livestock, given their respective production regions.

Typically, water consumption in the agricultural phase (in particular during irrigation), is orders of magnitude higher than water consumption during food processing. Therefore, key agricultural commodities that are likely to be produced under irrigated conditions have been prioritized during the water scarcity regionalization process. Other profiles in the database (e.g. truck transport, packaging materials, ...) use country averages, continental, or global water scarcity values, given the contribution of water consumption from these profiles to a typical food product would be minimal.

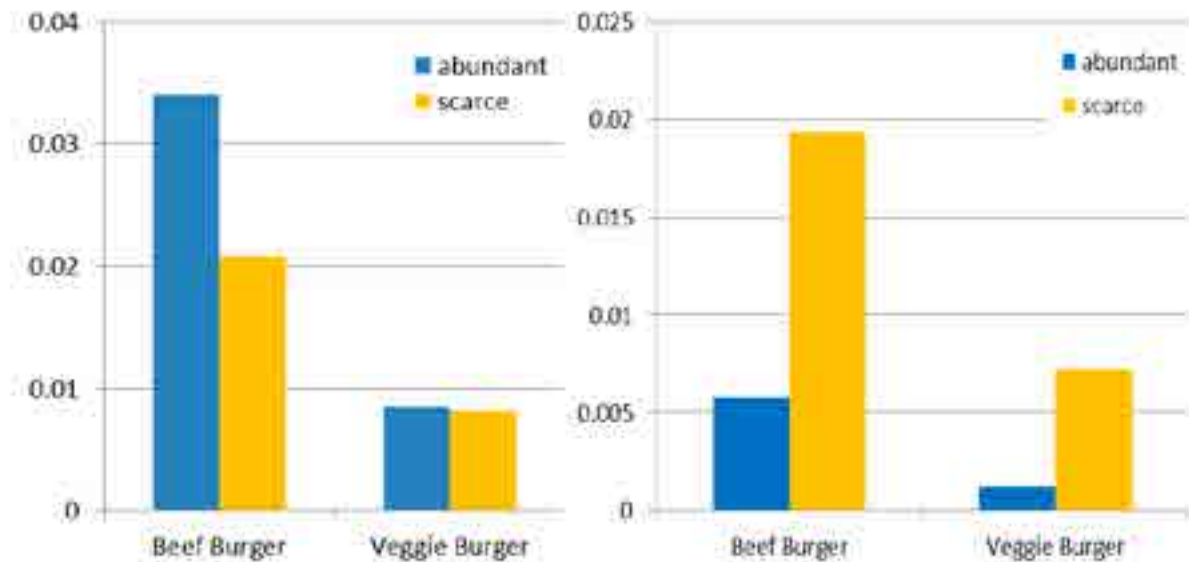


Figure 1: water withdrawal at inventory (left panel) and water withdrawal taking scarcity into account (right panel) of meat and vegetarian burgers. Blue bars represent sourcing from countries with abundant water, yellow bars sourcing from countries with water scarcity.

The regionalized LCI database is complemented by the new *Available Water REmaining* (AWaRe) consensus methodology on water scarcity (Matoshita et al, 2016), to improve the relevance of the LCIA method. A case study on meat and vegetarian burgers (see Figure 1) illustrates how decision making is improved using a regionalized assessment approach: by taking water scarcity into account (right panel), a vegetarian burger that is produced with ingredients from water-scarce regions (rightmost yellow bar) will have a higher water scarcity impact than a beef burger produced with ingredients from a country with abundant availability of water (leftmost blue bar on the right panel). This conclusion is not achieved when looking at water at inventory level (left panel).

3. Application in a business context

Tools like EcodEX allow our business units to better understand the sustainability hotspots in their value chain: by assessing representative products in their portfolio, strategic decision makers can identify the most relevant elements in their product category, and can optimally target sustainability projects. Furthermore, LCA can also contribute to strategic decision making, e.g. to favor one technology platform over another.

Wyeth Nutrition, a Nestlé business unit producing dairy-based infant nutrition has a strong presence in Ireland. Wyeth are focusing their sustainability efforts on their supply chain, because LCA has identified the agricultural phase (in particular dairy farming) as a relevant contributor to the environmental performance of infant nutrition.

“Origin Green” (Origin Green, 2016) is Ireland’s national sustainability program for the agricultural and food & drink sector. In collaboration with the Irish Food Board (Bord Bia, 2016), the dairy sector has developed an audited best practice program for sustainable dairy production in Ireland. Wyeth Nutrition have reviewed the sustainability plans of all their Irish dairy ingredient suppliers, extending the use of LCA throughout the supply chain. Responsible practices in the dairy sector can help improve the environmental performance of dairy operations. Relevant practices include manure management, improved nitrogen efficiency, and improved genetics and breeding.

Measures to optimize the environmental performance of a supply chain vary strongly from one region to another. Therefore, measures to improve environmental performance of dairy farming in Ireland have to be adapted to the regional context in other countries. The Nestlé dairy business has a very strong presence in Pakistan, and has been collaborating with local farmers and government agencies to establish a successful dairy industry in this region for the past 50 years. Measures to improve the environmental performance of the industry in Pakistan focus, among others, on prevention of losses and optimal drinking and feeding regimes for livestock.

4. Collaboration throughout the supply chain and across the industry

Wherever possible, Nestlé aims at making sustainability related data and tools widely available. This reinforces the credibility of the approach and improves the sustainability of the industry and supply chain partners. The EcodEX tool, for instance, has been developed by an external software provider, and is available commercially.

Nestlé has recently contributed to the development of the World Food LCA Database (Quantis 2016), a life cycle inventory database for the agro-food sector, financed by partners from agro-food industry and government entities. The database intended to develop key life cycle inventories for agro-food products which were not widely available in the existing life cycle inventory databases at the beginning of the project. The World Food LCA Database has developed hundreds of new life cycle inventory profiles from a wide variety of production systems and crop types. An overview of available life cycle inventory profiles is provided in Table 1.

Table 1: Database content for the World Food LCA Database

Product category	Number of LCI profiles
Arable plant products	50
Horticulture plant products	27
Perennial plant products	73
Meat products	76
Eggs	5
Dairy products	19
Fertilizers	10
Food processing	18
Food products	106
Mineral water	10
Plant production sub-processes	182
Animal production sub-processes	88
Land use change	169
Other	78

The World Food LCA database has decided to make the life cycle inventory profiles available outside the funding partners. Therefore, the database will be published through the Ecoinvent database. This increases the credibility of the World Food LCA database because a larger user base is able to test it and provide feedback. Furthermore, this also enables other user groups to benefit from the development of this LCI database. For example, work on Natural Capital Accounting or upcoming biodiversity standards could benefit from the data that have been compiled during the World Food LCA Database development.

There is the intention to continue the current version of the World Food LCA Database with a follow-up project, to which new partners can also contribute. Data on many areas of agro-food production are still lacking.

5. Conclusions

The eco-design and LCA approach taken by Nestlé has shown to be a helpful contribution to achieving more sustainable value chains. The benefits not only come from the improved decision making that is introduced by LCA, but also from the dialogue that is resulting from LCA and eco-design tools: internal stakeholders focus on aspects that really matter and develop more sustainable business strategies. Furthermore, a common language is found for discussions with external partners in the supply chain. Sustainability is a challenge that has to be tackled by the entire society and not just individual companies. Sharing tools and knowledge allows reaching the goal of more sustainable food systems faster.

6. References

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UNEP Sustainable Food Systems Report

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Food systems are key to at least 12 of the 17 Sustainable Development Goals (SDGs); the food we grow, harvest, process, trade, transport, store, sell and consume is the essential connecting thread between people, prosperity and planet. Food systems are the dominant users of many natural resources, particularly land, terrestrial and marine biodiversity, fresh water, nitrogen and phosphorus. While food production is a major driver of biodiversity loss, soil degradation, water depletion and greenhouse gas emissions, the use of natural resources goes beyond primary food production. For instance, large amounts of fresh water are used in food processing, biomass is widely used for food packaging and/or cooking, and fossil fuel energy is used extensively for almost all food system activities. The people who manage our food systems are thus the largest group of natural resource managers in the world and are hence crucial agents of change in the transformation of current production and consumption systems. So as to ensure all people have appropriate amounts of safe and nutritious food, natural resources need to be managed more efficiently, thereby reducing environmental impacts. As UNEP's International Resource Panel recently noted, 'resource-smart' food systems are needed to deliver on the SDGs.

Implementing sustainable agriculture – a role for gamification?

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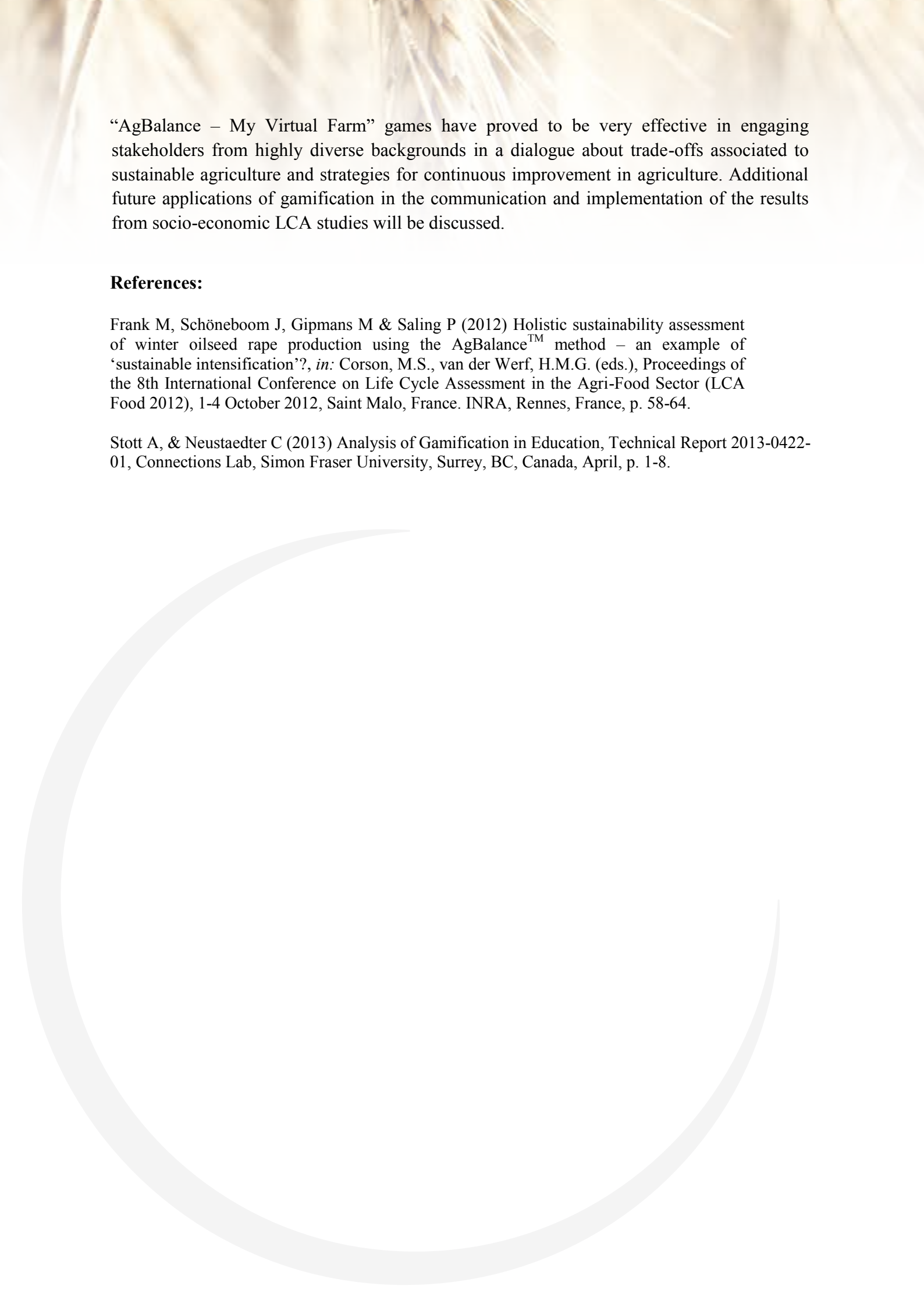
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Abstract:

Measuring sustainability with socio-economic LCA has the potential to guide continuous improvement programs towards a more sustainable agricultural production. However, a prerequisite for this is the translation of the study findings and implications into farmers' everyday language and experiential horizon. This is the basic idea of a suite of online games named "AgBalance – My Virtual Farm".

AgBalance™ comprises a multi-criteria socio-economic life cycle based methodology that uses hweighting and aggregation of individual results into a single sustainability score (Frank et al. 2012). The method provides insights into the three dimensions of sustainability and, thus, can enable farmers, food industry, policy makers and influencers to co-create strategies for the continuous improvement of agriculture. Although Life Cycle Assessment has proved a useful diagnosis tool to reveal the key drivers of sustainability in agriculture, the translation of the results of LCA studies into practically relevant scenarios for decision making on farm is often problematic. Gamification can facilitate this translation in an educational and entertaining way. Gamification refers to the application of game dynamics, mechanics, and frameworks into non-game settings. An educator interested in harnessing the dynamics that underlie games would be well advised by a focus on the game dynamics 'Freedom to Fail', 'Rapid Feedback', 'Progression' and 'Storytelling' as they to prove successful throughout the literature (Stott & Neustaedter 2013). By utilizing gamification carefully, teachers can direct their classroom environment towards success in raising both engagement and achievement.

The simulation games "AgBalance – My Virtual Farm" were designed to provide a self-learning exercise to better understand the complex field of sustainable agriculture. The objective of all versions of the games is to manage farms in different geographies of the world for a maximum of five seasons. The players decide on the crops to be grown and their rotational sequence, the use of subsidy schemes to foster agroecology, the degree of input and resource intensity of the production as well as the tillage system used. The goal is to reach the best "sustainability score" over the five seasons. This "sustainability score" consists typically one or two economic indicators (farm profit, economic risk, pesticide resistance risk etc.) contributing 50% to the score. In addition, a number of environmental indicators (e.g. Global Warming Potential, Eutrophication Potential, Ecotoxicity Potential, Soil Health and Biodiversity Potential) together contribute another 50%. Moreover, randomly applied action/event cards ('community chest') are included, e.g. extreme weather conditions, changes in market conditions or the political environment. The underlying logic of the games was adopted from the research literature, various LCI and LCIA databases and BASF's yearlong experience in LCA studies in agriculture.



“AgBalance – My Virtual Farm” games have proved to be very effective in engaging stakeholders from highly diverse backgrounds in a dialogue about trade-offs associated to sustainable agriculture and strategies for continuous improvement in agriculture. Additional future applications of gamification in the communication and implementation of the results from socio-economic LCA studies will be discussed.

References:

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Unlocking the Value of your Metrics: How leading companies are building strategies and reaching goals based on LCA

Emmanuelle Aoustin

Quantis, CEO

Plenary Panel session lead by Quantis CEO Emmanuelle Aoustin to feature high-level industry partners such as Mondelez, Nestlé, Mars or even AccorHotels (largest food service provider globally), that will discuss how LCA helped them build solid environmental sustainability strategies and reach their goals. Each partner will provide a business case about how they used metrics to create value for their business. The audience will see the clear benefits of doing LCA and how to concretely implement the results in their operations.

Impetus for Social LCA in the food industry

Dr. Catherine Benoit Norris

New Earth/ Harvard Extension

There is a wide and acknowledged movement for increased transparency on social sustainability. This movement is growing fast and currently benefits from two major efforts at the policy level: the UN Guiding Principles and the Sustainable Development Goals. One of the key aspects of the Guiding Principles is its focus on due diligence. Human rights due diligence is defined by the Guiding Principles as “a business’s ongoing processes for assessing its actual and potential human rights impact, integrating and acting upon its findings, tracking its responses and communicating how its impact is addressed” (United Nations, 2012). The Sustainable Development Goals invite governments, organizations and individuals to work together to meet a set of targets on 17 goals. These goals represent a common vision of what as a global society we are striving to achieve and are a push, motivating companies to not only reduce and manage their risk, but also to generate more positive impacts.

Social LCA is one of the best kept secrets of the Life Cycle tools family. Following the release of the UNEP SETAC Guidelines in 2009, which provided a first framework building on experts’ consensus, a large number of case studies, methodology articles and reviews have been published and databases made available (www.socialhotspot.org). Additional guidance documents were also published by the Social Roundtable and WBCSD (chemical industry group), attempting to bridge the gap between expert’s perspectives and companies’ imperatives. Food is one of the sectors which benefits from a large number of case studies applying and testing methods and data types, surveying stakeholders.

Because of the need for technical approaches to calculate supply chains social impacts, for internal use, for sustainable purchasing and for communication, Social LCA now finds itself in the spotlight. Social LCA’s systematic and iterative process is an attractive option but its strong LCA connection makes it hard for social responsibility managers to engage with, and the lack of clarity and diverging approaches on important aspects of the methodology can scare new professionals. In this case, translation between different fields of research and practice as well as the creation of tools that are adapted to different audiences are critical

Combining environmental and nutritional impacts & benefits in food LCA: Why have we waited so long?

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Food and agriculture have been from the start one of the focus domain of LCA, with multiple dedicated conferences, sessions, and has played a key role in pioneering multiple methodological developments, such as the allocation hierarchy, the development of inventory databases (e.g. ecoinvent, food LCA database), the development of land and water use indicators or the assessment of ecosystem services for the impact assessment side. Food is indeed a unique and dominant sector for a broad variety of impacts categories. We will therefore first review several of the main LCA milestones and progress in which food LCA has played a key role since the early nineties, including a) the eutrophication impacts of fertilizers and their dominant effects on marine and freshwater ecosystem, b) the impacts on humans of fine particulate due to ammonia emissions to air, c) the toxicological impacts of pesticides on aquatic and terrestrial ecosystem quality and on human health, d) the impact of land use on biodiversity, e) the impact of water used for irrigation and its incidence on water scarcity and subsequent effects for biodiversity and humans, f) the positive or negative incidences of biomass production and use on energy and material resources, carbon cycle and more recently as an emerging domain on g) the ecosystem services impacts related to soil quality. No surprise that in a sector with so many potential trade-offs, a holistic approach such as LCA is highly needed and applied.

Another dominant influence of food is of course the nutritional impacts and benefits on human health during the use phase. It is strange that despite its claim for comprehensiveness, LCA has often neglected to consider the direct impacts of product on consumers during the use phase, be it for cosmetics or for food. We will review the different recent attempts and progress made to include the nutritional impacts and benefits in food LCAs. A first type of approach has been to consider a quality corrected functional unit that takes the nutrient content of food products into account (e.g. when considering fat- and protein-corrected milk). A second approach is to define functional units based on a single nutritional aspect (e.g., protein content or caloric energy) or using nutritional indices to aggregate multiple nutritional dimensions into a single score. Attempting to force impacts and benefits of nutrition into the functional unit however creates conceptual dissonance within an LCA framework built on expressing impacts in the numerator and positive outcomes (function) in the functional unit denominator.

Nutritional epidemiology-based information, as captured by the Global Burden of Disease (GBD) consistently expresses in Disability Adjusted Life Years (DALYs) multiple risk factors often associated with food and with pollution. Building on this work, the Combined Nutritional and Environmental Life Cycle Assessment (CONE-LCA) framework evaluates and compares in parallel the environmental and nutritional effects of foods or diets, expressed in DALYs. We will demonstrate the application of CONE-LCA to several studies on dairy products and on fruits and vegetables, and present how the approach is generalized to address entire dietary consumption in different countries. We will finally discuss the potential and limits of these approaches and identify key needs towards improved reliability in considering both environmental and nutritional aspects in LCA Food studies.

How do we establish life cycle assessment methods to assess the impact of agricultural production on soil functions?

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ABSTRACT

Soil quality attributes, and their contribution to soil function, are inherently linked to agricultural productivity, and the long-term sustainability of agriculture relies on protecting and improving our soils. The influence of soil quality on productivity and ecosystem services has been under-represented in Life Cycle Assessment (LCA) to date. Recent efforts by the UNEP-SETAC Life Cycle Initiative and the European Commission JRC have devised impact assessment frameworks that capture the ecosystem service functions of land, including soil functions. However, there is still ambiguity over the terms used to describe soil processes, elementary flows, and impact indicators. This paper explores important soil processes, identifies how elementary flows can be estimated, and explores how different aspects of soil quality can be characterised to give an integrated assessment of impact. In doing this, we define the language and terms used in the impact pathway that will help delineate inventory development from development of impact assessment methods. We discuss how the tools now available, through the growth of GIS data on land use, soils, and climate, open up the opportunity to parametrise LCI directly with the relevant elementary flows and construct LCIA methods, so that they are matched to real production systems.

Keywords: soil processes, baseline properties, biomass production, elementary flows, life cycle impact assessment.

1. Introduction

Soil quality attributes, and their contribution to soil function, are important environmental values as they are inherently linked to agricultural productivity and long-term sustainability of farming operations. In this paper, we consider soil functions that are important from the perspective of biomass production and related ecosystem services. Broader soil functions such as the provision of building materials, anchoring support for human structures and protection of archaeological treasures are not directly covered.

Many processes in the soil that affect soil quality, and subsequently soil functions, are influenced by farming inputs such as fertiliser, management activities such as tillage practices, and the type and quantity of product produced and exported from the land. When considering soil function in a life cycle assessment (LCA) context, it is important that the impact of these agricultural interventions can be reliably assessed so that the choice of alternatives, such as synthetic versus organic fertiliser, can be compared across a comprehensive range of impact categories. Supply chain participants can then use LCA to benchmark the environmental performance of current practices and identify ways in which this profile can be improved, acknowledging that there may be trade-offs between alternative practices.

Soil qualities, and the contribution they make to soil function, have not been widely included in LCA studies, and this offers the opportunity to develop methods that are purpose-built, recognising the challenges that come with land-based production systems. Once the goal and scope of an LCA have been defined, there are two distinct phases before results can be interpreted – the collection of information related to the production system (“technosphere”) where the fate of substances is managed (as represented by life cycle inventory, LCI), and the assessment of impact on the natural environment (“ecosphere”, as represented by life cycle impact assessment, LCIA). The challenge with land-based agricultural systems is that there is not a clear delineation between technosphere and ecosphere, with the soil being considered as part of the agricultural “factory” while also being a resource from nature. The issue then arises as to whether changes in the soil should be included in modelling the LCI or LCIA.

For some impact categories such as global warming and eutrophication, a precedent has been established for soil carbon, nitrous oxide (both direct and indirect), nitrogen and phosphorus to be

included as inventory elementary flows (European Commission Joint Research Centre 2010). There has been considerable consultation on how pesticides should be modelled, with the consensus being that primary distributions to the various compartments (soil, air, water) should be included in LCI, while movement of pesticides through secondary processes (such as leaching or run-off) should be reported with inventory to inform LCIA (van Zelm *et al.* 2014; Rosenbaum *et al.* 2015).

To date, the work on soil function from an LCI perspective and from the perspective of land use impact assessment has not been well integrated. A large body of scientific research exists describing the impact that agricultural practices have on soil quality measures (SoCo Project Team 2009). Since it is a broad, integrative, and context-dependent concept, soil quality cannot easily be described by direct measurement. Instead the combination of several proxy measurements (e.g. soil pH, organic matter, bulk density) may provide indicators of how well the soil is functioning. While methods exist for assessing soil quality, the range and complexity of indicators used is not consistent, and there is little international agreement on a harmonised framework (Nortcliff 2002). The most prevalent research theme on soil quality focuses on indicator selection and evaluation (Karlen *et al.* 2003). Some authors have also contributed to the development of LCA that includes aspects of soil quality (Garrigues *et al.* 2013; Núñez *et al.* 2012; Oberholzer *et al.* 2012). However, it still remains for the LCA community to clearly articulate how these soil quality measures will be integrated into impact assessment, involving the development of new impact pathways and the connection with existing related impact pathways (e.g. climate regulation and biodiversity).

Considerable thought has gone into defining parts of the impact assessment pathways for soil function and land use (Garrigues *et al.* 2013; Koellner and Geyer 2013; Núñez *et al.* 2012; Oberholzer *et al.* 2012; Saad *et al.* 2013). Two recent initiatives, by the UNEP-SETAC Life Cycle Initiative and the European Commission Joint Research Centre, have drawn on these studies to devise impact assessment frameworks that capture the ecosystem service functions of land, including soil functions. The focus of the former has been to develop an impact pathway for biodiversity (Figure 1), while the latter has been in response to the need for a common approach to impact assessment of land use in the context of Product Environmental Footprints (PEF), with a specific focus on soil function (Figure 2).

While there is considerable overlap between these two impact pathways, there are areas of ambiguity, particularly regarding the definition of soil processes, LCI flows and LCIA mid-point indicators for soil function. To explore this area in more detail, a workshop was organised by a consortium of agencies (ADEME, France; Agroscope, Switzerland; CIRAD, France; CSIRO, Australia; EC JRC, Italy; and Life Cycle Strategies, Australia), which was held in conjunction with the Life Cycle Management Conference in Bordeaux in late August 2015. It was attended by 38 LCA scientists. The goals of the workshop were to build a shared understanding of the soil issues and research being undertaken, develop a roadmap for progressing the integration of soil function into LCA, and form an information network of relevant researchers and organisations. The format of the workshop was inspired by the Pesticide Consensus Group workshops (Rosenbaum *et al.* 2015).

An action from the soil workshop was to develop a framework for discussion by the international LCA community on integrating soil function into LCA that: 1) identifies all processes connected to soil quality; 2) establishes definitions for terms and the language used to discuss soil function; 3) indicates where these processes should be considered as elementary flows in inventory or parts of the impact pathway; 4) establishes more broadly which impact categories elementary flows contribute to; and 5) proposes a characterisation factor that allows diverse soil quality measures to be aggregated for impact assessment. This is an ambitious task and this paper starts the framework development by defining language, proposing what soil attributes are best described by elementary flows in LCI, identifying what impact pathways soil quality measures contribute to, and suggesting a possible approach to characterisation of aggregated soil impacts. These formed topics for further discussion at a follow-up workshop in Dublin, held in conjunction with LCAFood2016 Conference in October 2016, and will subsequently contribute to discussions in the UNEP-SETAC Sub-Task on Ecosystem Services.

2. Methods

The language of soil quality in an LCA context: Agreed language and a common understanding of the terms we use in LCA are essential for productive discussions on developing soil quality as a mid-point indicator for impact assessment end-points. In this paper we propose the use of the following terms and definitions as detailed in Table 1.

Table 1. Terms and definitions used in reference to soil quality, soil function and LCA

Term	Definition
Soil quality	The capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation (Karlen <i>et al.</i> 1997).
Soil quality measures	Measures that can be made on soil to indicate an improvement or a deterioration in soil quality, that affects the ability of soils to deliver important functions.
Soil function	Important functions that soils deliver: nutrient cycling, water regulation, biodiversity and habitat, filtering and buffering, and physical stability/support (soilquality.org 2011).
Soil process	Physical, chemical or biological processes that occurs in the soil.
Soil baseline properties	Baseline measure of intrinsic/inherent soil properties to be taken into account to determine impact or as a point of comparison. This baseline condition is distinct from the “reference state” used in LCIA, although baseline properties may be used for both.
Input, activity data or inventory reference flow	Input, farming practice or output from the production system that influences soil processes.
Inventory elementary flow	An emission or resource flow to or from the technosphere to the ecosphere, caused by the effect that an input, farming practice or output has on a soil process.
Midpoint impact	Intermediate impact along the cause-effect chain that relates changes in soil quality and function to subsequent related impact categories, e.g. biomass production and climate change.
Endpoint impacts	Impacts at the end of the environmental cause-effect chain, close to areas of protection.
Areas of protection	Currently resource use, ecosystem quality and human health.

Is soil part of the farm “factory” or part of nature? The boundary between technosphere and ecosphere is used to define what elementary flows should be in the LCI and what downstream effects are represented in LCIA. Estimating elementary flows between the technosphere and ecosphere becomes more complex for agriculture where the soil on the farm forms an important part of the technosphere but is also considered to be a natural resource within which an environmental impact can occur. The balance between these can vary depending on land use, where the soil under arable use could be considered as a “highly manipulated ecosystem” (van Zelm *et al.* 2014), hence part of the technosphere, while soil in extensive grazing land may be considered as part of the ecosphere.

Flows associated with agriculture can be categorised into three classes: those that are clear emissions to nature e.g. N₂O from fertiliser use; those that accumulate or deplete resources within the field boundary e.g. hydrogen ions causing soil acidity; and those that leave with the product e.g. heavy metals from fertiliser taken up by plant products. Guidance from the ILCDC Handbook (European Commission Joint Research Centre 2010) is that all of these flows should be recorded in the inventory, indicating that agricultural soil should be considered as part of the technosphere. The Pesticide Consensus Working Group (Rosenbaum *et al.* 2015) have reached the same conclusion, recommending that primary pesticide flows to air, soil and water should be included in LCI. The critical issue is to ensure that LCI and impact assessment methods are aligned so that there is no overlap in the modelling and neither double counting nor missing flows distort the impact burden.

From the soil quality perspective, we assume that flows associated with soil processes that are important to soil functions should be included in LCI, in order to better account for the influence of practices. This would call for agricultural soil be treated as part of the technosphere. However, this raises the issue of how damage to the technosphere (the soil) is accounted for in impact assessment, as

damage to the technosphere in this context is important. For example, an increase in soil acidity needs to be considered differently to damage to the floor of an industrial factory during use.

3. Results and Discussion

Establishing the framework: Based on the premise that agricultural soil is part of the technosphere, we have developed a framework for linking soil processes through to the impact on soil functions. We first define all the relevant soil process (Table 2). We then considered what might be important soil baseline properties that would need to be used to set the context of a particular elementary flow. Where the LCI has been regionally defined, this information could be documented in LCI as many of the soil attributes for the region under study are accessible as GIS data. However, the intended use of soil baseline properties is for impact assessment. For example, the elementary flow of hydrogen ions which makes soil more acid has an impact only once a critical soil pH is reached. Time to critical pH is a function of the starting pH and the inherent buffering capacity of the soil. Hence, to make an impact assessment of a change in flow of hydrogen ions these two pieces of information would be required for the system under study.

Once important soil processes are identified, the next step for inventory development is to understand how these are influenced by agriculture. For each of the soil processes we have identified which inputs (e.g. fertiliser, pesticides), activity data (e.g. tillage practices, irrigation) and reference flows (e.g. yield of product) have an effect. For instance, mineralisation of organic matter (and the reverse process of immobilisation) is influenced by tillage practices (e.g. no-till versus conventional), residue management (e.g. burning versus retaining stubble), and N fertiliser rates (through the effect that fertiliser quantity has on yield and subsequent quantity of crop residue returned to the soil).

The next step is to define and quantify the elementary flows that occur due to the effect that the inputs, activities and reference flows have on soil processes. When selecting elementary flows some principles need to be considered: flows need to be additive in a linear manner (i.e. twice as much is twice as bad/good); they should be modelled as substance flows (i.e. clearly inventory flows rather than impact assessment indicators); and should be generic and applicable in all regions and countries. In Table 2 we suggest appropriate elementary flows resulting from each of the soil processes. Many of these are familiar to LCA practitioners as they are elementary flows that are used for established impact assessment methods (e.g. CO₂, N₂O and NH₃ to air, N and P to water), while others are new as they are specific to the impact of agriculture on soil (e.g. sodium to soil) or because they have not been considered in current impact assessment methods (e.g. hydrogen ions to soil water).

The final step is to identify which impact categories these elementary flows contribute to, as some elementary flows from soil processes will be picked up by multiple impact pathways. The flow of biogenic CO₂ from soil as the result of mineralisation of organic matter will contribute to both soil's function to produce biomass, and climate regulation. Likewise, the flow of hydrogen ions to soil water contributes to the acidification impact category as well as soil function, while soil loss also contributes to eutrophication (via transported P to water ways) and respiratory inorganics (from airborne soil particles). Identifying all the impact pathways is an important step in making sure elementary flows are fully accounted for and aspects of environmental damage are not overlooked.

Untangling impact pathways that include soil function: The impact pathways described by the UNEP-SETAC Life Cycle Initiative (Figure 1) and the European Commission Joint Research Centre (Figure 2), originate from the perspective that land use is the intervention documented in the inventory, and impacts are then ascribed to particular land uses. This approach results in many of the mid-point indicators being soil processes (erosion, mineralisation of SOC, physical-chemical soil conditions). For example, in Figure 1 there is a pathway from compaction to soil stability to erosion. How does this connect to a soil function? Soil stability (in terms of reduced losses through erosion) could be simply connected to the soil function of biomass production, as we indicate in Table 2 but this is not at all the perspective in the framework in Figure 1, where "biotic production" and "erosion/regulation" are on the same level as midpoint indicators. We need to address these disconnects to be able to advance our thinking about how to incorporate soil function into LCA.

Table 2. Proposed framework for considering important soil processes, baseline properties for impact assessment, the relationship between soil processes and the agricultural system, the elementary flows produced by each soil process and the impact indicators to which these elementary flows contribute.

Soil process	Soil baseline properties	Input, activity data (farming practice), or output from the production system that affect soil process	Inventory elementary flow for soil processes	Midpoint impact assessment indicator
Mineralisation and immobilisation of organic matter	Stock of soil organic carbon (SOC)	Tillage practice; Crop residue management; N fertiliser (~ yield → crop residues); Organic fertiliser; Irrigation	Biogenic CO ₂ from and to air N ₂ O to air	Biomass production Climate regulation
Acidification	pH pH buffering capacity	Yield of product exported; Crop residues retained; Off-farm biomass inputs (manure, purchased fodder); N fertiliser (type and quantity); Lime	Hydrogen ions to soil water (accumulation within the technosphere)	Biomass production Acidification
Erosion	Slope Vegetation cover Soil texture Organic matter content	Tillage practice; Irrigation; Modification of slope and vegetation cover; P fertiliser	Tonnes of soil to air/water Kg P to water	Biomass production Eutrophication (soil-bound P to water) Respiratory inorganics (airborne soil particles) Climate regulation (oxidation of SOC in air borne soil particles)
Volatilisation of N compounds		N Fertiliser application rate Timing of fertiliser application	NH ₃ to air	Acidification Climate regulation
Denitrification of N compounds		Irrigation N Fertiliser application rate Timing of fertiliser application	Emission of N ₂ O, NO _x to air	Climate regulation Acidification Eutrophication
Aggregation		Bulk density of native soil Tillage practice	Change in bulk density (accumulation within the technosphere)	Biomass production
Salinisation	Natural depth of water table Sodium stock in soil water	Irrigation	Sodium to soil (accumulation within the technosphere)	Biomass production
Leaching of soluble compounds	Nutrient stock in soil	Irrigation; Pesticide application rate; Fertiliser application rate; Timing of fertiliser application	N to groundwater Pesticides to ground water	Ecotoxicity Eutrophication Human toxicity
Contamination	Natural background levels in soil	Fertiliser application rate; Type of fertiliser; Pesticide application rate; Type of pesticide; Yield of product	Grams of contaminant in soil (accumulation within the technosphere) Grams of contaminant in the product	Biomass production Ecotoxicity Human toxicity
Albedo	Reflectance	Tillage practice		Climate regulation

The impact pathway, proposed for midpoint assessment of soil functions (Figure 2), presents a more systematic construct that reflects the relationship between soil processes, soil quality measures and the subsequent impact of changes in these measures on soil function. It also incorporates the concept of intrinsic/inherent soil properties. However, it is still premised on land occupation/transformation as the only inventory flow to drive changes in soil function.

An alternate approach is to model soil processes in the inventory with the effect of interventions (inventory inputs, activities, and reference flows) on these processes directly expressed as elementary flows, and that the impact of these elementary flows on soil quality are what drive the impact assessment, rather than land use alone. Therefore, to facilitate a rational and clear link between elementary flows in LCI (that reflect the impact of interventions on soil processes) we suggest that in addition to information on land occupation/transformation, inventory include relevant elementary flows related to each of the soil processes. An example of this is how erosion is modelled as an inventory flow of soil loss (in grams) (Núñez *et al.* 2012).

Developing characterisation factors: The final step is to develop characterisation factors that link elementary flows to the mid-point impact indicators of soil quality and ability of the soil to produce biomass. This requires the additional work of identifying possible mechanisms for arriving at a common unit for soil function and its impact on biomass production. This is a significant area of work where a number of modelling approaches have been proposed. A recent review of these approaches (Vidal Legaz *et al.* 2016) concluded that none of the models provide a comprehensive solution; the more relevant a model was for assessing soil function the less applicable it was to LCA.

An alternate approach may be to use plant growth models such as APSIM (Keating *et al.* 2003) to determine how the change in soil attributes (SOC, pH, electrical conductivity, soil compression) affect biomass production. The characterisation factor then becomes a direct estimate of biomass in units of kg/ha, which can be easily characterised into impacts on available food, biofuel, carbon stores and vegetation cover, providing the link to climate regulation, ecosystem quality and human health.

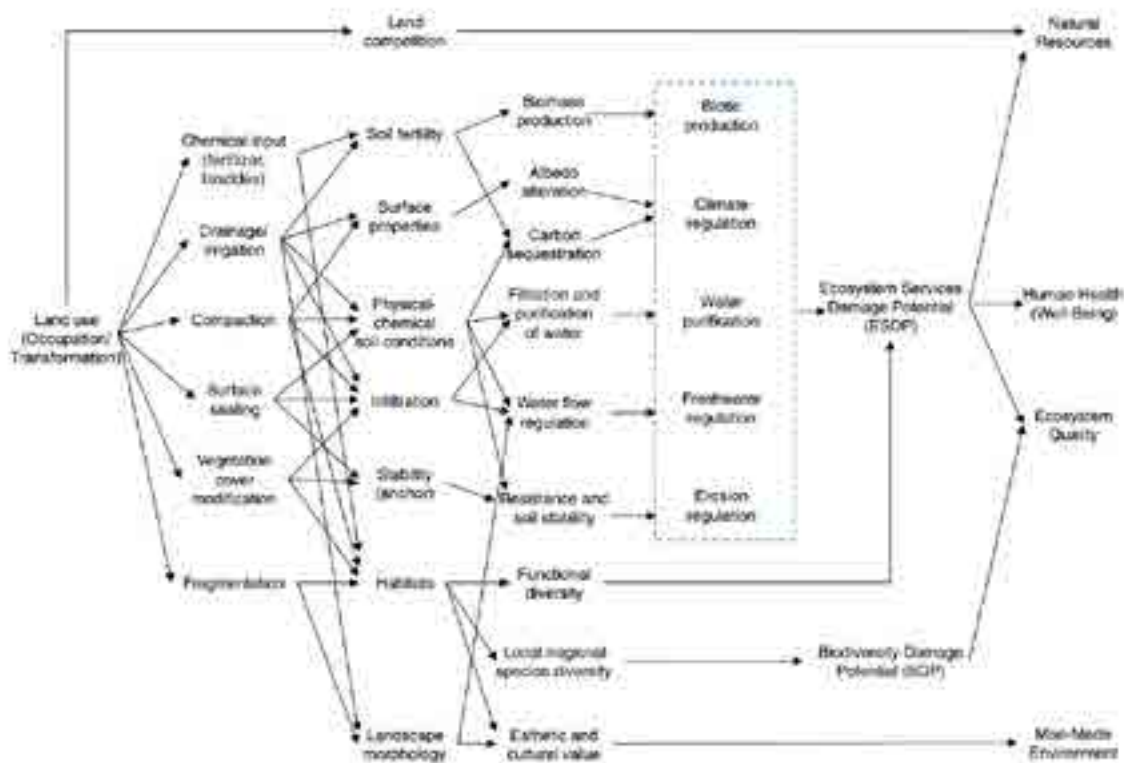


Figure 1. UNEP-SETAC guideline on land use impact assessment. From (Koellner *et al.* 2013)

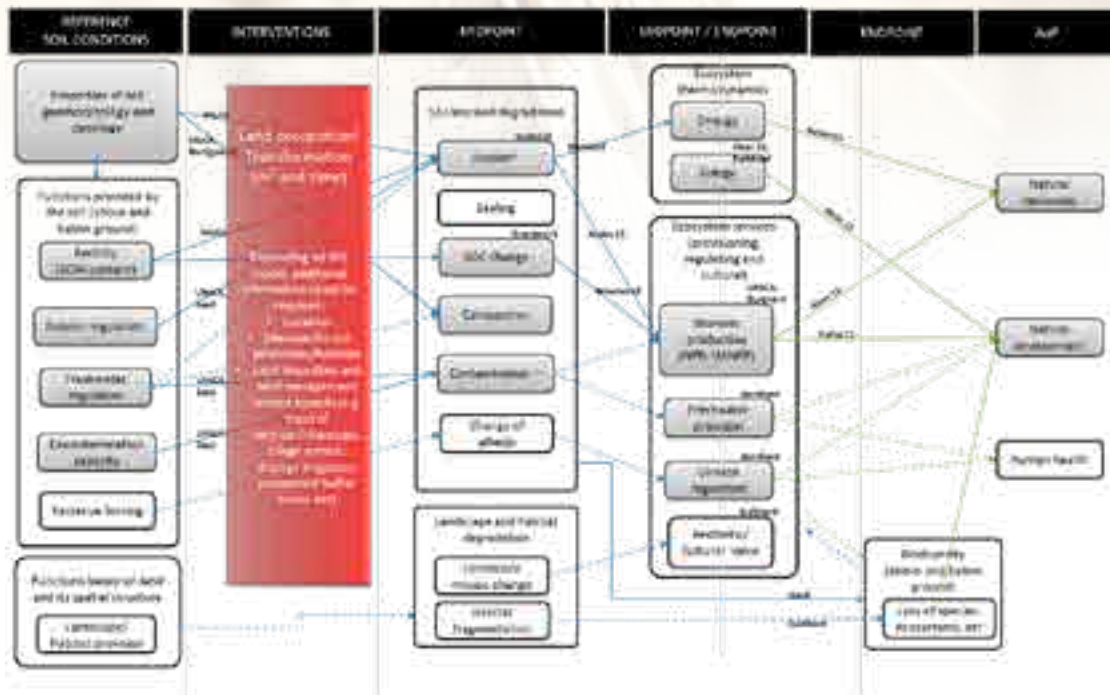


Figure 2. Land use impact pathway under discussion: Review of models, impact pathways, indicators and characterization factors towards a robust and comprehensive midpoint assessment of soil functions (Vidal Legaz *et al.* 2016).

Implementation of plant growth models is not easy due to the high level of parametrisation required. However, the increasing amount of data available from resources such as nation databases like the Soil and Landscape Grid of Australia (Terrestrial Ecosystem Research Network 2016), enables APSIM to be run spatially over many points, at the scale of an agro-ecological region. This approach is feasible and is currently being implemented to populate Australian agricultural LCI with elementary flows, such as change in SOC. APSIM could potentially be applied to LCIA, in manner similar to the way it is used to assess yield gaps (the gap between actual and potential crop yield) (yieldgapaustralia.com.au 2016). This concept is not that different from biotic production potential based on SOC (Brandão and i Canals 2013), but more representative of real production systems and encompassing the full range of soil quality attributes that contribute to biomass growth.

5. Conclusions

The development of LCA methods to incorporate soil function into LCA is at an exciting stage. We are now seeing scientists from across a range of domains exploring how to undertake this complex task. This will bring knowledge, skills and tools from a wide perspective which will stimulate innovative solutions. The tools we now have available through the growth of GIS data (on land use, soils, and climate) open up the opportunity to parametrise LCI directly with the relevant elementary flows, and construct LCIA methods that are matched to real production systems. It is important that this international engagement continues, building on research and exploring new approaches, through Consensus Workshops and the formal UNEP-SETAC Sub-Task on Ecosystem Services.

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Deforestation: No more excuses

Sebastian Humbert

Quantis Scientific Director

Keynote Session presented by Quantis Scientific Director Sebastian Humbert.

Sebastian will present why deforestation is an important topic - particularly in the food industry and to its industrial players. He will also provide examples of when measuring deforestation made a difference by looking at examples of when deforestation was considered and when it was neglected. Practical solutions that can be used to measure deforestation will also be presented. Updates will be provided on the status of developing international standardization (note: ISO standard).

LCA - potentials and limitations for decision support

Bo Weidema

2. -0 LCA Consultants

LCA has shown – and continues to show - its strength as a technique for focussing internal organisational resources on the issues and activities that are of largest environmental importance, for decision-making, and for communication between businesses and to consumers. But in the public arena, LCA has become entrenched in seemingly endless political and technical debates over modelling principles, data quality, and transparency. To provide support for much-needed, correct and timely decisions, LCA must avoid the treacherous pitfalls of insufficient and misleading information, high uncertainty, and powerful interests that lead to wrong or delayed decisions.

Value of Information analysis and Power Analysis are two tools that can be used to guide LCA safely over the political and technical pitfalls to cost-efficiently deliver its full potential.

Value of Information analysis provides tools to assess the information requirement of a decision situation, the already available (often sufficient) information, and the costs of additional information versus the benefits of avoiding false negative or positive outcomes. Value of Information analysis places the focus on the role played by uncertainty and ignorance. When applied to LCA as a post-normal science domain, this raises the questions of the inclusion of the consequences of non-action, of how to determine the acceptability of uncertainty, and of the practical applicability of the precautionary principle. This leads directly to the need for power analysis to determine the identity and interests of winners and losers, the weak and the strong, providing insight in the power game that LCA results are used in. In this power game, where science and democracy are the first victims to manipulation, there are three important tactical instruments for decision makers that seek to build coalitions for sustainable development: openness, due diligence, and economic investment. Governments, business and NGO's can play both positive and negative roles in this game.

ORAL PAPERS

WATER AND EUTROPHICATION

1. Water and Eutrophication

15. Global water scarcity resulting from Swiss food consumption

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ABSTRACT

Water scarcity is a severe environmental problem across the world. Food consumption is a driver of that scarcity, but its effects can occur at a location far distant from the location of food consumption. This study linked Swiss food consumption to global water deprivation and aimed at identifying impactful trade relations and products. To achieve that, multi-regional input-output analysis was coupled with regionalized life cycle assessment. Countries most affected by Swiss food consumption are Spain, India and Italy, and products with the highest leverage are olives, almonds and castor beans. Such information can help policy-makers as well as individual consumers to make more sustainable choices and to reduce their water scarcity footprint.

Keywords: Agriculture, international trade, environmental outsourcing, life cycle impact assessment

1. Introduction

While the human society is increasingly aware of the need for environmental sustainability, it faces the challenge of satisfying a higher food demand. Globally, water scarcity is among the most severe environmental pressures and affects human health, the economy as well as biodiversity (Scherer and Pfister 2016a). Agriculture is the largest water consumer (Shiklomanov and Rodda 2004) and crop yields highly depend on the water available to plants (Jägermeyr et al. 2016). Therefore, while food production increases also water demand does. It leads to increased water scarcity, but food production itself is affected by water scarcity.

International trade leads to large distances between local food consumption and remote food production. Such teleconnections (Hubacek et al. 2014) hide the impacts from the consumer. Previous studies that linked water consumption to trade disregarded impacts (Hoekstra and Hung 2002, Feng et al. 2011). However, providing information on water consumption can be misleading, as small volumes do not necessarily cause small impacts (Ridoutt and Pfister 2010). Therefore, this study aims at relating local food consumption to global water scarcity impacts and uses the example of Switzerland.

2. Methods

Life cycle assessment (LCA) was coupled with trade analysis. The consumed food (343 items) and direct trading partners of Switzerland were obtained from the Swiss Farmers' Union and the Swiss Customs for the year 2012. Since some exporters are not producers but re-exporters, some products were traced back to the trade origin using EXIOBASE (Wood et al. 2015), a multi-regional input-output database (differentiating 7 crop sectors). This approach results in import shares from multiple regions which are sometimes large. Therefore, the regions were disaggregated to countries using FAO export and production shares (FAO 2015). This combination of Swiss trade, MRIO and FAO data increases the precision of the results compared to previous approaches.

Derived products were converted to their primary counterparts (e.g. olive oil to olives) by mass and value fractions (FAO 2003, Chapagain and Hoekstra 2004). Livestock products were evaluated based on the feed the animals eat. More details on the trade analysis and product conversion can be found in Scherer and Pfister (2016b).

The impacts of crop water consumption were assessed by water scarcity indices (Scherer and Pfister 2016a) for two different irrigation scenarios (Pfister et al. 2011). Deficit irrigation represents a minimum estimate and expected irrigation is the geometric mean of deficit and full irrigation. The

impacts were expressed as water scarcity footprints and translated from a mass basis to a calorie basis (USDA 2016), as calories better represent the function of food.

3. Results

Switzerland produces half of the food it consumes within its boundaries. Nevertheless, the largest part (>99%) of the water scarcity footprint caused by Swiss food consumption lies abroad. This points to the environmental outsourcing facilitated by globalization. Especially Spain (exporting almonds, olives and grapes), India (exporting castor beans, coffee and cashews) and Italy (exporting olives, grapes and beef) are severely affected by Swiss food consumption (Fig. 1). If deficit irrigation is used as the measure, the United States (exporting almonds, other nuts and milk) become more relevant. Ecuador also bears a relatively high share in any case especially due to cocoa production.

Olives, almonds and cocoa were identified as the primary products with the highest impacts for the aggregated Swiss food consumption (Fig. 2). Per kilogram castor beans and almonds stood out among the products with a large consumption, whereas on a calorie basis asparagus (imported from Mexico, Spain and the United States) caused by far the most water scarcity. This makes sense, as vegetables are mainly consumed for other purposes than calories.

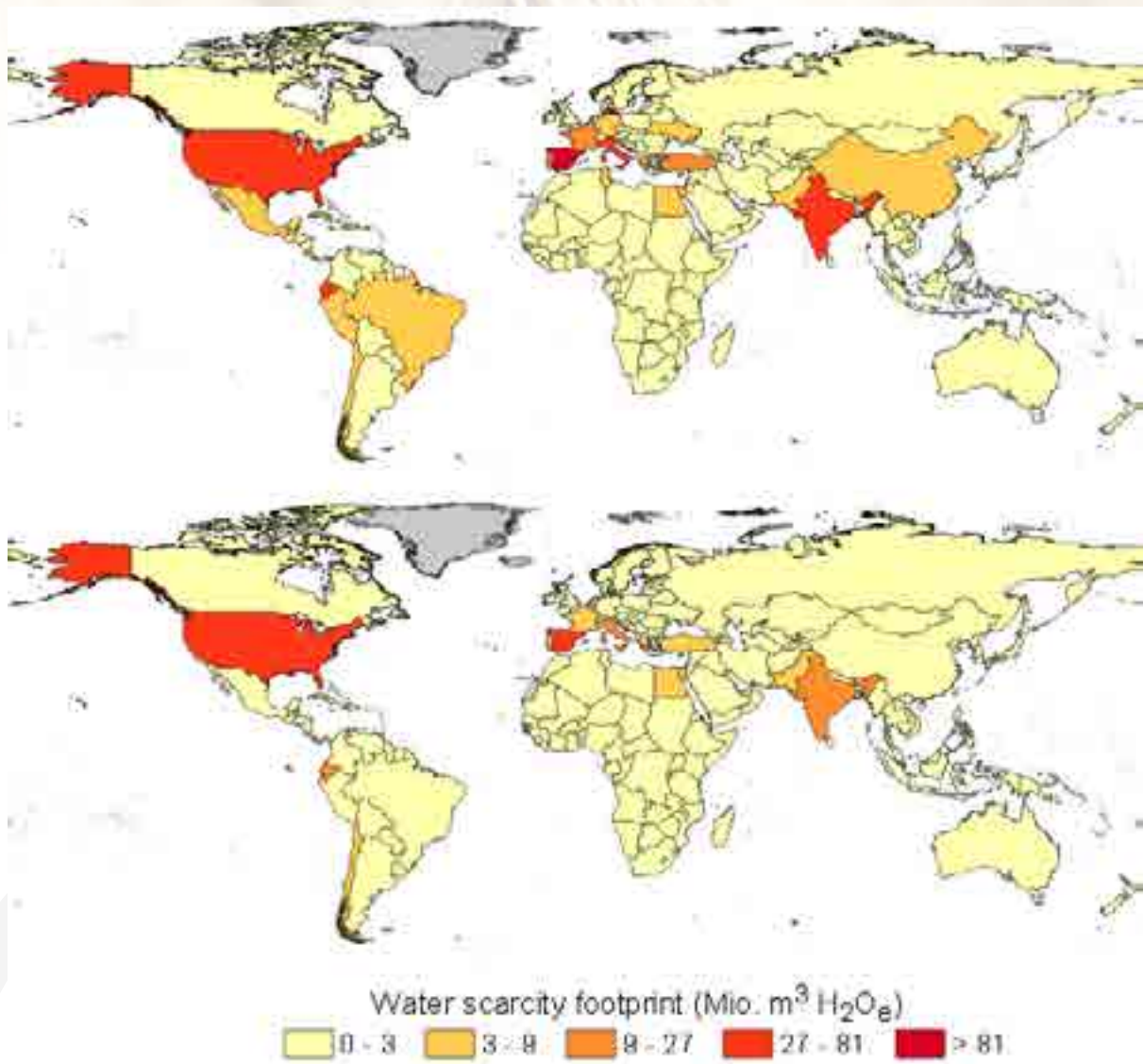


Fig. 1: Expected (top) and deficit (bottom) water scarcity footprint caused by Swiss food consumption. Grey colours indicate countries without food trade with Switzerland.

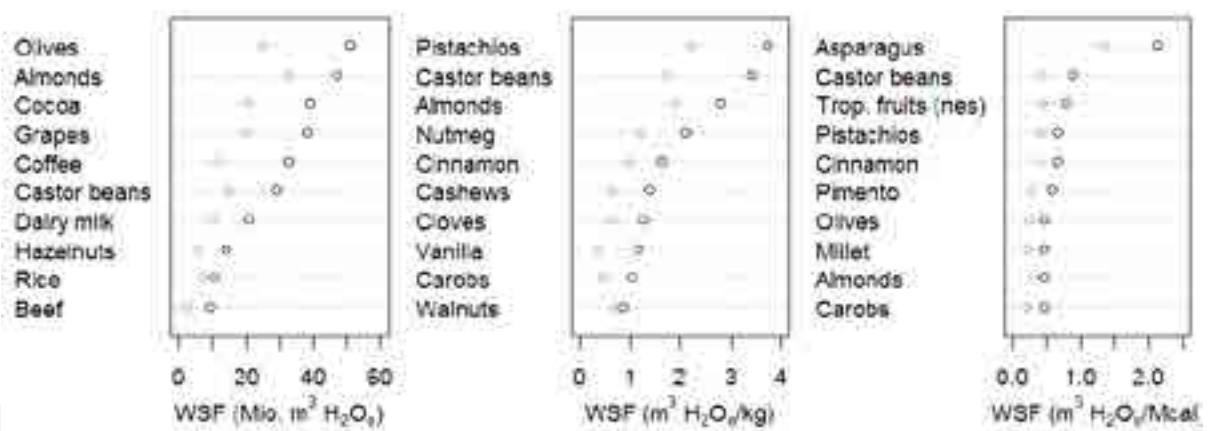


Fig. 2: Top 10 products concerning the water scarcity footprint (WSF) of Swiss consumption (left: total consumption, middle: per kg, right: per Mcal). Blue and black dots indicate deficit and expected water consumption.

4. Discussion

In a related study, the biodiversity impacts caused by Swiss food consumption were analyzed. Also in that case Spain, India and Italy were identified as countries highly affected by Swiss food consumption (Scherer and Pfister 2016b). A previous study investigated the mass-based carbon and water footprints of 34 fruits and vegetables consumed in Switzerland and identified asparagus for both categories as most impactful product (Stoessel et al. 2012). Both studies are in accordance with the obtained results. Nevertheless, the results are sensitive to the approach used for tracing back trade origins, which especially applies to the footprints derived from deficit water consumption (Scherer and Pfister 2016b).

Most studies quantify environmental footprints per unit mass of the product (e.g. Pfister et al. 2011, Scherer and Pfister 2016b); however, from a global food and nutrition security perspective, impacts per calorie content are more insightful. Similar to this study, a Californian study has shown that using nutrient-related footprints can lead to different conclusions and product rankings than those using a mass basis (Renault and Wallender 2000). More valuable than single nutrients would be the consideration of multiple nutrients and their balances (Heller et al. 2013). As an example, asparagus is low in calories but rich in proteins, dietary fibers, iron and vitamins A, C and K (USDA 2016), which demonstrates that the crop is not consumed for energy uptake but its nutritional quality. Furthermore, a mix of crops is required for a healthy diet and therefore impacts of crop combinations in different diets might be assessed in future.

Leverage is highest if the impacts per kg or calorie are high and at the same time the consumption is high. Consequently, the reduction in consumption of olives, almonds and castor beans have high leverages because they cause among the highest impacts with regards to the total Swiss consumption, but are also in the top 10 products per kg (almonds and castor beans) and per calorie (all three). When aiming at improving the Swiss consumption in terms of environmental impacts, not only water scarcity but other aspects such as land use need to be considered in order to avoid burden shifting.

5. Conclusions

Switzerland outsources most of the water scarcity (>99%) caused by its food consumption. The water scarcity footprint depends not only on the product, but also on the product origin. Therefore, changes in diets and trade relations can reduce the footprint. The identification of spatial hotspots and impactful products can support policy-makers as well as individual consumers in making decisions and also highlights where efforts for detailed assessments are most valuable.

The choice of the functional unit influences the comparative results and has to be chosen thoroughly. Instead of relating the impacts to the mass of the food products, as it is mostly done, we encourage to estimate nutritional footprints.

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102. Application of Environmental Impact models for water scarcity to a European basket of food products

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ABSTRACT

Addressing water scarcity and increasing water-use efficiency have been recently listed by the United Nations amongst the Sustainable Development Goals to be reached by 2030. A set of life cycle-based indicators was developed by the Joint Research Centre of the European Commission with the aim of assessing the environmental impacts of the final consumption of goods of an average European citizen, covering key areas of consumption such as mobility, housing and food, by adopting a life-cycle framework and including, among others, impacts associated to the use of water. This work has the objective of quantifying the impacts due to water consumed in the production, use and end-of-life stages of selected food products largely consumed in Europe, as well as to analyze how different modelling choices have relevant effects on the results. Therefore, a set of impact assessment models was applied to life cycle inventories of products largely consumed within the EU previously developed within the framework of Life Cycle-based Indicators. This set include the new consensus-based model resulting from the UNEP/SETAC Water Use in LCA (WULCA) working group - AWARE model, other interim options that were considered in the consensus building process such as demand-to-availability ratio (DTA), demand-to-availability combined with aridity (DTA_x) as well as LCIA models available in literature, including the ILCD-recommended water depletion indicator. The results show that the majority of the LCIA models are in substantial agreement in the identification of the top 3 contributors to water scarcity due to the EU food consumption patterns. Regional contribution analysis has shown additional insights for 4 selected case studies, revealing that flows with 'unspecified' geographic location are frequently those determining the most of the impacts. The AWARE100 model was proven to be applicable to the assessment of water scarcity as no major inconsistencies were found and results were compatible with results provided by other LCIA models.

Keywords: water footprint, regional contribution analysis, LCIA, WULCA, AWARE model.

1. Introduction

Addressing water scarcity and increasing water-use efficiency has been included within the United Nations' Sustainable Development Goals (UN, 2015) – Goal 6. According to UNEP (2012) several complementary tools to the quantification of water uses and their environmental impacts are needed at several levels of water management, including: statistical water accounting on a macroeconomic level such as input-output analysis, Water Footprint Assessment (WFA) and Water-use assessment and impact assessment in the context of Life Cycle Assessment (LCA). A set of Life Cycle-based Indicators on was developed by the Joint Research Centre of the European Commission in order to assess the environmental impacts of final consumption of goods of an average European citizen. This set of indicators, called Basket-of-Products, covers key areas of consumption such as mobility, housing and food, by adopting a life-cycle framework and including, among others, impacts associated to the use of water (EC-JRC, 2012; Notarnicola et al., 2016). In particular, a European Food Basket LCA model was developed on the basis of the selection of 17 products largely consumed within the EU, representative of the average food consumption per person in EU in 2010 (Notarnicola et al., 2016). Scientific literature and direct industrial sources were used for the foreground data, whereas background data were mainly taken from the Agrifootprint (Blonk Consultants, 2014) and Ecoinvent v.3 (Weidema et al., 2013) databases and EU-27 ELCD dataset for electricity (see details in Notarnicola et al., 2016). This work has the objective of quantifying the impacts due to water consumed in the production, use and end-of-life stages of selected food products largely consumed in Europe, by means of applying a wide set of impact assessment models for water scarcity. This set includes the most recent methods developed in the context of the UNEP/SETAC life cycle initiative (see Boulay et al., 2016) as well as a selection of LCIA models (and related characterization factors – CFs) which can be used for assessing 'water scarcity' as defined by ISO 14046 (ISO 2014).

2. Methods

In this work the full set of products included in the LCA model of the European Food Basket is characterized with a number of selected mid-point LCIA models for water scarcity (Table 1). The selection of mid-point models is based on previous work developed in the context of the update of the

LCIA impact categories for Product/Organization Environmental Footprint. The list includes the UNEP/SETAC interim recommended characterization model Available Water Remaining after demand is met (Boulay et al., 2016 submitted) as well as alternative versions of it based on different modelling assumptions (AWARE100_EWR50, AWARE10, AWARE1000) and interim options that were considered in the consensus building process, such as demand-to-availability ratio (DTA), demand-to-availability combined with aridity (DTA_x). The characterization is conducted by means of Simapro 8.0.5.13 software so to identify the major contributors to water scarcity. A regional contribution analysis is performed for 4 selected products: mineral water, coffee, sugar from beet and beef meat. This analysis is carried out by aggregating those flows sharing the same geographic specification, accounting for both water withdrawals and releases so to identify which regions are the ones affected the most by these impacts. Those flows not providing any geographic specification are aggregated in a specific class called ‘unspecified’. The aim of these analyses is to: identify which phases contribute the most the 4 selected products and assess whether LCIA models provide converging or diverging information.

Table 1: Selected LCIA models

Model	Indicator	References
GENERIC MIDPOINT		
ILCD v1.07	Water depletion [m ³ eq.]	Frischknecht et al., 2009; EC-JRC 2011
ILCD v1.07 (as implemented in SimaPro)	Water depletion [m ³ eq.]	Frischknecht et al., 2009; EC-JRC 2011
Swiss Ecological scarcity 2013	Ecological scarcity [Eco-points]	Frischknecht and Büsser Knöpfel, 2013
Boulay et al. 2011 – simplified (a)	Water scarcity [m ³ eq.]	Boulay et al., 2011
Pfister et al. 2009 - WSI	Water stress (WSI) [m ³ eq.]	Pfister et al., 2009; Pfister & Bayer 2013
Hoekstra et al. 2012	Blue water scarcity [m ³ eq.]	Hoekstra et al., 2012
Berger et al. 2014	Water depletion [m ³ eq.]	Berger et al., 2014
Available Water Remaining after demand is met AWARE100; and alternatives: AWARE100-EWR50; AWARE10; AWARE1000; DTA _x and DTA	User deprivation potential [m ³ world eq.]	Boulay et al. 2016 submitted
Yano et al. 2015	Water scarcity footprint [m ³ H ₂ O eq.]	Yano et al. 2015
HUMAN HEALTH – MIDPOINT		
Motoshita et al. 2014	Agricultural water scarcity [m ³ eq.]	Motoshita et al. 2014
NAURAL RESOURCES – MIDPOINT		
Pfister et al., 2009 - resources	Surplus energy [Joules]	Pfister et al., 2009

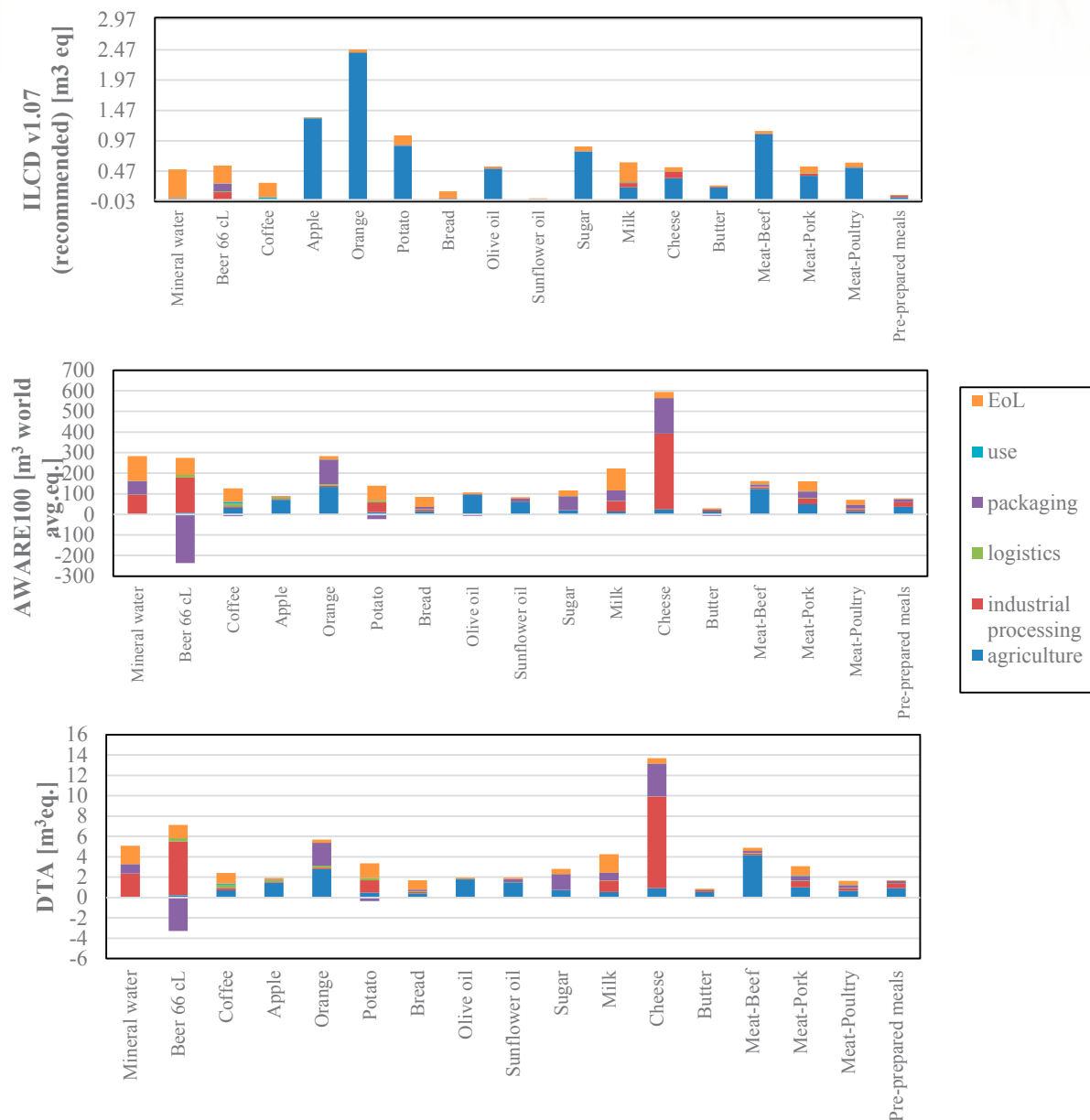
3. Results

3.1 Contribution to water scarcity by product and LCA phase

According to the majority of the LCIA models considered, the highest contributor to the impacts associated to water scarcity is cheese consumption. The only exception to these results is ILCD v 1.07 (recommended version), for which the highest contributors are oranges, apples and beef meat. It is less straightforward to identify the second highest contributor as different models provide different outcomes. The consumed products which are highlighted with the highest frequency are: mineral water, beef meat, oranges and milk. Models which are based on a similar thinking such as the set of Water scarcity model (Boulay et al. 2011), AWARE and DTA models (see Boulay et al., 2016) provide similar results. WSI and Surplus energy (Pfister et al., 2009) and Blue water scarcity (Hoekstra et al., 2012) provide results not dissimilar from those of the AWARE models. Ecoscarcity 2013 and ILCD as implemented in SimaPro provide very similar results as they both rank cheese, beef meat and oranges as the top three most contributing products. It is interesting to note that the consumption of beer leads to negative impacts for 5 models. This is due to the benefits coming from

the recycling of beer packaging at the end of life. The contribution analysis by product and life cycle phases provide similar results across LCIA models. In figure 1 is reported a selection of the most different outcomes.

Figure 1. Contribution analysis by LCA phase for selected LCIA models: ILCD v1.07, AWARE100 and DTA



3.2 Regional and process contribution analysis for selected case studies

3.2.1 Consumption of mineral water

Regional contribution analysis has shown that impacts are principally due to water consumption which had no regional specification for this product (Table 2). For the majority of the LCIA models considered the contribution of this flow to the totals ranges between 15% to 99%, exception made for the method by Berger et al. (2014). The set of elementary flows characterized as 'RoW' contributes significantly to the impacts, ranging from 8% to 58% of the totals, exception made for Yano et al. (2015). These results can be explained by the water balances associated to the flows 'Unspecified' and 'RoW', respectively 81% and 8% of the total. According to the Ecoscarcity (2013) method together with the ILCD version implemented in SimaPro, water consumed in Saudi Arabia contributes

15% to 37% of the total impacts, although the contribution to the inventory of this flow is little above 0%. For a number of LCIA models a significant share of impacts occurs in 'Other countries'.

3.2.2 Consumption of coffee

The regional contribution analysis of impacts associated to EU consumption of coffee (Table 2) has shown that the highest share of water consumed (79%) is not associated to any specific geographic location within the LCA model. As for mineral water, the majority of the LCIA models identify the 'unspecified' flow as the one contributing the most to the total impacts, exception made for WAVE (Berger et al., 2014). The country in which the most of impacts occurs is China, according to the majority of the models. The only exception is Ecoscarcity 2013, which presents the most significant impact occurring in Saudi Arabia. The second most relevant contributor is the aggregated flow "Other countries". Quite a high variability amongst models is observed for the other geographic locations.

3.2.3 Consumption of sugar from beet

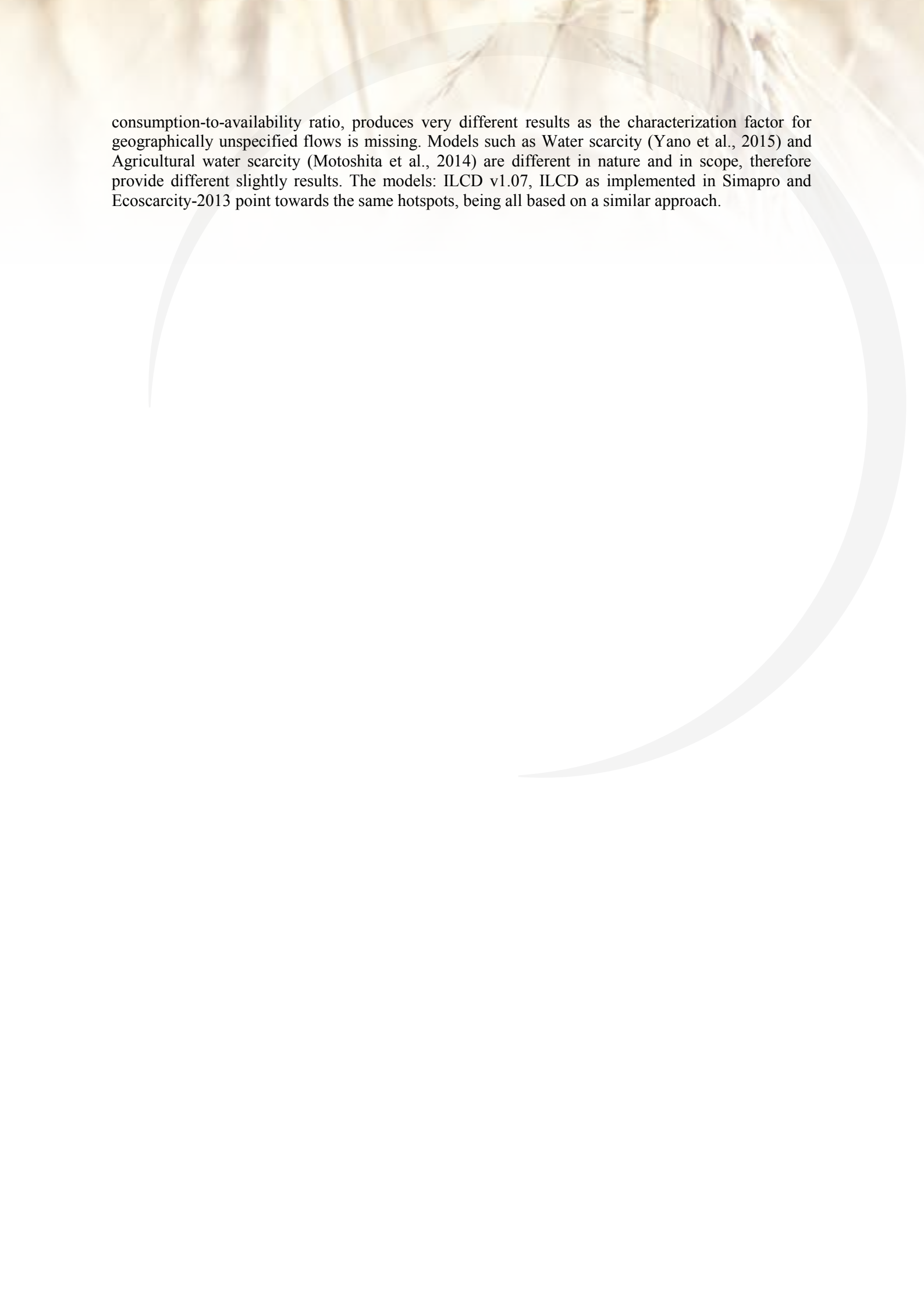
The regional contribution analysis of impacts associated to EU consumption of sugar from beet (Table 2) has shown that almost 62% of the water balance in mass is not associated to any specific region, whereas 9% is allocated to the Ecoinvent 'Rest-of-the-World' regions. In this case the majority of the impact is related to RoW, unspecified and other countries. China and USA are identified as significant contributors for a number of models. An exception is made for both the ILCD models and Ecoscarcity 2013, which identify Germany and Saudi Arabia as hotspots, due to the high value of the characterization factors for these two countries.

3.2.4 Consumption of beef meat

Results for this product have shown that although the vast majority of water consumption (77% of the inventory) does not have a geographic specification (Table 2). The majority of the impacts are due to processes occurring in USA and Pakistan (ILCD v1.07, ILCD as in Simapro, Ecoscarcity 2013, Agricultural water scarcity (Motoshita et al., 2014)), or, for a lower number of models (DTA and DTA_x), in Ireland.

4. Discussion

First of all, the contribution analysis of the product composing the European Basket of Food shows that a number of models agree in the identification of the top three products of large consumption generating the majority of the impacts associated with water scarcity (Table 3). However, the selection of models is not balanced amongst models typologies. The most different results are obtained from ILCD v 1.07 (recommended version) which only characterizes withdrawals and not releases, therefore this it is expected to provide results which are different from all other models, as observed in Figure 1. From the regional contribution analysis, it can be observed that for the totality of the considered case studies the flows with 'unspecified' geographic location dominate the underlying inventory. This stems from the use of background inventories which do not provide any specific regionalization (e.g. ELCD EU27 average electricity mix) or from the use of foreground data which are not regionalized. Nevertheless, the share of impacts which has geographic information associated to it is relatively high, although only in very few cases impacts (i.e. beef meat) occurring in a specific country are contributing to >50%. Therefore, the geographically 'unspecified' flows, and the underlying average characterization factors – normally the most uncertain ones, become those driving the results the most. The model AWARE100, recommended as interim by the UNEP/SETAC life-cycle initiative, provides results comparable with its alternative variations (AWARE100_EWR50; AWARE10; AWARE1000) as well as alternative models considered within the WULCA initiative (DTA and DTA_X). Results are also similar to those obtained with a large number of models both based on consumption-to-availability ratios such as Blue water scarcity (Hoekstra et al., 2012) and Water scarcity (Boulay et al., 2011), as well as withdrawal-to-availability ratios, such as WSI and Surplus Energy (Pfister et al., 2009). More relevant differences are expected to occur when LCA models with higher regionalization are characterized, especially if foreground analyses are performed at the level of watersheds. The model WAVE (Berger et al., 2014), although being based on a



consumption-to-availability ratio, produces very different results as the characterization factor for geographically unspecified flows is missing. Models such as Water scarcity (Yano et al., 2015) and Agricultural water scarcity (Motoshita et al., 2014) are different in nature and in scope, therefore provide different slightly results. The models: ILCD v1.07, ILCD as implemented in Simapro and Ecoscarcity-2013 point towards the same hotspots, being all based on a similar approach.

5. Conclusions

The results have demonstrated that a majority of LCIA methods for assessing water scarcity agree in identifying the top 3 most contributing products to the impacts associated with EU consumption patterns (mineral water, beef meat and oranges) as well as on the most relevant life cycle phases, exception made for the ILCD v1.07 model and few other diverging results. The applicability of the AWARE model and its alternative versions is proved and documented as no major inconsistencies are found and the results are substantially compatible to those obtained from other models. Nevertheless, such similarity may be partially due to the low level of regionalization which is provided by the European Food Basket in its current version.

Future developments of the underlying LCA model should improve the geographic specification of processes, as regional variability is a key aspect in water scarcity assessment.

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235. The WULCA consensus for water scarcity footprints: Assessing impacts of water consumption based on human and ecosystem demands

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ABSTRACT

The need for consensus on water footprint indicators used in LCA led to the Water Use in LCA (WULCA) working group of the Life Cycle Initiative to receive the mandate in 2013 to build such a consensus, in line with the newly developed standard on water footprint ISO 14046:2014. The result of this two-year project is the AWARE (Available WATER REmaining) method for water scarcity footprint, assessing the potential to deprive other users of water when using water in a given area. It provides an indicator, ranging between 0.1 and 100, based on the relative amount of water remaining per surface area, once the demand for humans and ecosystems has been met, compared to the world average. This paper discusses specifically the spatio-temporal variation and associated interpretation of this new method.

Keywords: water use impact, scarcity index, water deprivation

1. Introduction

In May 2013, WULCA received the mandate from the UNEP SETAC Life Cycle Initiative (Project on Global Guidance on Environmental life cycle impact assessment indicators) to lead the consensus building for a harmonized method assessing water use impacts in LCA. WULCA set out to fill this mandate in the two-year time frame 2014-2015. The method developed was presented and approved as a recommendation by the Life Cycle Initiative in January 2016. In that time frame, two journal articles were published (Boulay et al. 2014; Boulay et al. 2015) describing the process and the reasons for moving towards a new indicator, and the final deliverable is currently under review (Boulay et al. 2016). The present paper presents briefly the method and discusses in more details the spatio-temporal variability and associated interpretation and applicability of the method.

2. Methods

The AWARE method represents the relative Available WATER REmaining per area in a watershed, after the demand of humans and aquatic ecosystems has been met. It assesses the potential of water deprivation, to either humans or ecosystems, building on the assumption that the less water remaining available per area, the more likely another user will be deprived (Boulay et al. 2016).

It is first calculated as the water Availability Minus the Demand (AMD) of humans and aquatic ecosystems and is relative to the area ($\text{m}^3 \text{m}^{-2} \text{month}^{-1}$). In a second step, the value is normalized with the world average result ($\text{AMD}_w = 0.0136 \text{ m}^3 \text{m}^{-2} \text{month}^{-1}$) and inverted, and hence represents the relative value in comparison with the average cubic meter consumed in the world (the world average is calculated as a consumption-weighted average). The indicator is limited to a range from 0.1 to 100, with a value of 1 corresponding to a region with the same amount of water remaining as the world average, and a value of 10, for example, representing a region where there is 10 times less available

water remaining per area than the world average. Equation 1 below shows the calculation of the characterization factors, for values of the denominator (AMD) ranging between 0.136 and 0.000136 m³ m⁻² month⁻¹ (corresponding to 10 times more or a 100 times less than AMD_w). Larger and lower values are set as maximal and minimal in the CF set at 100 and 0.1 respectively. The effect of these cutoff choices are quantified in Boulay et al. (2016).

$$CF = \frac{AMD_w}{(Availability - Human Consumption - Ecosystems demand) / m^2 \cdot month}$$

Equation 1

The Availability represents the renewable water available (annual surface run-off and groundwater recharge), and is taken from the WaterGap model averaged over a 50 year timeframe (1960-2010) (Müller Schmied et al. 2014). The data are provided on river basin level, with the world's largest 34 river basins being divided into sub-basins. Availability is provided for the most downstream cell of each (sub-)basin. Human consumption is also taken from Watergap (Florke et al. 2013), representative of year 2010, and includes domestic, manufacturing, electricity, irrigation and livestock sectors. Ecosystem's demand is assessed using the Variable Flow Method (VMF) (Pastor et al. 2013). The VMF uses algorithms to classify flow regime into high, intermediate and low-flow months to take into account intra-annual variability by allocating environmental flow requirements with a percentage of mean monthly flow between 30-60%, which should be maintained to preserve aquatic ecosystems in "fair" condition with respect to pristine state. Sensitivity of this choice is discussed in (Boulay et al. 2016).

The indicator is calculated at the sub-watershed level and monthly time-step, and whenever possible, this level of resolution should be used. However, in LCA this can be challenging and different aggregation approaches are proposed based on the level of information available. First, since the characterization factors are aggregated, to country and/or annual resolution, when both of these resolutions are not available to the practitioner, data to reduce uncertainty on the most variable one should be chosen. The variability associated with the temporal and spatial aggregation has been quantified using a standard deviation, and the most variable aspect is identified in order to guide the user towards the additional information that should be collected in order to reduce the uncertainty. When aggregation on one or both of these aspects cannot be avoided, this aggregation can be done in different ways to better represent the spatial or temporal pattern for the water use being assessed. Agriculture uses water in different regions and months than industrial and domestic uses. Characterization factors for agricultural and non-agricultural use are therefore provided, as well as default ("unknown") ones if the activity is not known.

3. Results

Characterization factors for the AWARE method are presented in Figure 1, both at the (sub)watershed level and at the country level, showing the different types of user aggregation: non-agricultural, agricultural and unknown.

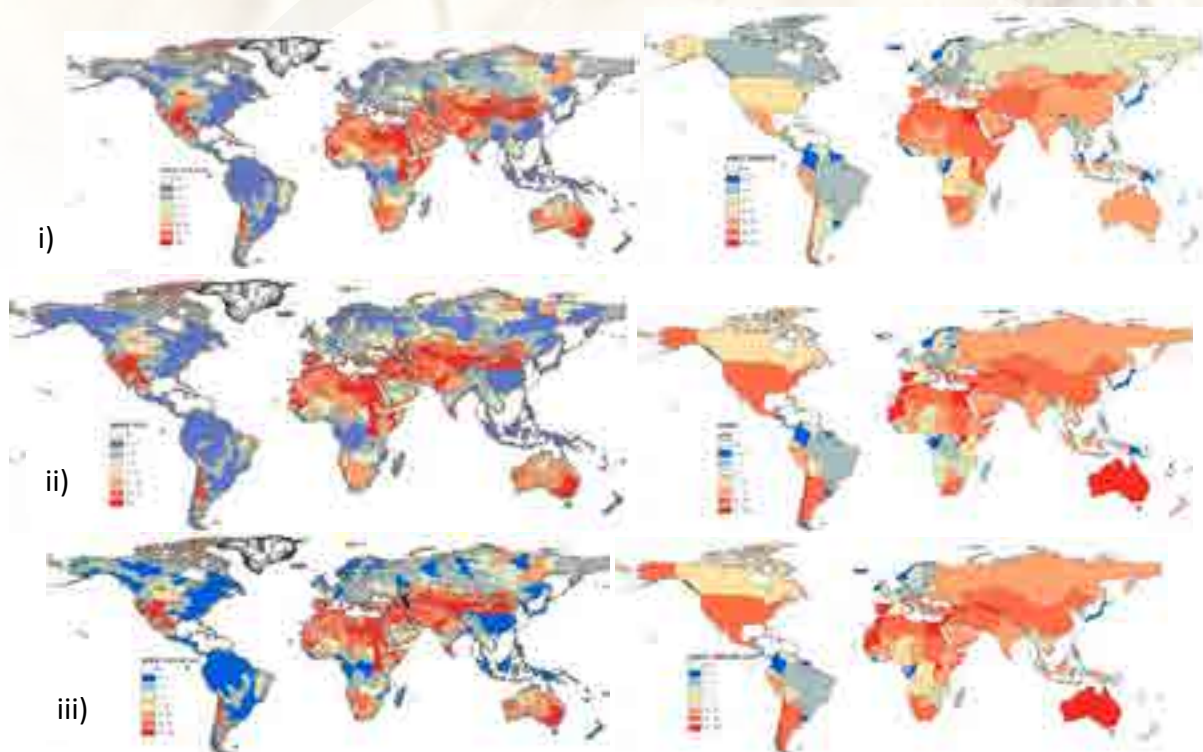


Figure 1: AWARE characterization factors, annual, shown for (sub)watershed level (left) and country level (right), aggregated annually based on i) non-agricultural users, ii) agricultural users and iii) unknown user (all water consumption).

It should be noted that a factor value of 1 is not equivalent to the factor for the average water consumption in the world, i.e. the world average factor to use when the location is not known. This value is calculated as the consumption-weighted average of the factors, which are based on $1/AMD$ and not AMD directly, hence the world consumption-based average CF has a value of 43 for unknown use and 20 and 46 respectively for non-agricultural and agricultural water consumption respectively. In general, spatial variation within a country is larger than temporal variation, and more efforts should be invested in increasing geographical resolution, i.e. finding out the watershed where the water use assessed is occurring. However this is not true for all countries, as shown in Fig.2. For some countries, including Chile, Morocco, Algeria, Tunisia, Pakistan, Saudi Arabia, Indonesia, etc, obtaining information on the month of the water consumption would decrease the spatio-temporal uncertainty more than improving the geographical information.

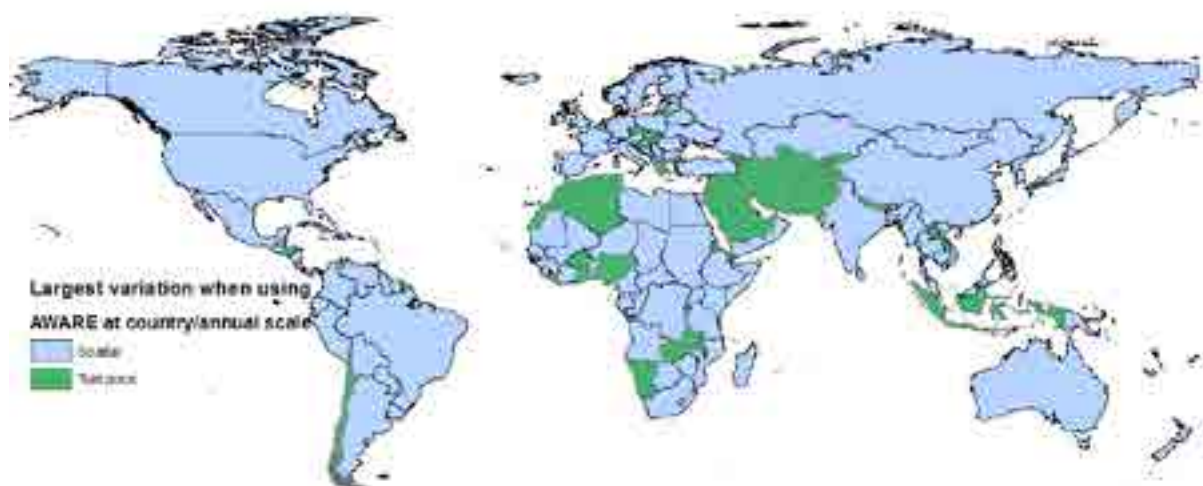


Figure 2: Identification of the most variable aspect for uncertainty reduction at the country/annual level.

While a geographical representation of the CF can be useful, as shown in Fig. 1, in order to gain a better understanding of the world's pattern in water consumption, it is informative to distribute the water consumed in the world based on the CF, as shown in Fig.3. On one side (Fig.3a), this is done using the monthly water consumption and the monthly CF, whereas in the other side (Fig.3b), it is done on the annual level. In both cases, a (sub) watershed level is used (total of 11050 basins).

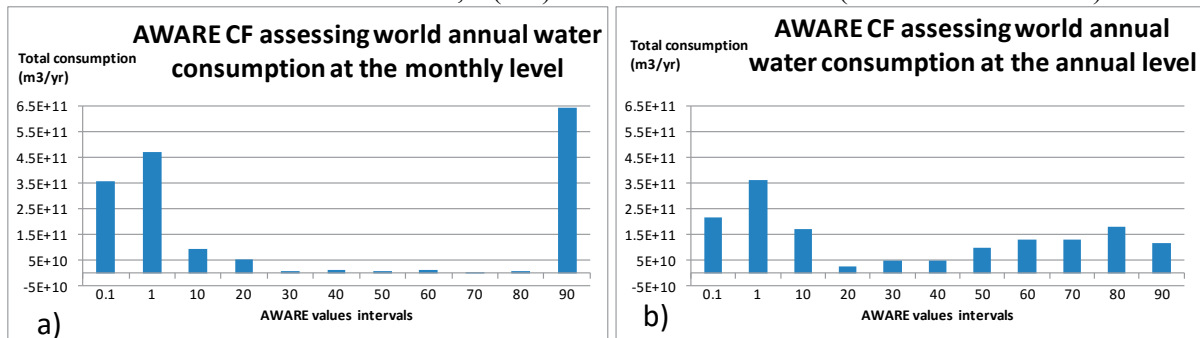


Figure 3: AWARE CF assessing the world annual water consumption, based on a) monthly indicators or b) annual indicators.

4. Discussion

Figure 1 shows the large variation of results within a country which is lost when using country scale results. It also shows that “agri” and “unknown” type of aggregation at the country level are very similar, while non-agri differs. This is because agricultural water use is dominant in most of the world and hence the unknown user type of aggregation is mostly influenced by irrigation. At the (sub)watershed level, this distinction is less obvious since it is only based on temporal variation, which is confirmed by Figure 2, showing that temporal variation is often less in comparison to spatial variation.

When interpreting the CF presented, it should be kept in mind that an aggregated value at country/annual level based on consumption:

- 1) Does not represent the “average picture” of the country/year. It may completely exclude large regions where no/very low consumption occur (i.e. deserts, most of Canada, etc.).
- 2) Is strongly influenced by agricultural water use (in both “unknown” and “agri” values)
- 3) Represents where/when water is most consumed: often in dryer months/regions

For these reasons, if a value seems not representative of the situation assessed, the spatio-temporal resolution should be increased.

From Figure 3, it can be seen that at the native resolution of the model, monthly and watershed scale, most consumption in the world occurs in region with either high values (90-100) or low values (0.1 – 10). This is consistent with findings of previous methods which also obtained binary results when assessing scarcity (Boulay et al. 2011). This can be simplistically explained by the fact that either a region has a lot of water and hence a reduced potential of deprivation when using water in this region, or it has less water and consequently also higher needs for irrigation, hence a stronger competition and potential for deprivation. This is true at the monthly level, and one region may be in one situation one month and in the opposite the following month when monsoon arrives for example. When aggregating at the annual level, this binarity is less prominent and the distribution among all values from 0.1 to 100 is more evened out. Ultimately, this gradation is hence directly linked to the number of months that the region faces a high user competition or potential for deprivation.

5. Conclusions

The LCA community can now benefit from a consensus-based generic method to assess water use impact assessment at the midpoint level as well as perform a water scarcity footprint. While a high spatio-temporal resolution is optimal at the impact assessment level, recommended at the (sub)watershed and monthly scale, it is often impractical at the inventory level over the entire life cycle. For this reason, different recommendations are provided to reduce the uncertainty associated with larger resolution. In general, spatial resolution should be prioritized of temporal resolution, unless occurring in some specific countries identified in this work. When using a larger resolution (year and/or country level), aggregation of native CF based on consumption pattern of specific user reduces uncertainty, but also represents regions/time of higher water consumption and may misrepresent specific cases occurring outside of this pattern.

While this method is already applicable at the country and watershed scale level, it is designed for marginal applications, i.e. water consumption that is marginal in comparison to the background. Since the concept of water footprint has been used in the past at the country level to assess, for example, the water footprint of a nation, specific characterization factors should be developed to address non-marginal water consumption.

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13. Addressing the water use of a medium-sized wine company: the case of Portuguese ‘vinho verde’

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ABSTRACT

This study is devoted to the water scarcity footprint of a Portuguese wine (the white ‘vinho verde’) using the current midpoint life cycle assessment (LCA)-based methods: Pfister et al. (2009), Frischknecht et al. (2009) (ESM), Ridoutt et al. (2010), and Milà i Canals et al. (2009). The strengths and constraints of each LCA-based method are addressed and the wine production stages that mostly contribute to the water scarcity footprint are identified. The water footprint profile, i.e. the compilation of quantitative and degradative environmental impacts related to water use is also addressed.

The same functional unit (one bottle of ‘vinho verde’ with a total volume of 0.75 L) lead to different water scarcity footprints: 1.0, 1.6 and 18.7 L eq, following the Pfister et al. (2009), Ridoutt et al. (2010), and Milà i Canals et al. (2009) methods, respectively, and 1.2 eco-points by using the ESM method. A large variability of these results is mainly due to different modelling choices and characterisation factors considered by each method.

The results obtained by Pfister et al. (2009) and Ridoutt et al. (2010) identified the energy carriers from viticulture background sub-system and from the wine production background sub-system as the unit processes that contribute most to the water scarcity footprint (40 % and 38 %, respectively). Concerning the ESM method, the energy carriers of the wine production background are responsible for more than 40% of the water scarcity footprint. Following the Milà i Canals et al. (2009) method, the viticulture foreground is responsible for more than 95 % of the water scarcity footprint. The viticulture stage is also the main hotspot in the degradative impacts. The viticulture foreground sub-system is particularly important to eutrophication, with a contribution higher than 80 % due to emission of nitrate resulting from the use of fertilisers.

This study contributes to the on-going debate about the LCA-based methods to be used to assess the impact derived from water use. The high contributions of the background sub-systems show the relevance of widely include the water use in LCA databases, avoiding duplication in data compilation and allow a spatially explicit water scarcity footprint.

Keywords: life cycle assessment, water scarcity footprint, quantitative and degradative environmental impacts

1. Introduction

In many regions, human activities have been increasing water stress (due to water appropriation and quality degradation), leading to competition for its use and resulting in a loss of freshwater functionality for downstream users (Bogardi et al. 2012; Falkenmark and Rockström 2004).

Over the last few years, several methods have been developed to quantify the water scarcity footprint from a life cycle perspective (e.g. Bayart et al. 2010; Boulay et al. 2011a,b; Frischknecht et al. 2009a; Milà i Canals et al. 2009; Pfister et al. 2009; Ridoutt et al. 2010). These LCA-based methods have also been applied to agro-industrial products such as pasta sauce and peanut (Ridoutt et al. 2009), broccoli (Milà i Canals et al. 2010), asparagus and tomato (Frischknecht et al. 2006), among others. The LCA community has been recommending their application in case studies in order to understand the individual significance of each one (Kounina et al. 2013). This study focus on the production of a Portuguese white ‘vinho verde’ by the largest producer of ‘vinho verde’ in Portugal, Aveleda S.A. The wine sector assumes a relevant role in Portugal, as the country is among the 12 leading countries for wine worldwide exportation.

The goals of this study are:

1) to assess the water scarcity footprint a Portuguese white ‘vinho verde’ using the LCA-based methods developed by Pfister et al. (2009) (from now on referred to as the Pfister method), Frischknecht et al. (2009a,b) (Ecological Scarcity Model (ESM) method, from now on referred to as the ESM method), Ridoutt et al. (2010) and Ridoutt and Pfister (2013) (from now on referred to as the Ridoutt method), and Milà i Canals et al. (2009) (from now on referred to as the Milà i Canals method);

2) to carry out the water footprint profile, i.e. the compilation of quantitative and degradative environmental impacts related to water use, throughout the white ‘vinho verde’ life cycle (from cradle to gate);

3) to identify the production stages and sub-systems that mostly contribute to the water footprint profile.

The degradative impact related to water use to produce the 'vinho verde' under study was evaluated by Neto et al. (2013), in which eutrophication, freshwater aquatic ecotoxicity, marine aquatic ecotoxicity and acidification impacts were considered.

2. Methods

The various LCA-based methods differ significantly concerning the type of water, water scarcity level and characterisation factors (CFs).

The Pfister method assesses the environmental impact of blue water consumption. This method proposes water scarcity indexes (WSIs) estimated at the sub-watershed level as midpoint CFs. The WSI is a modified withdrawal-to-availability ratio that accounts monthly and annual variability in precipitation influence. It is calculated using the WaterGAP2 global hydrology and global freshwater models (Alcamo et al. 2003; Marker et al. 2003).

The ESM method is based on the distance-to-target principle (i.e. comparison of existing emission flow and target flow) and provides eco-factors (CFs) for multiple environmental impacts. To assess the impact of blue water consumption, the ESM method provides eco-factors at country level, which are calculated based on withdrawal-to-availability ratios (Frischknecht et al. 2009a,b). Unlike the Pfister method, it does not account for monthly and annual variability in precipitation and, consequently, may overestimate or underestimate water scarcity. In this study, we used the eco-factor available for Portugal, $0.260 \text{ eco-points.L}^{-1}$ of blue water (Frischknecht et al. 2009a).

The Ridoutt method assesses the environmental impact of green and blue water. Ridoutt et al. (2010) suggested that the use of green water in land use activities, per se, does not necessarily leads to a reduction in surface water and groundwater or contribute to water scarcity. As such, this method suggests to estimate land use effects on water availability as the difference between the evapotranspiration from the crop under analysis and the evapotranspiration of a reference land use. The question of what consider as reference land use remains (Chiarucci et al. 2010; Hickler et al. 2012; Loidi et al. 2010). the Ridoutt method considers herbaceous vegetation as reference land use whereas the Milà i Canals method considers forest land use as reference land use. Further research is needed to give scientific consistence to this procedure. The impact assessment is performed using midpoint CFs that are the same as the regional WSI developed by Pfister et al. (2009) divided by the global average WSI as defined by Ridoutt and Pfister (2013).

The Milà i Canals method evaluates the influence of blue and green water, as well as land use changes, on water scarcity at the watershed level. The green water is only considered at the inventory phase because no environmental mechanism to evaluate its impacts is provided. Therefore, at impact assessment level, only blue water consumption and land use changes effects are assessed. In this study, we used a water use per resource (WUPR) CF of 0.164 as suggested by Milà i Canals (2009).

2.1. Scope, inventory and impact assessment

This section presents the functional unit (FU), describes the system boundary and explains how inventory data, impact assessment and water footprint profile were obtained.

2.1.2 Functional unit and description of the system

In the current study, the FU is defined as the volume of one bottle of white 'vinho verde' with a total liquid volume of 0.75 L.

The system boundary, schematically presented in Figure 1, includes two stages: the viticulture processes at the vineyard level and the wine production processes at the winemaking plant. Quinteiro et al. (2014) provides a full description of the system under study.

The production of ancillary materials was considered in the viticulture and wine production background sub-systems. However, all the transportation operations as well as the production of cork stoppers, labs and caps have been excluded from the system boundary.

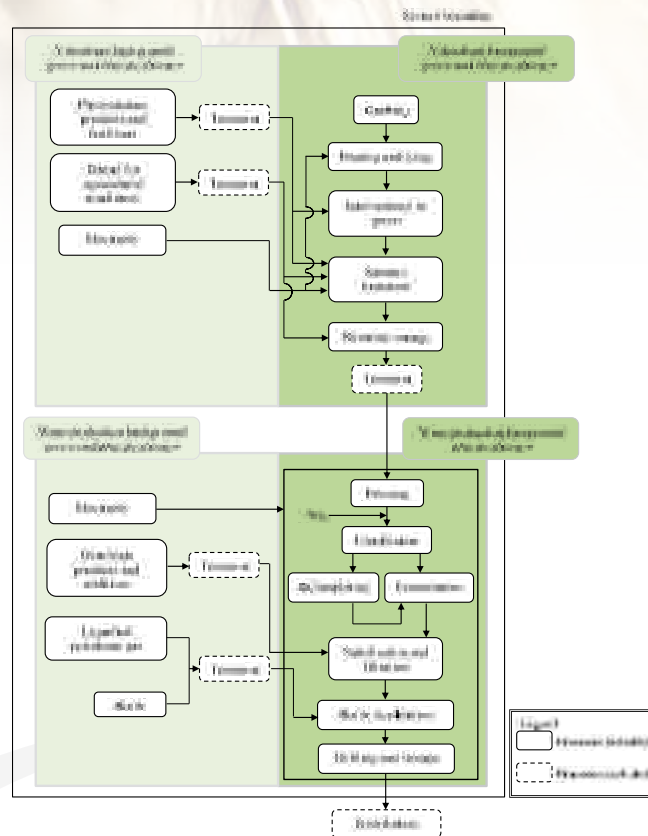


Figure 1. System boundary of the white ‘vinho verde’.

2.1.3. Inventory

Regarding the viticulture foreground sub-system, and since no irrigation systems were present, the blue water only includes the amount of water used for spraying during phytosanitary treatments. It was assumed that this amount of water completely evaporates into the atmosphere.

The total green water of the viticulture foreground sub-system was obtained by determining the total amount of green water evapotranspiration that occurred throughout the entire growing period of the grapes, plus the green water fraction incorporated into the harvested grapes. The green water evapotranspiration was calculated using the CROPWAT 8.0 model and the irrigation schedule as ‘no irrigation (rain-fed)’ (FAO 2009). Input data for the model included information on climate factors, red sandy loam soil properties and vine characteristics. The green water fraction incorporated into the harvested grapes was estimated based on a moisture content of 75 % in the grapes.

The land use effects on freshwater availability according to Milà i Canals method were calculated based on the change in rainwater availability for land infiltration and runoff in relation to forest (reference land use). In the absence of specific data for vines, both data from ‘lost’ percentage of precipitation of arable non-irrigated land in Europe (73 %) and ‘lost’ percentage of precipitation of forested land (67 %) (reference land use) were used, as suggested by Milà i Canals et al. (2009).

The Ridoutt method also requires the calculation of the impact of land use on blue freshwater resources (green water that is accessible only through land occupation) in relation to herbaceous vegetation (reference land use). The green water evapotranspired from this vegetation was calculated using Zhang et al. (2001) equation, as suggested in the Ridoutt method.

Regarding the wine production foreground sub-system, it only considers the evaporation component of blue water, which occurs during de-sulphiting and bottle sterilisation, since there is no incorporation of freshwater into the wine.

For both viticulture and wine production background sub-systems, data of water were taken from the GaBi professional database (PE International 2012).

Further details on the inventory establishment can be found in Quinteiro et al. (2014).

2.1.4. Water footprint profile

The water scarcity footprint was assessed by multiplying the volume of blue water and land use effects collected at the inventory level by the CFs of each applied LCA-based method.

The degradative component was obtained from Neto et al. (2013) that carried out a conventional LCA for the Portuguese white ‘vinho verde’ under analysis. The degradative impact assessment was carried out according to the CML 2001 methodology considering the same system boundary and FU of the current study. Eutrophication (EP), freshwater aquatic ecotoxicity (FE) and marine aquatic ecotoxicity (ME) were the considered impact categories related to water. The water use (WU) impact was calculated accordingly to the four water use LCA-based methods analysed in this study.

3. Results

3.1. Water scarcity footprint

The results obtained with the Pfister, ESM and Ridoutt methods are different although they rely in the same FU (one bottle of ‘vinho verde’ with a total volume of 0.75 L).

Based on the Pfister method, the water scarcity footprint was 1.0 Leq, with the viticulture being responsible for 56 % of the total impact. With the ESM method, water scarcity footprint was 1.20 eco-points. In this case, the wine production stage was the most relevant, being responsible for 76 % of the water scarcity footprint.

The relative contributions to the water scarcity footprint obtained with the Ridoutt method are similar to those obtained with the Pfister method, although the water scarcity footprint is higher. Although the Ridoutt method considers the land use effects at the inventory level, they were considered to have no impact on water resource availability, as recommended by Ridoutt et al. (2010), because they were smaller than zero.

Following the Milà i Canals method, the viticulture foreground sub-system is responsible for more than 95 % of the water scarcity footprint due to the contribution of the land use effects. The water scarcity footprint calculated with this method was the highest compared with the other methods (18.7 Leq), although the total blue water use impact is the smallest one (0.8 Leq).

Figure 2 shows the relative contributions to the water scarcity footprint of each unit process/system considered in the white ‘vinho verde’ life cycle. Regarding the Pfister and Ridoutt methods, the energy carriers from viticulture background sub-system are the unit processes with the largest relative contribution to the overall water use impact (40 %), while the energy carriers from the wine production background sub-system appears as the second largest contributors (38 %). Production of phytosanitaries and synthetic fertilisers accounts for 7 % of water scarcity footprint.

Concerning the ESM method, the energy carriers of the wine production background sub-system are responsible for more than 40 % of the water scarcity footprint.

3.2. Water footprint profile

The relative importance of water scarcity footprint and the degradative environmental impacts related to freshwater – EP, FE and ME impact categories – is illustrated in Figure 3.

The viticulture stage is the major contributor to the water scarcity footprint, except for the ESM method, for which the major contributor is the wine production stage. The viticulture stage is also the main hotspot in the degradative impacts contributing with 92, 72 and 68 % to EP, FE and ME, respectively. The viticulture foreground sub-system is particularly important to EP, with a contribution higher than 80% due to emission of nitrate resulting from the use of fertilisers, whereas the viticulture background is the sub-system that most contributes to FE and ME with 58 and 60 % of the total impacts, respectively. Barium and polyaromatic hydrocarbons emitted during the production of diesel, as well as formaldehyde and zinc emitted during the production of phytosanitary products, explain this high percentage in the FE impact. The impacts related with ME are mainly caused by transition metals used in the production of phytosanitary products and electricity.

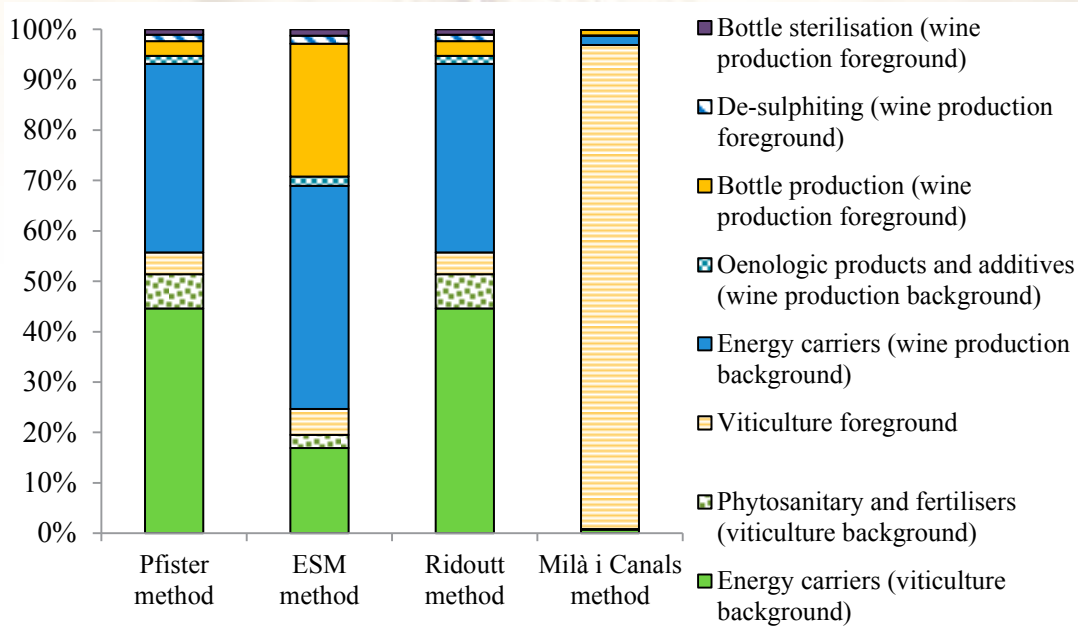


Figure 2. Relative contributions to water scarcity footprint of each unit process and sub-system considered in the white 'vinho verde' life cycle.

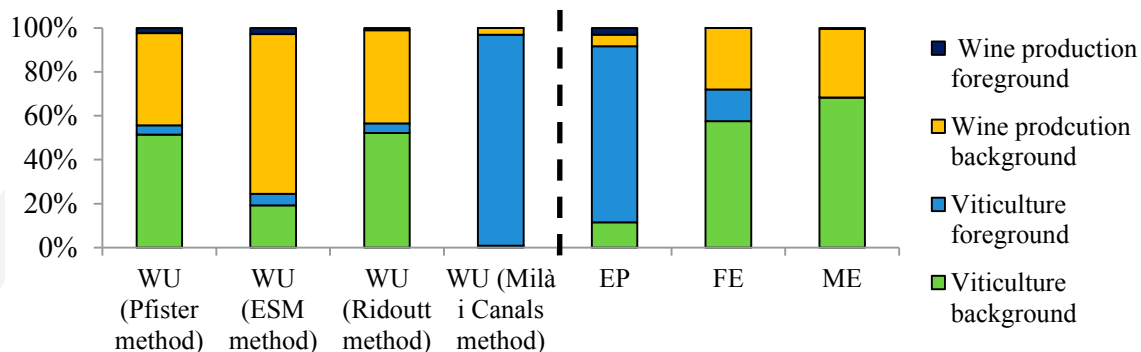


Figure 3. Water footprint profile of the Portuguese white 'vinho verde'. WU acronym: water use.

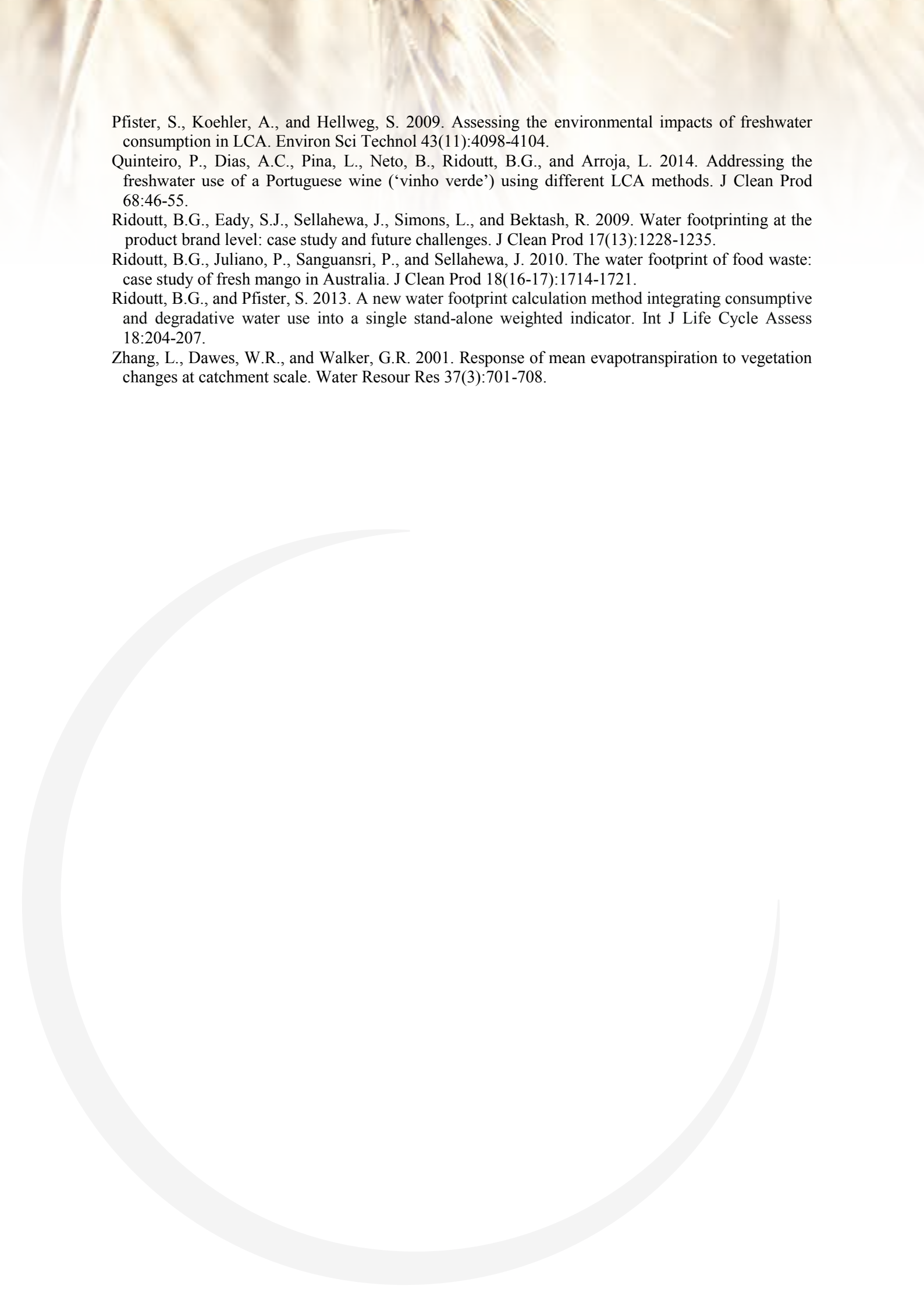
5. Conclusions

This study contributes to the on-going debate about the methods to be used to assess the impact derived from water use under the LCA framework.

The production stages and sub-systems that mostly contribute to the water footprint profile also differed depending on the water scarcity method and water degradation impact category. The Pfister and Ridoutt methods, as well as the FE and ME impact categories, identified the viticulture background sub-system as the major hotspot. On the other hand, the ESM method identified the wine production background sub-system as the major hotspot, whereas viticulture foreground sub-system appeared as the major hotspot using the Milà i Canals method and for EP impact category. The relevant contribution of these background sub-systems indicates that it is important to widely include the water use into LCA databases to facilitate the life cycle inventory, avoid duplication in data compilation and allow a reliable and representative water footprint profile of products. Moreover, further research is needed to identify and plan actions to reduce both quantitative and degradative water use impacts for the identified hotspots.

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165. Evaluating the suitability of three water scarcity footprint methods: case study for the Spanish dairy industry

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ABSTRACT

Water footprints are becoming an important metric used by the business sector to identify environmental performance improvement opportunities and to report performance to stakeholders. This has been supported by the publication of a new International Standard on the subject (ISO 14046). Presently, the greatest interest is in water scarcity footprints which specifically address consumptive water use. However, right now the comparability of different water scarcity footprint results is inhibited by the use of different characterization factors for local water stress. Using a case study based on dairy production in northern Spain, water scarcity footprints were calculated using three alternative methods (revised WSI, WSI_{HH_EQ} and AWaRe). The results varied significantly, from 2.8 to more than 3,000 L H₂Oe/kg product. In the near term, operators of product water footprint programmes or initiatives should specify the particular water stress factors to be applied. This will enable comparability of product water footprints within a programme. However, over time, effort should also be made to achieve consistency between programmes, and this study discusses some important considerations such as the reference unit and range.

Keywords: LCA, Aware, WSI, ISO14046, sustainable consumption and production

1. Introduction

According to the European Science and Technology Observatory the food industry is one of the activities that contribute to the impact on the environment (Tukker et al, 2006). Therefore, both European policymakers and consumers are demanding more sustainable products with the environment. Moreover, agriculture must feed another 2 to 3 billion people in the next 50 years, putting additional pressure on water resources. More than 70% of the world's 850 million undernourished people live in rural areas, and most depend directly or indirectly on water for their livelihoods.

According to the European Environment Agency (EEA, 2009) water scarcity occurs where there are insufficient water resources to satisfy long-term average requirements. It refers to long-term water imbalances, combining low water availability with a level of water demand exceeding the supply capacity of the natural system. Water resources include surface water (stream, river, lake and reservoirs) and groundwater. Although water scarcity often happens in areas with low rainfall, human activities contribute to the problem, especially in areas with high population density, tourist inflow, intensive agriculture and water demanding industries.

Water use efficiency (WUE; water use per unit process) has been widely used as a performance indicator in regards to water use. It is a measure of the technical efficiency of a factory or of an appliance (e.g. a washing machine). Traditionally water measurement has been focus to identify potential cost saving in the production chain.

However WUE does not make sense from a life cycle perspective as there is no impact characterization when aggregating water of different types from locations of different water scarcity levels. As such, WUE cannot be meaningfully applied to the life cycles of products and services. In 2014, the International Organization for Standardization addressed this problem with the publication of an International Standard for water footprinting (ISO 14046). This was a major development, providing a consistent framework for businesses to evaluate and report impacts related to water use from a life cycle perspective. ISO14046 also provides a basis for businesses to make water footprint claims with respect to products and services, akin to the product carbon footprint claims (i.e. labels and declarations) which have begun to appear in many jurisdictions.

ISO14046 includes within its scope both water consumption and pollution. When only water consumption is evaluated the term water scarcity footprint is recommended. A limitation at present is that ISO14046 does not specify a particular characterization model for water scarcity. This is not such a problem for LCA studies designed to provide strategic insights for environmental improvement, as the focus is generally upon identification of hotspots and it is the relative size of the contribution from different life cycle stages that matters. However, for water footprint claims where there is communication to stakeholders in society (including consumers), the absolute numbers do matter.

Water use (WU) in dairy production is a matter of discussion all over the world because water scarcity and water pollution are issues of concern in a large number of regions (Sultana et al, 2014b). Moreover the use and also the quality of water on farms and in milk and dairy processing plants are significant focus on dairy producers and processors (Matlock et al., 2012; Henderson et al, 2012).

Within this framework a water scarcity footprint study was carried out for a large dairy producer located in the north of Spain. Since there is not one commonly agreed method for the evaluation of the local water stress, three different methods were tested and compared. It is important to highlight that the water pollution or quality lost has not been accounted in this case study.

2. Methods

Within the Life Cycle Assessment approach several methods have been published to quantify the environmental impacts of water use (Kounina et al., 2013). In relation to consumptive water use, among the most relevant and recent methods are the following:

- The revised Water Stress Index (WSI) developed by Ridoutt and Pfister (2010) as recommended by the European Food Sustainable Consumption and Production Roundtable in the ENVIFOOD Protocol (Food SCP Rt, 2013). This index utilizes the regionalized withdrawal-to-availability based factors reported by Pfister et al. (2009), but with moderation by the global consumption weighted average WSI. In this way, water scarcity footprint results are reported in units of H_2O_e (equivalent) where the reference situation is water consumption at the global average WSI.
- The regionalized WSI of Ridoutt and Pfister (2014) that integrates separate models for water stress on humans and ecosystems. In some regions, the human health impacts of water stress are moderated by the importation of water intensive goods, such as food, as well as investments in alternative water supply technologies and water use efficiency measures. As such, the water withdrawal-to-availability ratio can be a poor proxy for environmental harm. This alternative index (WSI_{HH_EQ}) overcomes this problem and delivers results which are consistent with LCA endpoint models for water use (Pfister et al., 2009).
- Aware (Available Water Remaining) method (<http://www.wulca-waterlca.org/>): This new index is the preliminary recommendation by the UNEP-SETAC Life Cycle Initiative's project group on water use (<http://www.lifecycleinitiative.org/>).

2.1. Case study: Water footprint of dairy production in northern Spain

The selected functional unit or target of the study was the processing of 1 kg of raw milk at a dairy company located in Asturias (North of Spain). The system boundary was from cradle to gate, taking into account the production of feed from raw materials to the processing of milk into dairy products.

Direct water resources used for feed crop irrigation, farm operations and dairy plant operations were taken into account. The farming subsystem, which was a conventional system supplemented by pasture and purchased feed, was modeled using data for 54 farms providing raw milk to the dairy plant. The feed used was a conventional feed for milking cows, prepared by 45 % of maize produced in Spain, 12 % of Barley produced in Spain, and 10 % of soy imported from Argentina and Brazil among other supplements. These data covered the 2013 financial year. The dairy factory subsystem was modeled using data provided by the plant engineer. This system includes inputs of reticulated town water, the recovery of water from the evaporator, on-site wastewater treatment, selected water

reuse, and the discharge to a local stream of freshwater, which is regarded by the local water management agency as an environmentally beneficial flow.

Additionally, water evaporation from hydropower reservoirs was taken into account, using evaporation rates given by the Spanish Ministry for Agriculture, Food and Environment for each region in Spain. In this dairy product case study, so-called green water (soil moisture from natural rainfall) consumed through pasture evapotranspiration was not included in the inventory of water use.

Background inventory data for the crop cultivation, electricity generation and water supply datasets were taken from Ecoinvent and Agri-Footprint (Blonk Agri-footprint BV, 2014) databases. In addition, emissions to freshwater from fertilizers, pesticides, and industrial processes were not included because the environmental impacts are generally considered under other LCA impact categories such as eutrophication and freshwater ecotoxicity and do not form part of a water scarcity footprint.

To calculate the water footprint, each instance of consumptive water use was multiplied by the relevant WSI and then summed across the product life cycle.

3. Results

The production of 1 kg of raw milk in Asturias, northern Spain, was found to depend on 37.7 L of water consumption. Almost 95 % of this water consumption was associated with the supplementary irrigation of the maize grown in Spain used for animal feed. On the other hand, 4.8 L of water were used in the dairy plant to process each 1 kg of raw milk into dairy products. Almost all of this water was used for the pasteurization of the raw milk (Table 1).

Table 1: Inventory of direct and indirect water consumption to process 1 kg of raw milk.

Farm subsystem	water consumption	
	L	%
Livestock direct	1.46	3.44
Hydropower	0.46	1.09
Feed	35.83	84.27
<i>Barley (SP)</i>	<i>4.63</i>	<i>10.90</i>
<i>Maize (SP)</i>	<i>31.17</i>	<i>73.32</i>
<i>Soy (BR)</i>	<i>0.00</i>	<i>0.01</i>
<i>Soy (AR)</i>	<i>0.02</i>	<i>0.05</i>
Dairy industry		
Direct use	3.93	9.24
Hydropower	0.83	1.96
TOTAL	42.52	100

The water scarcity footprint results are presented in Table 2. Regardless of which method was used, feed production was identified as the most important life cycle stage (87 to 92 % of the overall water scarcity footprint). In all cases, the production of maize was most critical, although differences in the contribution of the different feed components was evident when the different methods were compared. In all cases, dairy processing contributed less than 10% of the overall water scarcity footprint (Table 2). These results demonstrate that for strategic LCA studies, where results are used internally within the business and not shared with stakeholders, the choice of water stress index is not so critical as the relative importance of different life cycle stages do not differ so much.

Table 2: Water scarcity footprint results for the processing of 1 kg of raw milk calculated using three methods (WSI, WSIHHEQ and AWARE). See text for details about the methods.

	WSI		WSI _{HHEQ}		AWARE	
	L H ₂ Oe	%	L H ₂ Oe	%	L world eq.	%
Farm subsystem						
Livestock direct	0.03	0.30	0.04	1.47	8.01	0.26
Hydropower	0.46	4.39	0.04	1.53	15.31	0.50
Feed	9.00	86.64	2.51	90.30	2803.54	91.20
<i>Barley (SP)</i>	3.38	32.57	0.33	11.82	335.60	10.92
<i>Maize (SP)</i>	5.61	54.01	2.18	78.37	2466.83	80.24
<i>Soy (BR)</i>	2.4E-04	0.00	1.2E-04	0.00	1.0E-02	0.00
<i>Soy (AR)</i>	7.1E-03	0.07	2.8E-03	0.10	1.1E+00	0.04
Dairy industry						
Direct use	0.08	0.79	0.11	3.95	21.52	0.70
Hydropower	0.82	7.88	0.08	2.75	225.84	7.35
TOTAL	10.39	100	2.78	100	3,074.22	100

However, the absolute results varied enormously, from less than 3 when measuring with WSIHHEQ to more than 3,000 L world e/kg when evaluating with new AWaRe methodology (Table 2).

4. Discussion

In accordance with previous studies, water consumption associated to crop production was the most relevant water consumption stage (Matlock et al., 2012; Sultana et al., 2014). However, being European average for blue water consumption in a dairy farm around 43 – 374 L H₂O/ECM (Sultana et al., 2014) the inventory of the northern Spain dairy farms are located in the low water demanding farms accounting for just 37 L / ECM.

But in this study the focus is pointed into the large deviation of the values regarding the impact accounting. This variation is explained by the differences in the ranges of the different water stress indexes applied: in this study from 0.17 to 0.99 for the revised WSI, from 0.01 to 14 for the WSIHHEQ and from 1.11 to 100 for the AWaRe methodology. Therefore, a major concern arises when selecting the final value to report or communicate. It is therefore considered critical that in the near term, operators of product water footprint programmes or initiatives specify the particular water stress factors to be applied. This will, at least, enable comparability of product water footprints within a programme. If companies making water scarcity footprint claims have liberty to choose any WSI method, then the results going into the public domain will vary so much that there will be no meaningful basis for non-technical people to make any sensible comparisons.

More complex discussion arises when referring to lack of available water due to water pollution. At this point the authors agreed that the pollution of water is already accounted when evaluating freshwater eutrophication or ecotoxicity and thus, in an LCA approach, it could lead into double counting if water pollution is mixed with the water scarcity.

5. Future recommendation for Water Scarcity communication

Overall, around the world there is concern regarding quantity of water required for food production. With the growing population projections this demand will increase significantly upcoming years. Still today, the way people and stakeholders along the food value chain use water in agriculture and processing industries is the most significant contributor to ecosystem degradation and to water scarcity. This situation requires immediate attention from government institutions. Moreover, water professionals need to communicate these concerns better, and policymakers need to be more water-aware.

Over time effort should also be made to achieve consistency between programmes. In this regard, consideration must be given to both reference unit and range. Using this dairy product case study as an example, consumers are likely to have a very different attitude to the product if its water scarcity footprint is reported as 2.8 L H₂Oe per kg compared to more than 3,000 L H₂Oe per kg. Water scarcity footprint results may need to be scaled so that they are comparable to other forms of direct

water use that consumers are familiar with, e.g. flushing a toilet (3 to 6 litre) or taking a shower (7 to 15 L/min), operating a dishwasher (10-12 L per cycle). As such, the reference unit is most important. Expressing water footprints in relation to water consumption at the global average water stress index is helpful in this regard.

Another consideration is the variation in characterization factors between water rich and water scarce locations. If the contrast is too great, it could discourage water efficiency behaviors in water scarce regions because even small amounts of water use lead to very large water scarcity footprints.

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183. E.T.: An operational field water and salt flows model for agricultural LCA illustrated on a Moroccan Mandarin

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ABSTRACT

Objectives: Water inventory and agri-food LCA databases do not fully support the application of LCIA methods assessing the impacts of consumptive and degradative water use (so-called water availability as proposed by Boulay and colleagues), and are not appropriate for LCA-based ecodesign of cropping systems. For herbaceous crops, the FAO Aquacrop model constitutes a relevant and operational model for estimating field water and salt flows, but no dedicated model is available to-date for perennials. The objectives of this work were (i) to develop a simple and operational model for the estimation of field water and salt flows, aiming at discriminating practices for all types of cropping systems including perennials, (ii) to test and discuss its feasibility in a demanding case study. **Methods:** After a review of modelling approaches, we elaborated a tailored model, called the E.T. model, for the inventory of field water and salt flows for use in LCA of cropping systems. The model has a daily water and salts balance, accounting for specific soil, climate and agricultural practices. We explored the model relevance and robustness in a case study of a mandarin crop grown in Morocco, based on farm primary data. We compared the model outputs with the literature and water databases, and calculated water availability impacts. **Results:** The E.T. model is simple and operational on perennial crops, and estimates evaporation, transpiration, deep percolating water and runoff water, in terms of volume and salinity. Its outputs compared well with literature and measurements, and allowed the simulation of scenarios of agricultural practices. A comparison with crop water consumption estimates from databases highlighted a difference of up to 60%. E.T. model outputs are water elementary flows and salinity and can be used for assessing the impacts of consumptive and degradative water use in LCA. **Conclusions:** The E.T. model supports the calculation of water availability impact while discriminating agricultural practices. Its domain of validity and accuracy could be extended based on recommendations from the authors.

Keywords: water inventory, water impacts, crop, perennial, evapotranspiration

1. Introduction

In the context of flourishing eco-labelling programs and environment policy for food products, LCA applied to agricultural systems faces the challenges of being operational, accurate and exhaustive. Indeed, LCA has to include all major environmental impacts including water deprivation. This is particularly challenging for the water use impact assessment in LCA, which is relatively new, with many LCIA methods only recently developed. Among these methods, we can distinguish water scarcity indicators (that only recognise the contribution of water quantity to water deprivation, e.g. Pfister et al. 2009) from water availability indicators (where both water quantity and quality contribute to water deprivation, e.g. Boulay et al. 2011a&b).

Water inventory databases (e.g. WaterStat (WFN 2015)) and agri-food LCA databases (e.g. World Food LCA Database (Nemecek et al. 2015)) contain default water elementary flows for average crop and crop products. These databases support the application of LCIA methods to assess the impacts of water consumption (but not water degradation), for agricultural systems at the background level in the life cycle of food products. They provide only theoretical crop water consumptions that may differ a lot from the water actually withdrawn and consumed, and rely on data and approaches with important limitations, making them inappropriate for agricultural LCA studies where an adaptation of practices is sought (Payen 2015, Payen et al. submitted). Furthermore, in these databases, salt flows in relation to agricultural practices such as irrigation are not accounted for, even though salinisation represents one of the major threats for agricultural systems over the world.

For the LCA-based ecodesign of cropping systems, the inventory of water flows should be based on a model simulating evapotranspiration, deep percolation and runoff accounting for crop specificities, pedo-climatic conditions and agricultural management practices (Payen, 2015). In

particular, the model should account for possible water, saline and nutrient stresses; assess evaporation and transpiration separately and estimate runoff and drainage according to the systems specificities. Yield should not be a model output but primary datum (Payen, 2015).

A review of water and salt flows models (Payen et al. submitted), revealed that the recent FAO model Aquacrop (Steduto et al. 2012) is a relevant tool for estimating water and salt flows for the inventory of agricultural LCA, due to an optimal balance between accuracy, simplicity and robustness. However, while this model is operational for annual crops, no similar model exists to-date for perennial crops. Because the need for an operational tool for field water and salt flows in perennial cropping systems is urgent, even with a scarcity of available data, the objectives of this work were:

- (i) to develop a simple and operational model for the estimation of field water and salt flows, aiming at discriminating practices for all types of cropping systems including perennials,
- (ii) to test its feasibility in a demanding case study; assessing water availability impacts for a mandarin crop grown in Morocco.

2. Methods

2.1. Model description

We did not create a new model from scratch, but elaborated on a tailored model fulfilling our objectives based on old and robust formalisms completed with recent data on transpiration. Based on actual water supply (volume and salinity) and the soil, climate and practice specificities, the model estimates the water consumed through evapotranspiration, and the water released in the environment through deep percolation and runoff. Regarding the water quality, the model accounts for salinity through two aspects: i) its effect on the water balance, and ii) its effect on the environment through emissions. Indeed, salinity of soil water may reduce evapotranspiration (due to osmotic effects), thus affecting all other water flows. The model estimates the salinity of deep percolating water and soil water. Salinity is estimated through the electrical conductivity (in $\text{dS}\cdot\text{m}^{-1}$) of the water, and assuming a conversion factor to $\text{g}\cdot\text{L}^{-1}$ equal to 0.64 (USDA-NRCS 2015).

The model consists of daily water and salt balances (Figure 1). In a schematic way, soil is considered a uniform reservoir in which the soil water and salt content changes as a result of incoming and outgoing water and salt flows. The input data required (on a daily basis) are the depth (in mm) of irrigation water and rainfall, and the electrical conductivity (in $\text{dS}\cdot\text{m}^{-1}$) of the irrigation water. Climate data (e.g. relative humidity, rainfall frequency) and crop physical characteristics data (e.g. crop mean height, fraction of ground covered by vegetation) are also required for the calculation of evaporation and transpiration. Evaporation and transpiration are calculated separately, which is particularly relevant for orchards or vineyards where evapotranspiration is more complex than a uniform herbaceous crop. The transpiration module is specific to the crop, which in our case study is citrus, through the key factor of leaf resistance. A lack of water or an excess of salts in the soil water can reduce crop water consumption. Thus, to estimate actual evapotranspiration ET_a , the model computes stress coefficients that reduce the potential evapotranspiration ET_c according to Allen et al. (1998). The model is called E.T. because the EvapoTranspiration module is crucial. Default data can be used if primary data are not available for crop physical characteristics (Allen et al. 1998), water salinity (UNEP 2009), soil characteristics (Batjes 2006, FAO/IIASA/ISRIC/ISS-CAS/JRC 2012) and climate data (Mitchell and Jones 2005, New et al. 2002).

The E.T. output variables are water flow volumes and salinity. Additional calculations are required to convert the hydrological water flows to water inventory flows usable in LCA. The total actual evapotranspiration (ET_a) is further divided into its green ($\text{ET}_{a\text{ green}}$, effective rainfall) and blue ($\text{ET}_{a\text{ blue}}$, irrigation water) components. The effective rainfall is the part of the rainfall which is actually available to the crop for evapotranspiration; rainfall minus losses through runoff and deep percolation.



Figure 1: The modular structure and main formalisms for the E.T. model. NRCS: Natural Resources Conservation Services.

2.2 Model validity domain

The E.T. model is based on old and robust formalisms having a large domain of validity. For example, surface runoff is estimated based on the Natural Resources Conservation Services (NRCS) Curve Number (Mishra and Singh 2003), accounting for soil type, land use, hydrologic conditions, and antecedent moisture condition. Nevertheless, the model should be modified in situations where soil has a very low saturated conductivity or a shallow aquifer. Regarding the crop, the E.T. model is operational for citrus because the transpiration estimation is based on a recent development specific to citrus crops (Taylor et al. 2015). However, the model is potential valid for all crops if the leaf resistance is adjusted accordingly (refer to Allen and Pereira (2009) and Steduto et al. (2012)). Regarding the technology (agricultural practices) validity domain, this model is tailored for localised irrigation modes. The model has to be adjusted to model surface irrigation.

2.3 Model testing

The model was applied to a perennial mandarin crop grown in Morocco, over seven crop cycles, from planting in October 2007 to April 2015. Most data were primary data collected from a mandarin farm located in central Morocco, owned by “Les Domaines Agricoles”. Table 1 shows a selection of key model input and parameter values and data sources. A few data gaps in climate data were filled following specific rules depending on the climatic parameter. The output variables of the model were compared to field measurements (from bibliography and case study) and water inventory databases (Water Footprint Network; Pfister et al. 2011; Pfister and Bayer 2014). The sensitivity of the E.T. model outputs was assessed against model parameter range testing, based on realistic values rather than arbitrary values for initial conditions of soil water stock ($S(i)$) and soil water salinity ($EC_{\text{soil water}(i)}$), and the average fraction of Total Available Water (TAW) that can be depleted before a water stress occurs, namely “ p ”, (p is crop-specific, and defined in Allen et al. 1998). The E.T. model was used to simulate different scenarios of practice through the use of different input data. We simulated: a larger wetted zone by irrigation, deeper rooting depth for adult trees, bigger (+20%) and smaller (-50%) tree canopy size and plantation density, and a land use type less favourable to runoff through a smaller curve number.

Table 1: Selected key model input variables and parameters, average values for the case study, data sources and assumptions

Input variable/ parameter	Description	Average values for the different mandarin cropping phases ¹			Data source
		Non- productive	Growing yield	Full production	
S	initial soil water stock [mm]	Field capacity			Farm antecedent practices
EC _{soil water}	initial soil water electrical conductivity [dS.m ⁻¹]	1.46			Farm soil analysis
p	fraction of TAW that can be depleted before water stress occurs	0.5			Allen et al. (1998)
%G	ground cover fraction of the tree canopy [%]	20	50	70	Allen et al. (1998) and Taylor (2015)
h	mean height of the vegetation [m]	2	2.5	3	Observation at farm and Allen et al. (1998)
r _{leaf}	mean leaf resistance [s.m ⁻¹]	316* ET _o -61			Citrus orchards : Taylor et al. (2015)
I(t)	Irrigation water [mm]	Total irrigation: 6053			Daily farm records. Data gap: Monthly irrigation disaggregated
EC _{iw}	Electrical conductivity of irrigation water [dS.m ⁻¹]	1.258	1.258 - 1.165	1.187	Irrigation water analysis
ET _o	reference evapotranspiration [mm]	Total ET _o : 15264			Weather station at farm. Data gap: based on average ET _o of known years, or monthly ET _o disaggregated
P(t)	rainfall [mm]	Total rainfall: 1566			Weather station at farm. Data gap: monthly P disaggregated based on rainy days per month
Yield	Total crop yield [ton.ha ⁻¹]	249.5			Farm records : yield from 2008 to 2015

¹Based on the yield, cropping phases are split into a non-productive [0 to 2 years], growing yield [3 to 6], and full production [7 to 25] phase. Full production is when the maximum yield is reached.

2.4 Model application for water use impact assessment

The impacts of water use for mandarin cultivation were assessed with the water availability indicators defined by Boulay et al. (2011a&b). This is the most scientifically-sound method since it accounts for both water quantity and quality effects on water deprivation, and proposes characterisation factors specific to the water compartment for withdrawal and release. Impacts were expressed in m³_{eq} deprived per ton mandarin cultivated. The model outputs served as water elementary flows for the mandarin cultivation stage, and were multiplied by the corresponding characterisation factors. The inventory water flow requirements for applying this method are presented in Figure 2.

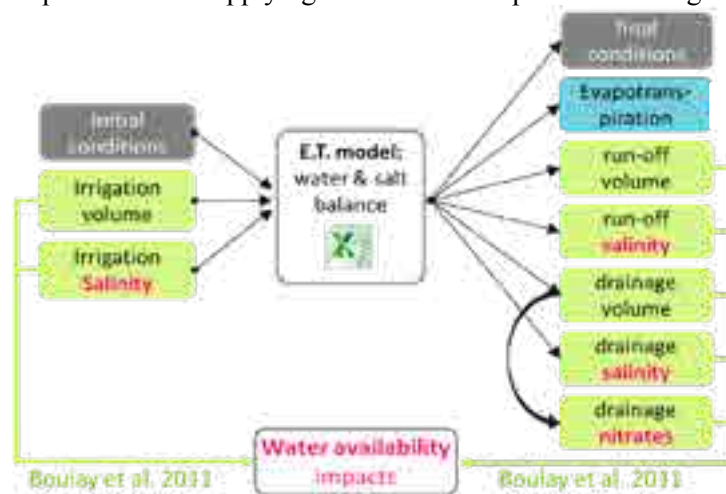


Figure 2: E.T. model integration for water availability impacts calculation: input and output water flows are converted to impacts on the environment following Boulay et al. (2011)

We first compared water inventory flows estimates of the E.T. model, with databases (Pfister et al. 2011; Pfister and Bayer 2014). Then, we compared water deprivation impact results based on different inventory methods.

3. Results and discussion

3.1. Model testing

The sensitivity of the model outputs to initial conditions and the parameter p (fraction of TAW that can be depleted before a water stress occurs) was low, which demonstrated the robustness of the E.T. model. For example, for p values ranging from 0.1 to 0.8, the model outputs were slightly affected: the maximum variation observed was for the actual evapotranspiration originating from rainfall ($ET_{a,green}$) which varied from -5.4% to +2.5%.

The estimations of evaporation and transpiration from the E.T. model compared well with literature and measurements on citrus. For example, Villalobos et al. (2009) measured average evapotranspiration and soil evaporation for a mandarin orchard cultivated in the south of Spain in August and in May at 2.6 and 2.1 $mm \cdot day^{-1}$ while corresponding E.T. model estimates (using previous 3-year data) were 2.8 and 2.0 $mm \cdot day^{-1}$ respectively.

Figure 3 shows that the E.T. model allowed the simulation of scenarios of agricultural practices: land use type less favourable to runoff resulted in a smaller runoff, a bigger tree canopy size and plantation density resulted in greater transpiration, a deeper rooting depth for adult trees resulted in a smaller deep percolation, and a larger wetted zone by irrigation resulted in a greater evaporation.

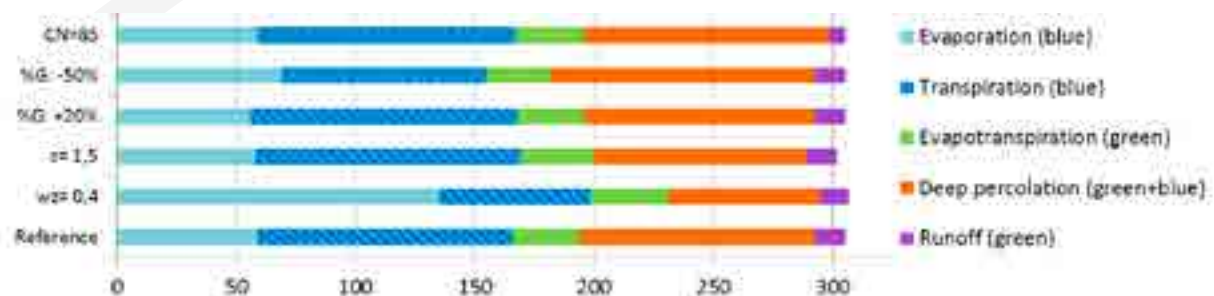


Figure 3: Water flows estimated with the E.T. model (in $m^3 \cdot ton^{-1}$ mandarin cultivated) for different scenarios. Blue and green refer to the origin of water (blue from ground water, green from precipitation). The input variables tested are: CN= curve number (for the runoff calculation), %G= percentage of ground covered by vegetation, z= rooting depth, wz= wetted zone by the irrigation. The reference scenario was: CN=91, z=0.4, wz=0.1.

We used the model salinity outputs for two purposes: (i) estimating the reduction of evapotranspiration due to salinity stress and (ii) estimating the average amount of salts percolating toward the aquifer in deep percolating water. If saline stress was not accounted for, the transpiration was overestimated by 22.3%. This shows the relevance of accounting for saline stress when estimating crop water consumption. The E.T. model estimated an average salt concentration of deep percolating water of 1.88 $g \cdot L^{-1}$.

Comparing the variation of model outputs with the variation of model inputs showed that all water flows were very sensitive to the basal crop coefficient value (describing plant transpiration); this highlighted the importance of an accurate calculation of this “crop transpiration coefficient” specific to crop and practices.

This first testing of the model demonstrated the discriminating power of the model, its low sensitivity to key parameters, and the importance of the crop coefficient value. Nevertheless, beyond

this first testing of the model and its robustness, a proper sensitivity analysis and a Monte Carlo analysis would be warranted in future work, notably to assess the effects of uncertainty interactions and cumulative effects. In particular, we should investigate the models discriminating power regarding the site context (in particular the soil texture which is influencing several parameters), but also we should test the use of default rainfall data from the Climwat database (FAO 2010).

3.2. Model data requirements

Regarding the additional efforts for running the model, in comparison with an LCA study reporting only water volume, the additional effort required for data collection is reasonable since most farmers record irrigation supply volume, at least on a monthly basis. Cross-checking monthly irrigation volumes with irrigation frequency allows data to be disaggregated at a daily time-scale. Other critical data is on water quality, as highlighted by Boulay et al. (2015), but this is the weakest aspect of methods addressing degradative use of water. When water quality analyses are not available, global datasets can be used such as GEMStat (UNEP 2009). Other E.T. model input data (climate, soil...etc) should preferably be primary data, but default values can also be used if necessary (default data sources are provided in Payen, 2015).

3.3 Model limitations and improvement perspectives

It is important to notice that the salt balance is a rather simplified approach since several mechanisms are neglected including precipitation and dissolution of salts, and salt uptake by plants. In addition, salt conversion factors from electrical conductivity to concentration are only an approximate equivalence factor (USDA-NRCS 2015). The effect of salinity on plant nutrition is not accounted for since this would require a nutrient budget and the modelling of complex interactions between salinity, nutrient, water and the crop.

Regarding the stresses of the crop, we only considered water and salinity stresses. Stress coefficients are approximate estimates of salinity and water impacts on evapotranspiration. In particular, evapotranspiration might be underestimated at high salinity levels. Thus, the model could be improved regarding the inclusion of salinity stress and evaporation mechanisms.

The stress coefficient calculation should be revised to account for: (i) the effect of climate and growth development stage, as it is implemented in the Aquacrop model (Raes et al. 2012), (ii) the effect of salinity on the water stress threshold since salinity can lower the minimum soil water content at which a crop will start to be negatively affected (Raes et al. 2012), and (iii) the possible non-linear curve response factor to stresses.

3.4 Comparison of model outputs with water databases

Since water databases do not provide information about water quality or water released, the only E.T. model outputs we could compare with water databases were $ET_{a\ blue}$, and $ET_{a\ green}$. We compared the total blue water consumption ($ET_{a\ blue}$) of the mandarin cultivation stage calculated with the E.T. model and with Pfister et al. (2011) and Pfister and Bayer (2014) databases (providing an estimation of water consumption by crop at a country scale). The lowest $ET_{a\ blue}$ estimate was from the Pfister and Bayer database (2014) at $149\ m^3\cdot ton^{-1}$, and the highest estimate was from the Pfister database (2011) at $237\ m^3\cdot ton^{-1}$, whereas our estimate with the E.T. model (accounting for salinity and water stresses) was $181\ m^3\cdot ton^{-1}$.

3.5 Water impact results

The E.T. model allows the calculation of water impacts using Boulay's approach, which is not possible with water databases. The water availability impacts score was $189\ m^3_{eq}$ per ton mandarin. This water availability indicator addresses both consumptive and degradative use. However, the quality degradation of deep percolating water originating from rainfall was not accounted for in our implementation of the Boulay's method because it relies on the estimation of the evapotranspiration of a "reference state" if the crop was not in place. The rainfall and irrigation water partitioning (so-

called green and blue waters) is arbitrary and fails to properly represent the water cycle. It constitutes an important drawback in the assessment of water use impacts. Additionally, an analysis of the characterisation model from Boulay et al. (2011b) reveals that the ground water specific characterisation factors (0.565) may be underestimated for this area in Morocco. Indeed, the reliability of ground-water specific characterisation factor is questionable since data on groundwater resources do not have a sufficient quality in existing hydrological models (on which the characterisation factors are based) (Boulay et al. 2015).

4. Conclusions

The E.T. model was developed to fill a gap, i.e. the lack of a simple water and salt flow model for perennials, and to meet an objective of determining the inventory of field water and salt flows for the LCA of a cropping system. The E.T. model is a modular and original integration of old and robust concepts for water balance with more recent modules for transpiration estimation. This is a tailored model rather than a new model. Its advantages are its simplicity, transparency and flexibility. It meets the requirements of estimating evaporation, transpiration, deep percolating water and runoff water accounting for possible water and salinity stresses, and based on effective irrigation supply and cropping system characteristics. It also provides information about the quality of water flows via salinity of deep percolating water and the soil water stock. When applied to a perennial crop (mandarin grown in Morocco), the E.T. model outputs compared well with literature and measurements, and allowed the simulation of scenarios of agricultural practices. Its validity domain (in terms of agricultural practices and natural site characteristics (aquifer depth, salinity level)) and accuracy could be extended based on recommendations provided in this work. The E.T. model outputs can serve as water inventory elementary flows to assess the impacts of water use and when LCIA models will be available, to evaluate salinisation impacts (Payen et al. 2016). The use of E.T. for estimating field water and salt flows will ease the application of water use impact assessment methods, including the method addressing both consumptive and degradative water use (e.g. Boulay et al. 2011).

The most scientifically-sound result for water use impacts is the one based on water flows estimated with the E.T. model (accounting for water and salinity stresses), and characterised with Boulay et al. (2011a and b). However, an analysis of the Boulay's characterisation factors showed that developing characterisation factors specific to the water source (surface or groundwater) is very relevant, but their current quality is hampered by the lack of good quality data on groundwater resource state in the global hydrological models they use. Thus, the E.T. model is a relevant tool for the inventory of water flows, but it is important to keep in mind that there are still improvement margins for the impact assessment of water uses.

5. Acknowledgements

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86. LCA of a 4-year switchgrass crop in Eastern Spain

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ABSTRACT

The goal of this study is to evaluate the environmental impact of switchgrass cultivation for electricity production in two locations of *Comunidad Valenciana* (Eastern Spain), under different edaphoclimatic conditions. To analyze the influence of the agricultural practices and the subsequent yield variability throughout the cropping cycle, the environmental performance has been evaluated for 4 years, including both the crop establishment and full production stage. Experimental plots were established under irrigation in two Mediterranean locations, i.e. Moncofar and Orihuela, providing inventory data over the period 2010-2013. The following midpoint indicators have been quantified from a cradle-to-farm gate perspective, per 1 t of switchgrass (on a dry basis): climate change and terrestrial acidification, according to ReCiPe; ecotoxicity and human toxicity following USEtox; and freshwater ecosystem impact (FEI) and green water impact (GWI), for which characterization factors (CFs) have been estimated at the watershed level, based on available methodologies.

The switchgrass production in the southern location (Orihuela) is more input intensive, especially in the first year that includes the crop establishment. Impacts are greater in Orihuela for almost all the productive years and impact categories, despite the fact that the yields achieved in the subsequent years are higher than those in the northern location (Moncofar). This is mainly due to the contribution of the production of fertilizers, machinery use and watering. In terms of impacts from water consumption, FEI is always higher in Orihuela, due to the combination of larger irrigation doses and a higher CF for the corresponding watershed. This translates into a greater GWI in Moncofar, according to the approach considered, which measures the difference in green water consumption between the system studied and a reference system, i.e. citrus. Results show that *ad hoc* decisions on the crop management are critical to the environmental impact. Further efforts are needed to optimize crop management in tune with climate policy demands. LCA can be a valuable tool to evaluate synergies between input intensity and biomass productivity when considering a mass-based functional unit. Given the contribution that low-productivity stages make to the overall impacts from perennial short cycles, it is recommended to gather inventory data for more than one year, including the crop establishment.

Keywords: bioenergy, crop management, perennial crops, switchgrass, water scarcity

1. Introduction

Renewable energies are an essential part of the countries' strategies to meet their climate goals due to their ability to replace fossil fuels in the energy mixes. Specifically, in the European Union (EU), Directive 2015/1513, amending Directive 2009/28/EC –which is commonly known as Renewable Energy Directive (RED)–, requires that 20% of the gross final energy consumed in the Member States comes from renewable sources by 2020. The RED-II, foreseen before the end of 2016, is expected to reinforce this commitment for the period 2020-2030. These renewable sources include biomass, in which the carbon that is ultimately combusted has been previously sequestered from the atmosphere, mainly by photosynthesis. Biomass is the most important renewable energy source in the EU-28, accounting for 64% of primary renewables production in 2014 (Eurostat, 2016). However, this share mostly corresponds to Northern countries, while it is substantially smaller in Spain (38%). Although biomass is largely available, its future market penetration depends on agricultural and technical progress and policy support (IEA, 2012; Scarlat et al., 2015).

Recent public policies (e.g. RED, United States RFS2) clearly promote the use of lignocellulosic and cellulosic biomass such as perennial, herbaceous, non-food energy crops. Specifically, C4 grasses are characterized by a high productivity and resource use efficiency (van der Weijde et al., 2013). Among them, switchgrass (*Panicum virgatum* L.) is adapted to a wide range of soils and climates, and can potentially be grown in cold and warm regions of the EU, even in marginal lands (Alexopoulou et al., 2015; Lewandowski et al., 2003; Smeets et al., 2009). However, the cultivation of switchgrass in the EU for energy purposes, even at the experimental stage, is still very limited (Alexopoulou et al., 2015; Monti et al., 2009; Schmidt et al. 2015). Lowland ecotypes such as the cultivar *Alamo* are taller, coarse, and may lead to high yields under Mediterranean conditions (Maletta et al., 2012). Additionally, the high efficiency of carbon fixation, potential for carbon storage in soil, low nutrient requirements, and long life make switchgrass an interesting option to reduce carbon footprints from bioenergy production (Lerckasemsan and Achenie, 2013; Schmidt et al. 2015). Other potential environmental impacts should be evaluated though, mostly related to the agricultural feedstock

production, such as acidification and eutrophication (Cherubini and Jungmeier, 2010; Sinistore et al., 2015) or water deprivation (Smeets et al., 2009). Life Cycle Assessment (LCA) is considered to be the appropriate method to evaluate the GHG performance of bioenergy compared to that of reference fossil alternatives, according to both the RED and European Commission (EC, 2010).

This study evaluates the environmental impact of switchgrass cultivation for electricity production in two locations of the Mediterranean region of Spain, where there is little experience on this crop. Differences in edaphoclimatic conditions lead to further differences in terms of water availability and agricultural practices. The ultimate goal is to identify hotspots throughout the cropping cycle, in order to identify how differences in management can compromise feedstock sustainability in compliance of the current EU legislation.

2. Methods

Experimental plots (600 m²) for switchgrass cultivation were established under irrigation in two Mediterranean locations, i.e. Moncofar (Castellón) (UTM coordinates X: 742561.8; Y: 4410136.5) and Orihuela (Alicante) (UTM coordinates X: 695011.56; Y: 4198478.3). The most important edaphic and climatic features of the two locations are shown in Table 1. A randomized complete block design with three replications was carried out under the 4-year project of Chueca and Moltó (2012), in order to study the effect of the factor “cultivar” on yields. In accordance with the partial results obtained by Maletta et al. (2012), the *Alamo* cultivar was chosen for the present LCA, since it is better adapted to semiarid conditions. Plantations were established from year 2010 to 2013, including the crop establishment and the year of full production. Agricultural input and output data are summarized in Table 2.

Table 1: Edaphoclimatic characteristics of the two locations for switchgrass production.

	Moncofar	Orihuela
Climatic parameters		
Average annual rainfall (mm)	500-600	300-400
Average annual temperature (°C)	15-17	17-19
Cold period (months)	3-4.5	3-4.5
Warm period (months)	1.5-2	2.5-3
Reference evapotranspiration (ET ₀ , annual) (mm)	1,000-1,060	1,120-1,180
Martonne aridity index	Semi-arid (Mediterranean)	Arid (semi-desert)
Aridity index (precipitation-potential evapotranspiration ratio, P/ET)	0.40	0.25
Soil parameters		
Sand (%)	57	80
Clay (%)	16	20
Silt (%)	27	0
Organic matter content (%)	1.0	0.9
pH	7.8	8.7

2.1. Goal and scope

The Functional Unit (FU) has been defined, from a productive point of view, as 1 ton of switchgrass on a dry basis, in order to capture the influence of yield variability on the environmental performance. To deliver this function, the system implies the following stages:

- *Soil preparation*: a subsoiler attached to a tractor and a rototiller powered by a tractor were used in the two locations prior to the crop establishment.

- *Transport of agricultural inputs*: a distance of 15 km was assumed for the transport from the storehouse to the farm.
- *Seedtime fertilization*: sheep manure was applied in Orihuela with a drawn dry spreader attached to a tractor, while ammonium sulphate was manually applied in Moncofar.
- *Fertilizer application*: ammonium sulphate was applied by hand in Moncofar in 2011 and 2012, while ammonium nitrate was added by fertirrigation in Orihuela during the whole period.
- *Pesticide application*: 2,4-D was applied in Orihuela for weeds control, by means of a mounted herbicide sprayer, while in Moncofar all the herbicides were applied by using a backpack; metaldehyde was also used against snail pests in the first year.
- *Watering*: flood irrigation was applied in Moncofar, based on deficit irrigation strategies, while a drip irrigation system was implemented in Orihuela; a much greater amount of water was applied though in Orihuela due to the sandy soil.
- *Carbon fixation*: perennial crops are presumed to increase the carbon content of the soil, which constitutes an advantage over annual crops.
- *Harvesting and baling*: once a year, a rotary cutter powered by a tractor was used for harvesting; the mowed material was then air dried and baled with a squared baler implement.

Table 2: Average input consumption (per hectare) and yields from the *Alamo* experimental plots.

(*ha ⁻¹ year ⁻¹)	Moncofar				Orihuela			
	2010	2011	2012	2013	2010	2011	2012	2013
Sheep manure (kg)					12,048			
Ammonium sulphate (21% N w/w) (kg)	208.3	416.7	416.7					
Ammonium nitrate (33.5% N w/w) (kg)					360	125	125	125
2,4-D (kg active substance)					1.88	0.94		
Glyphosate (kg active substance)	4.73							
MCPA (kg active substance)	2.54	1.97						
Fluroxipir (kg active substance)	0.76	0.44	0.44					
Metaldehyde (kg active substance)	6.67							
Water (m ³)	5,396	4,521	1,458	3,500	11,420	8,464	17,413	18,444
Diesel (MJ)	8,398	5,262	4,099	4,098	5,849	4,760	4,760	4,760
Yield (tons)	14.31 ± 0.07	15.39 ± 2.43	16.16 ± 0.25	11.31 ± 0.10	8.18 ± 0.06	25.72 ± 0.70	23.15 ± 0.47	19.16 ± 0.25

The processes included within system boundaries, under a farm-to-gate approach, can thus be summarized as follows: a) machinery use, b) fertilizer production, c) herbicide production, d) pesticide production, e) input application (and subsequent field emissions), d) watering, e) carbon fixation. It must be pointed out that technicians based their crop management decisions on their previous experience in other crops in the same area, given the edaphoclimatic conditions.

2.1. Life cycle inventory

The life cycle inventory is based on the following data sources:

- *Agricultural input manufacturing*: Ecoinvent 2.2 database was used except for fluroxipir, for which the method proposed by Audsley et al. (2003) was followed. Emissions from manure storage were allocated to the manure producer and hence excluded (Nemecek and Kägi, 2007).
- *Electricity for watering*: in both locations, the water came from a well (75.0 m and 123.6 m deep, respectively), and the power needed was calculated from the water volume (L) and the pressure (m of water column), also considering the pressure in the watering heads and the pump efficiency. Impacts from electricity consumption were obtained from the Spanish mix in the Ecoinvent 2.2 database.
- *Machinery use*: emissions were calculated based on the power of the tractors employed in each location, the number of hours dedicated to the agricultural works, and the impacts from the associated diesel production in Ecoinvent 2.2.
- *Input application*: EMEP/EEA (2013) was used to calculate NH₃ emissions; N₂O emissions from denitrification were calculated according to IPCC (2006); NO₃⁻ leaching was based on N balances for different Spanish regions (MAAM, 2014); P leaching was estimated by means of Nemecek y Kägi (2007); pesticide emissions were calculated from Berthoud et al. (2011) and Lewis et al. (2015).
- *Carbon fixation*: given the uncertainties in carbon sequestration results and the lack of primary data, an average value of 0.6 t C·ha⁻¹·year⁻¹ was assumed (Cherubini and Jungmeier, 2013).

2.2. Life cycle impact assessment

The following midpoint indicators were quantified: climate change (CC) and terrestrial acidification (TA), according to the ReCiPe 1.08 characterization method; and ecotoxicity (ET) and human toxicity (HT) following USEtox. Additionally, since switchgrass is necessary produced under irrigation in Mediterranean regions, where water scarcity is becoming a matter of priority, the freshwater ecosystem impact (FEI) and green water impact (GWI) were also estimated. For these two last impact categories, characterization factors (CFs) were defined at the watershed level, as detailed below. Orihuela belongs to the Segura basin, while Moncofar corresponds to the Jucar basin.

FEI is a midpoint indicator that captures the impact of blue water consumption, which means consumption of any surface and groundwater through irrigation. It measures the volume of water likely to affect freshwater ecosystems from an ecological point of view. For Orihuela, the CF for groundwater from Hospido et al. (2013) was directly applied: 1.445 m³ ecosystem-equivalent water/m³ irrigation water. For Moncofar, the CF was calculated at 1.035 m³ ecosystem-equivalent water/m³ irrigation water, following the same method. Official data from CHJ (2015) was used for the calculation of the total annual freshwater extraction for human uses and the annually available renewable water supply. A CF greater than 1 means that there is an incomplete coverage of the irrigation demand of the basin by using only groundwater. The impact derived from green water consumption, which is precipitation and soil moisture consumed on-site by vegetation, was addressed according to Núñez et al. (2012). Under the productive scope, GWI was obtained from the difference between the green water consumed by the system studied and the green water consumed by a reference system, which was assumed to be the prevailing crop in the region, i.e. citrus. This difference is finally multiplied by the CF for blue water consumption in each river basin. The amount of green water consumed by both switchgrass and citrus is given by the effective precipitation and the crop evapotranspiration (IVIA, 2015), shown in Table 3.

Table 3: Green water consumed per FU (m³/t).

Year	2010	2011	2012	2013
Moncofar	171.96	177.85	180.48	307.42
Orihuela	178.72	51.11	89.06	102.97

3. Results

Yearly impact results for the two locations are shown in Fig. 1; differences are due to variability in weather, inputs and yields, but also in the physiological stage of the plant. The first year entails the greatest impacts due to the more intense agricultural practices, especially in Orihuela. This is mainly due to the heavy irrigation as combined with the lowest yield, which was 43% lower than that in Moncofar. The yields achieved in the subsequent years are, however, higher than those in the northern location (Moncofar), leading to closer impact values for 2011-2013. Despite this yield effect, impacts are greater in Orihuela for almost all the productive years and impact categories, with the exception of ET in 2011 and GWI. The *production of fertilizers* (i.e. ammonium sulphate) makes the difference in the first case, while, as for GWI, the difference between the two locations lies in the climatic conditions. The first year is the only one for which the GWI is greater in Orihuela, and this is because, under the productive scope, switchgrass withdraws much more water from the soil than citrus does (per ton of product), due to the aridity conditions. In 2010, GWI is negative for the two locations, which indicates that citrus is more demanding in terms of green water than switchgrass, mainly because the increased yields of the latter, especially in Orihuela. In the two last years, higher average temperatures and less rainfall lead to lower green water availability in Orihuela (note the high irrigation doses in Table 2), hence the difference is again positive and greater in Moncofar. In terms of FEI, the impact is between 1.6 and 11.6 times higher in Orihuela than that in Moncofar, as a result of the combination of high water consumption and the high characterization factor of the Segura basin, since the water withdrawals in this watershed exceed availability than in the Jucar one to a greater extent.

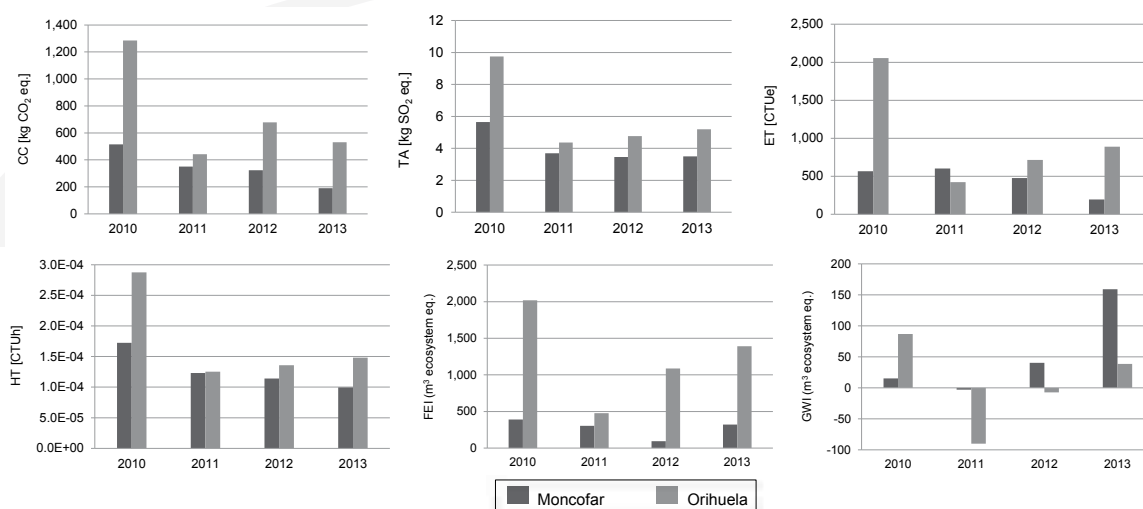
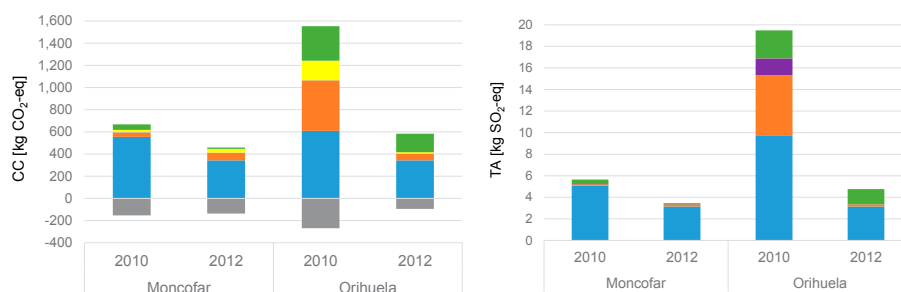


Figure 1: Impact results for the crop's entire production cycle in the two locations. CC: climate change; TA: terrestrial acidification; ET: ecotoxicity; HT: human toxicity; FEI: freshwater ecosystem impact; GWI: green water impact.



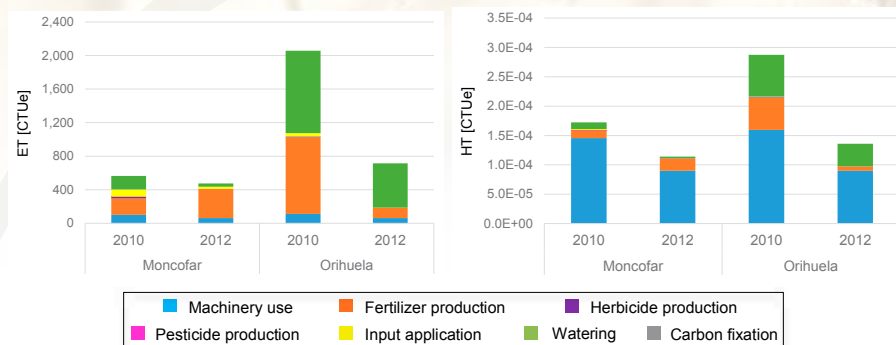


Figure 2: Contribution assessment for the years 2010 and 2012 in the two locations.

In order to analyze the differences between impacts in the two locations further, it is interesting to describe the relative contribution of each sub-process to the overall impact. This has been done for both the first year and a full production year (i.e. 2012), for which the greatest differences can be observed in terms of the input intensity-yield relationship. The results from this contribution assessment are shown in Fig. 2; FEI and GWI are not included since both are entirely caused by watering.

The *machinery use* is responsible for the largest share of CC, TA and HT in the two locations, due to the associated diesel consumption. In absolute terms, the impact of this sub-process is greater in 2010 than in 2012 for a given location, as a consequence of the soil preparation and seedtime fertilization. For the same reason it is greater in Orihuela than in Moncofar in 2010, while machinery use generates the same impacts in 2010, since they arise only from harvesting and baling and tractors of similar characteristics were used in the two locations. The contribution of *watering* is significant in Orihuela, especially in 2010; the same is observed for *fertilizer production*, due to the consumption of ammonium nitrate. These two sub-processes represent the largest share of ET. The field emissions from *input application* are only relevant for the impact categories CC and ET; in the first case, because of the N_2O from fertilizers and, in the second case, because of the emissions of pesticides, mainly to air. The contribution of *input application* to TA, which is negligible, is essentially due to the NH_3 emission from fertilizers. Finally, *carbon fixation* plays an important role in reducing CC per ton of switchgrass, especially in the first years, since yields are lower.

4. Discussion

Results highlight the importance of not focusing only on a full production year when performing an LCA of perennial crops, since impacts vary significantly depending on the agricultural practices and crop cycle stage. According to Smeets et al. (2009), once established, switchgrass takes 3 years to come to its full production potential, reaching an average annual yield of 16 t dry biomass/ha. While the maximum yield in Moncofar is in line with this value, the highest yield obtained in Orihuela is around 26 t dry biomass/ha. This is the result of different management techniques in the two locations, besides differences in soil characteristics, which are critical for switchgrass yields (Alexopoulou et al., 2015). The contribution assessment shows that agricultural practices (i.e. machinery use and watering) are indeed decisive for the impacts of switchgrass cultivation in the Mediterranean region of Spain. In this case, fertilization doses were established based on the type of soil and the very low organic matter content in the two locations, but following conservative criteria (Berg, 1995). Sheep manure was chosen in Orihuela as a seedtime fertilizer due to the higher percentage of sand; this ensures a slow release throughout the cropping cycle, besides quick-release nitrogen is easily washed out of the root zone in this kind of soils. No fertilizers were used in Moncofar in 2013 because technicians wanted to analyze the residual effect of N on the last year yield, considering that part of the ammonium sulphate used is in slow-release form. Nevertheless, it must be noted that these agricultural practices correspond to two case studies and cannot be taken as optimal crop management schemes. Further research must be carried out on the links between input application, biomass productivity and environmental impacts in *Comunidad Valenciana*.

When comparing the LCA results with those from other studies, we see that switchgrass may generate greater impacts under Mediterranean conditions. While Smeets et al. (2009) found that 6.4-7.5 kg CO₂-eq./GJ were produced during cultivation, the values for Moncofar vary between 10.2 and 27.7 kg CO₂-eq./GJ, while in Orihuela they are in the range of 23.8-69.1 kg CO₂-eq./GJ. As for TA, the values reported by Sinistore et al. (2015) are around 0.1 kg SO₂-eq./GJ, while the impact results are in the range 0.19-0.30 kg SO₂-eq./GJ for Moncofar and 0.23-0.52 kg SO₂-eq./GJ for Orihuela. These differences arise from the fact that both studies are based on North American conditions, where there is longer experience on switchgrass cultivation, which translates in a less intensive use of inputs, especially irrigation water, which is not needed.

In accordance with the goal of the study, the GHG savings generated if the biomass produced in the two locations was employed for electricity production have been quantified by following EC (2010), shown in Table 4. Life cycle emissions from Sastre et al. (2015) have been included and electricity from natural gas has been taken as the fossil reference. The net calorific value at constant pressure of the switchgrass is 18.6 MJ/kg (Chueca and Moltó, 2012). In Moncofar, three years after the crop establishment, the biomass would deliver substantial GHG savings. Hence, the switchgrass produced in Moncofar is more likely to meet the 60% of the RED –after 2017– if further efforts are made to improve crop management. However, this implies that the biomass produced in the unproductive phases should be diverted to other uses, which can be a problem for the producer. Better estimates of carbon fixation capturing climate variability in the two locations would increase the reliability of the GHG saving outcomes.

Table 4: GHG savings (%) generated by electricity from switchgrass from the two locations, relative to the fossil fuel reference (143 Mg CO₂-eq/TJ), under Spanish conditions (Sastre et al., 2015).

	2010	2011	2012	2013
Moncofar	32.4	53.4	56.8	73.7
Orihuela	-65.8	41.7	11.4	30.3

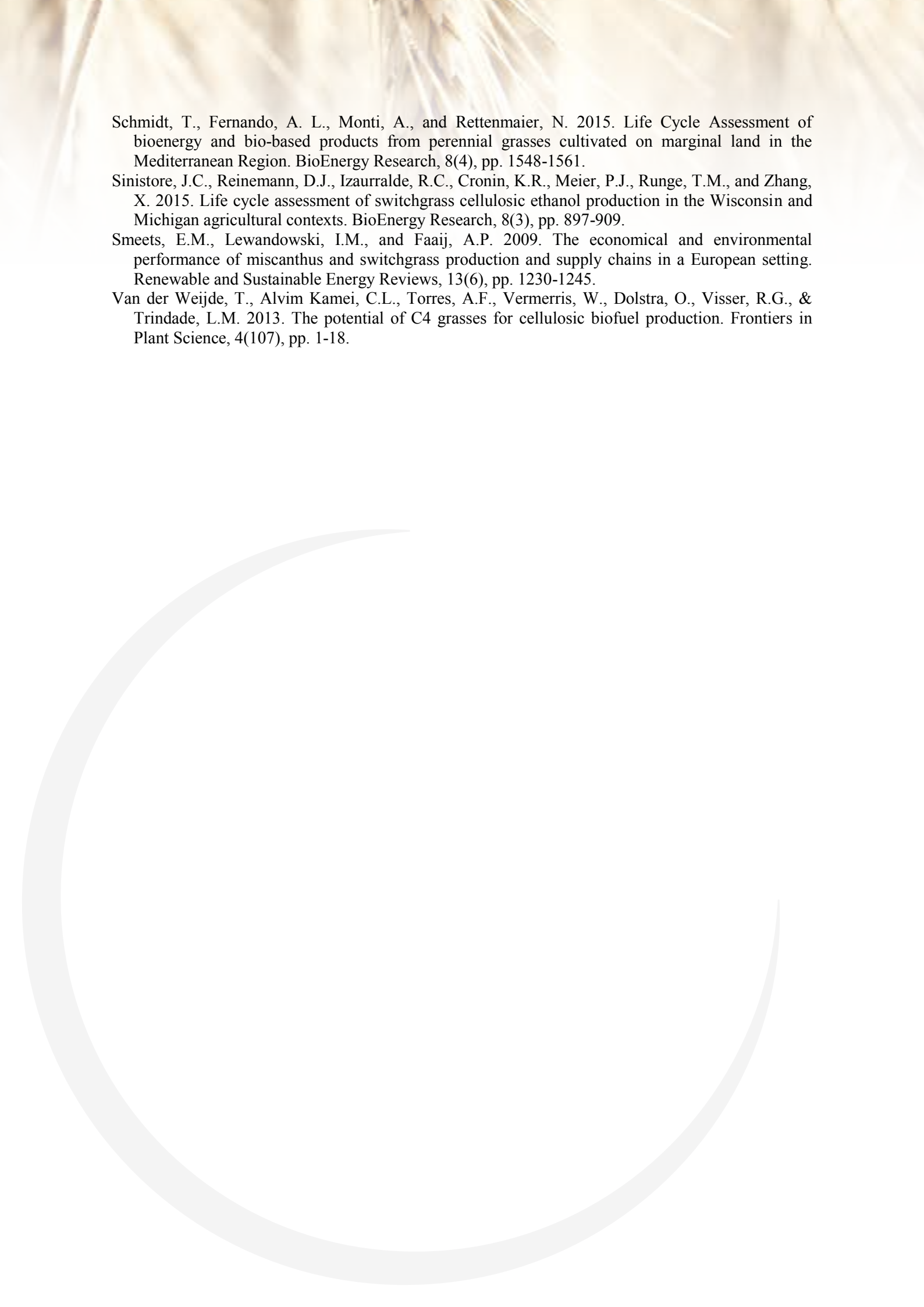
5. Conclusions

An LCA has been performed on switchgrass production in two Mediterranean locations of Spain. Figures point to more intensive agricultural practices (mainly fertilization and irrigation) relative to those in North America, where switchgrass is native. Results show that *ad hoc* decisions on the agricultural techniques are critical to the environmental impact. Further efforts are needed to optimize crop management in tune with climate policy demands. LCA can be a valuable tool to evaluate synergies between input intensity and biomass productivity when considering a mass-based functional unit. Given the contribution that low-productivity stages make to the overall impacts from perennial short cycles, it is recommended to gather inventory data for more than one year, including the crop establishment. The impacts of this stage should otherwise be allocated to an average full production year in order not to obtain misleading results. Other options such as breeding and general (bio)technological improvements should be explored in order to meet sustainability requirements, allowing for cultivars specifically adapted to Mediterranean conditions to be obtained.

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224. An Aquaponics Life Cycle Assessment: Evaluating an Innovative Method for Growing High Quality Produce and Protein

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ABSTRACT

The vast majority of our food currently comes from the globalized supply chain, which is controlled by a few multinational corporations. The industrialized global system depends on artificial fertilizer, pesticides, and fossil-fuel which are devastating to the environment, human health, and local economies. While the globalized food system produces huge quantities of food, it threatens food security and sovereignty, and limits local food production. One of the proposed solutions to inhibit this negative feedback loop is to actively promote and invest in local food systems. It is imperative to analyze alternative food systems as a long-term solution because of the expected decrease in food production from climate change, the increase demand for food due to population growth, and the extreme amount of nutrient pollution from current agriculture methods. In this study we performed a life cycle assessment on the innovative alternative food production systems known as aquaponics. Aquaponics is a promising system design to produce protein and vegetables using minimal resources and waste production. This research evaluated if aquaponics can offer decreased global warming potential, energy use, and water dependency, in comparison to traditional agriculture, hydroponics and aquaculture. These results indicated that aquaponics had a higher yield than traditional agriculture by 25.91 kg/m², and a lower yield than hydroponics by 18.98 kg/m². Aquaponics had a lower water dependency than traditional agriculture by 0.04 m³/kg, but a higher water dependency than hydroponics by 0.19 m³/kg. The energy use for aquaponics was much lower than hydroponics at 79.42 mJ/kg, but higher than traditional agriculture by 9.48 mJ/kg. Aquaponics has a high GWP compared to other aquaculture systems, due to the reliance on natural gas consumption to heat the greenhouse air and water. In this study, aquaponics had the lowest energy use as well as the lowest water dependency compared to all other aquaculture systems recorded. Commercial aquaponic technology is in the early stages of development worldwide and understanding the system costs and benefits may lead to better system management and long-term decisions on the viability of aquaponics as a potential global solution to the food crisis.

Keywords: agriculture; sustainability; water; global warming potential; energy use

1. Introduction

This research will assess the operational production and sustainability potential of a commercial aquaponics system known as “Flourish Farms” located in Denver, Colorado in the United States. The global food production system is projected to decline in crop output due to climate change (Nelson 2009), and population growth will continue to exceed the carrying capacity of the planet (Barrett & Odum 2000), which will lead to a greater percentage of the world’s population receiving inadequate nutrition on a daily basis. Current agricultural methods are some of the primary contributors to climate change and environmental degradation, and if they are further expanded to meet the increasing demand, environmental collapse is expected (Edenhoger et al 2014). Alternative food production systems, such as hydroponics, aquaculture, urban gardening, and local food production offer alternatives to current food production, and towards healthier and more sustainable crop output while revitalizing the environment. Aquaponic technology is a system designed to produce protein and vegetables using minimal resources and waste production. This technology is still used as a niche farming method with only 257 systems out of the 809 United States systems surveyed in 2014 operating on the commercial scale, with all others classified as backyard or hobby systems (Love et al). However, aquaponics is a rapidly growing field as over 600 systems have been built in the United States from 2010 to 2013 (Love et al 2014). Completing a Life Cycle Assessment (LCA) on one of the well-founded commercial systems in Denver will elucidate the water dependency, resource use and global warming potential of this aquaponics system. Only one other LCA on aquaponics has been completed using theoretical production numbers (Bainbridge 2012), so this study will be the first based with data generated fully from an aquaponics system.

Aquaponic farming is a promising technology for local, sustainable food production. **Aquaponics** combines **aquaculture** and **hydroponics** in a recirculating engineered ecosystem that utilizes the effluent from aquatic animals rich in ammonium by circulating it to nitrifying rhizobacteria to fertilize hydroponic vegetables. *Nitrosomona* species oxidize the toxic ammonia (NH₃) into nitrite, and then *Nitrospira* bacteria convert nitrite (NO₂⁻) into nitrate (NO₃⁻), which is less harmful to the fish and a nutrient for the plants. The water, now stripped of most ammonia and nitrates after flowing through

the bacteria matrix and root system, circulates back to the aquaculture subsystem (McMurty et al 1997) This system design can annually produce up to 41.5 kg/m³ of tilapia and 59.6kg/m² of tomatoes in a 1.2m wide, 0.33m deep and 0.86m long tank with 4 plant plots (McMurtry et al 1997).

Objective

In order to further examine and assess aquaponics as a method to grow high quality food, we performed an LCA on the commercial aquaponics system in Denver, Colorado, which compared the global warming potential (GWP), energy use (EU) and water dependency (WD) to literature recordings of resource use in conventional agriculture, aquaculture, and hydroponics.

2. Methods

Study Site

The LCA took place at Flourish Farms, run by Colorado Aquaponics, within the GrowHaus. The GrowHaus is in a historic 1,858 square meter greenhouse which functions as a non-profit indoor farm, marketplace and educational center. They aim to create a community-driven, neighborhood-based food system by serving as a hub for food distribution, production, education and job creation (www.growhaus.com). Food is produced year-round at the GrowHaus with three separate sustainable and innovative growing farms: hydroponics, permaculture and aquaponics. The scope of this study will concentrate on the aquaponic farm ‘Flourish Farms’ which occupies 297 square-meters within the GrowHaus (Fig. 1).

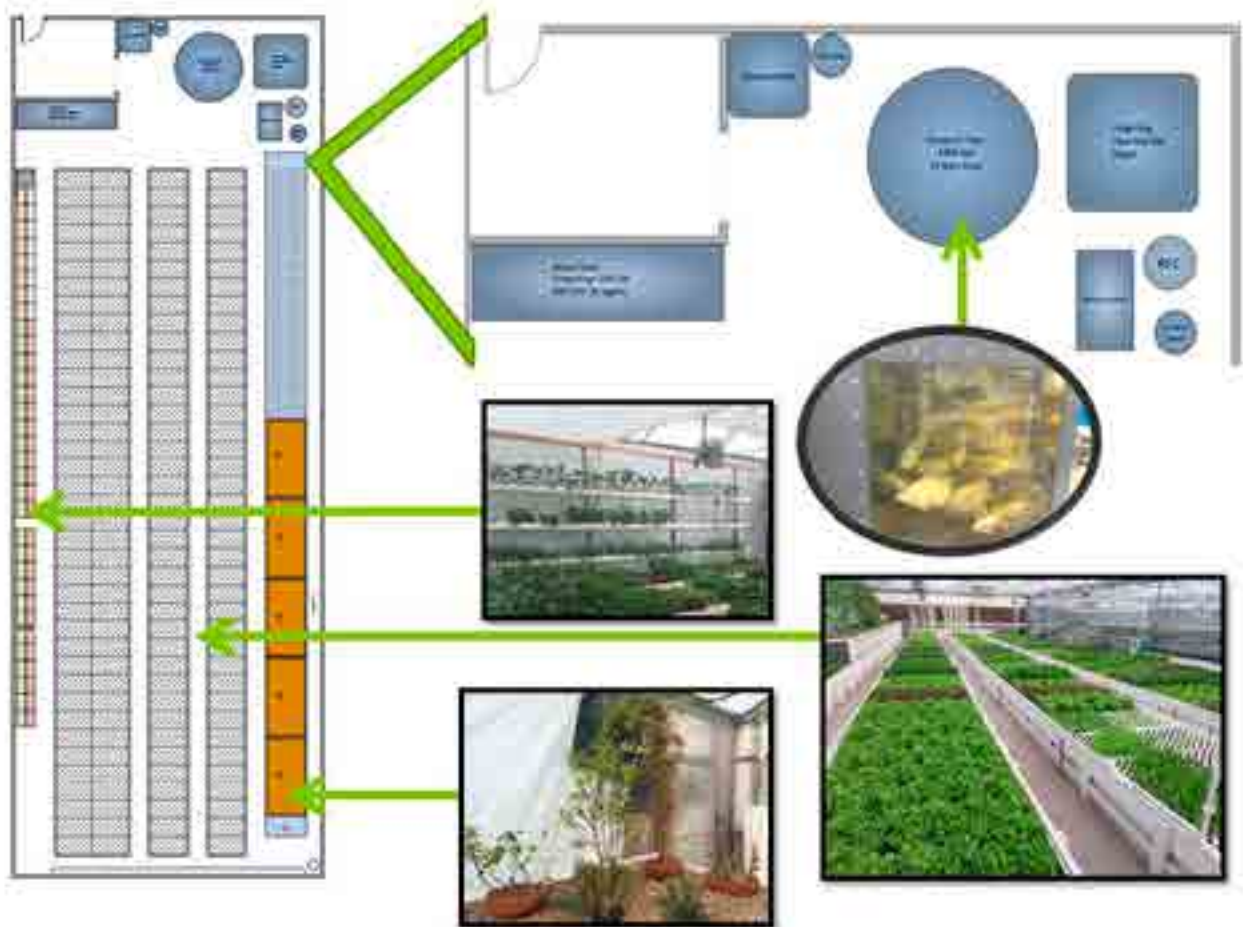


Figure 1: A layout of the aquaponic facility “Flourish Farms”, located in Denver, CO. The image depicts the integration of DWC, NFT and media beds for growing produce.

Flourish Farms was founded in 2009 by owners and CEOs Tawnya and JD Sawyer. The farm serves not only as a commercial production center, but also as a model system that has been mimicked in schools, community buildings, correctional facilities, and homes. As part of Colorado Aquaponics' mission, they provide aquaponic training, curriculum, consultation and support programs that can be delivered to individuals, schools, institutions and communities looking to take charge of their own sustainable farming and food security (coloradoaquaponics.org).

Flourish farms contains three types of aquaponic systems, deep water culture (DWC), nutrient film technique (NFT) and media beds, as the owners showcase the various construction designs for aquaponics systems. The farm used a tilapia and koi carp combination for many years, due to these species resilience and rapid growth even under high stocking densities. However, starting in 2014 they gradually switched to hybrid striped bass (HSB), recognizing a greater value and preference for this fish in their customer core (Tawnya Sawyer, Personal communication 2015). They have also successfully raised catfish and bluegill. Since Flourish Farms moved into the GrowHaus in 2012 they have grown hundreds of different varieties of vegetables and have sold over 13,607 kg of food within an eight kilometer radius.

Goal and Scope

The LCA for Flourish Farms is intended to exhibit the environmental impact of aquaponics systems, as compared to traditional agriculture, hydroponics and aquaculture. The goal is to provide quantitative data to justify usage of one system versus another, which will be an increasingly important question. The product outputs in aquaponics are the vegetables produced by weight and fish harvested by estimated weights. The system boundary is a single issue LCA approach, with an Order II analysis, focusing on the carbon, energy and water within the farm for the entire 2014 year. The scope includes the energy carriers, natural gas consumption, water use, and the input of fish feed into the system (Goedkoop et al 2013). For this study the capital infrastructure and minor inputs, such as nutrient adjustments and integrated pest managements, were listed and measured, but excluded from the LCA analysis.

Methods

Flourish Farms has kept detailed records which contain information on vegetable and fish species output, fish food input into the system, electrical requirements, utility bills and necessary equipment for operational activity. Each monthly utility meter reading was recorded for natural gas consumption and water use. The electrical requirements were calculated based off of kWh operational data listed on each piece of equipment. Each piece of electrical equipment was evaluated for the average of hours per day it would run, and seasonal variation. These values were summed to produce the total kWh the farm uses in one year. This value was converted into milliJoules for this study, and reported as the energy use (EU). In order to calculate the global warming potential (GWP), operational information from the City and County of Denver was acquired to transfer all metrics into relevant kg CO₂e per kg of output. The city reported that the current Xcel Energy emissions factor for Denver, Colorado was 0.79 kg CO₂e/kWh (Ramaswami et al 2007). This factor was applied to all collected kWh 2014 data from the farm, and transferred into GWP by converting kWh into metric tons of CO₂ production. The natural gas consumption for the farm was recorded in therms, which was converted into metric tons of CO₂/therm using the following equation, which was derived from data from the EIA (2016), EPA (2016) and IPCC (2006):

$$\frac{0.1 \text{ mmbtu}}{1 \text{ therm}} \times \frac{14.46 \text{ kg C}}{\text{mmbtu}} \times \frac{44 \text{ kg CO}_2}{12 \text{ kg C}} \times \frac{1 \text{ metric ton}}{1,000 \text{ kg}} = \frac{0.005302 \text{ metric tons CO}_2}{\text{therm}}$$

This value was then converted into kg CO₂e and added to the kg CO₂e from the electrical output to form the total GWP value. These values were then divided by the total production of vegetables and fish to produce the kg CO₂e/kg value. The water dependency (WD) was collected using meter pulls from Denver Water for the farm within the GrowHaus through 2014. These data include *all* water used by the farm, not just what would be inserted into the system to replace daily evaporation and transpiration. Additional LCA data was obtained from Skretting's Annual Sustainability Report

(2014), a cradle to gate LCA analysis, which was incorporated into the study to account for the fish food input.

Since the farm produces tilapia, HSB and various vegetables from the same sources, the input data were allocated to two categories of production using economic profits, as practiced by other aquaculture LCA studies (Ayer et al 2008). This resulted in 16.3% of the resources contributing to the aquaculture production, and 83.7% of the resources to the vegetable growth.

3. Results

Inventory Analysis

Flourish Farms contains many components in order to run at a commercial scale. The main infrastructures are four raft beds, a wall of NFT pumps, a wooden fish tank for younger fish, and a main tank for mature fish. There are also two filtration systems to remove the solid fish effluent from the system in order to prevent waste accumulation and root damage (Table 1). The building uses equipment to control temperature, humidity, light, and water flow. These include horizontal airflow fans, modine heaters, vent fans, a wet wall pump, circulation pump, HID metal halide lights, intermediate bulk container power pumps, a main valuflow 6100 water pump, media bed water pumps, an NFT pump, nursery pumps, and an S31 regenerative air blower. The system also uses fish tank boilers to heat the water.

Table 1: Necessary infrastructure in Flourish Farm's aquaponic system

Component	Volume (m ³)	Dimensions (m)
Raft Bed 1(Media and DWC)	7.86	1.22 W x 23.16 L
Raft Bed 2 DWC	7.86	1.22 W x 23.16 L
Raft Bed 3DWC	5.76	1.22 W x 7.92 L
Raft Bed 4 DWC	11.52	2.44 W x 23.16 L
NFT Pumps	-	30.48 L
Wood Fish Tank	2.03	0.92 W x 3.35 L x 0.61 Deep
Main Fish Tank	3.18	2.29 Diameter x 1.02 Deep
Blue Tank (Cone Bottom)	1.76	1.57 W x 1.57 L x 0.71 Deep
Brush Filtration Tank	0.68	0.66 H x 1.35 L x 0.94 W
Clarifier Filter (Cone Bottom)	0.45	0.71 Diameter x 1.47 H

Results

Flourish Farms used a total of 26,846.90 kg of CO₂ equivalency in order to produce 2,699.54 kg of lettuce and 252.09 kg of fish in 2014. The total EU for the system was 33,670.21 mJ, and the WD was 420 m³ for all operations (Table 2). Flourish Farms has zero waste, as all solids removed from the clarifier filter were remixed into a fertilizer solution for use in soil based gardens, lawns, compost and foliar sprays. All roots from the vegetables are either sold with the product, or trimmed and used in composting bins.

Table 2: The total global warming potential (kg CO₂e), energy use (mJ) and water dependency (WD) for Flourish Farm lettuce and tilapia and hybrid striped bass per kilogram in 2014.

	Mass Produced (kg)	Units Produced	Economic Allocation (%)	GWP (kg CO ₂ e/kg/y)	EU (mJ/kg/y)	WD (m ³ /kg/y)
Tilapia/HSB	252.09	1,685	16.30	17.36	21.77	0.27
Vegetables	2,699.55	30,553	83.70	8.32	10.44	0.13
Feed	-	-	-	9.91	0.17	0.10
Total	2,951.64	32,238	100	35.59	32.38	0.50

Flourish Farm's main operational components consists of fish feed, integrated pest management (IPM) and nutrient additions. The farm used a total of 174.72 kg of fish feed, 0.50 m³ of various nutrient additions, and 0.13 m³ of various IPM in 2014.

The results from this study were then compared to data from the literature to evaluate environmental costs to hydroponic systems, aquaculture systems and traditional agriculture (Table 3, 4). Aquaponics had a higher yield than traditional agriculture by 18.12 kg/m²/y, and a lower yield than hydroponics by 18.98 kg/m²/y. The data indicated that aquaponics had a lower WD than traditional agriculture by 0.04 m³/kg/y, but a higher WD than hydroponics by 0.19 m³/kg/y. EU was the highest in hydroponic systems, with aquaponics lower by 79.42 mJ/kg/y. Aquaponics had a higher EU than traditional agriculture by 9.48 mJ/kg/y.

Table 3: Comparative of annual land use, water dependency and energy use in aquaponics, hydroponics and traditional agriculture for lettuce production. The aquaponic data for this comparison used the 83.7% allocation to calculate the combined vegetable and fish feed WD and EU.

Agricultural Type	Yield (kg/m ² /y)	WD (m ³ /kg/y)	EU (mJ/kg/y)	Reference
Aquaponics	22.02	0.21	10.58	This study
Hydroponics	41 ± 6.1	0.02 ± 0.01	90.00 ± 11.00	Barbosa et al 2015
Traditional Agriculture	3.9 ± 0.21	0.25 ± 0.03	1.10 ± 0.08	Barbosa et al 2015

The fish production of aquaponics was compared to various aquaculture LCA studies. The results indicated that aquaponics had a high GWP compared to other aquaculture systems. While the Bainbridge aquaponic study showed a mid-range EU, this study's aquaponic system had the lowest EU of all systems. Both aquaponic systems had the lowest WD compared to all other aquaculture types.

Table 4: Comparison of Global Warming Potential, Energy Use, and Water Dependency of various aquaculture systems with values in terms of one kg produced. The aquaponic data for this comparison used the 16.3% allocation to calculate the combined fish and fish feed GWP, WD and EU.

Fish Type	System Type	GWP (kg CO ₂ eq/kg/y)	EU (mJ/kg/y)	WD (m ³ /kg/y)	Reference
Tilapia & HSB	Aquaponics	18.97	21.79	0.29	This Study
Tilapia	Aquaponics	7.18	121.25	0.01	Bainbridge 2012
Turbot	Recirculation	6.02	290.99	4.81	Aubin et al 2006
Rainbow trout	Flow through	2.02	34.87	98.80	Roque d'Orbcastel et al 2009
Rainbow trout	Flow through	2.75	78.23	52.60	Aubin et al 2009
Rainbow trout	Recirculation	2.04	63.20	6.63	Roque d'Orbcastel et al 2009
Seabass	Net pen	3.60	54.66	48,720.00	Aubin et al 2009
Arctic Char Atlantic	Recirculation	28.20	353.00	-	Ayer and Tyedmers, 2009
Salmon	Net Pen	2.07	26.90	-	Ayer and Tyedmers, 2009

4. Discussion

The field of aquaponics has been rapidly growing over the past decades but there are very few rigorous, peer-reviewed systems research published on the topic. Because of this, assessment of these systems is desperately needed in order to provide stakeholders valuable information on the benefits and costs of aquaponics as well as the potential these systems have for providing sustainable food production and economic development. This LCA demonstrated that while aquaponics has beneficial reductions for some environmental impacts associated with food production, it has a higher impact in some categories than other agricultural systems. Aquaponics showed a great potential for increasing

yield per land area, while decreasing water use compared to traditional agriculture, however, the energy use would increase significantly. Aquaponics was also outperformed by hydroponics in regards to yields and water use, however this aquaponic system used far less energy than the comparative hydroponic studies. In regards to fish production, aquaponics contributed more to GWP than all other types of aquaculture except for a recirculation system. However, the EU and WD for this aquaponic study was the lowest of all aquaculture systems, which has the potential for huge natural resource conservation.

This study identified areas where efficiencies could be built into aquaponics in order to have a more sustainable system. The GWP for aquaponics was very high, and could be reduced by farms considering alternative energy solutions, such as purchasing wind energy from their source. The farm currently has plans to install solar panels which will reduce both the GWP and the EU for the system. Currently 65% of Flourish Farms GWP is from the natural gas consumption of the modine heaters and fish boiler units. Part of this high natural gas consumption comes from the temperate continental climate, which generates hot summers and very cold winters, requiring high temperature mediation. Additionally, the fish feed constituted 27% of the total GWP. This high percentage may indicate that other feeding mediums should be explored, such as duckweed and compost which have little to no GWP. Another aspect to consider is the building where Flourish Farms is located is in a repurposed historic greenhouse, which lacks modern infrastructure to more efficiently retain heat. Solar thermal heating could be applied to the building to reduce the GWP, as well as a climate battery, which could store hot air underground to use during the cold weather. One advantage aquaponics has in comparison to traditional agriculture is the local customer base. Flourish farms sells and delivers all of its products within an 8 kilometer radius. In future studies the transport for this aquaponic system should be included in the LCA boundary to examine the GWP benefits of reducing transportation distance in agriculture.

While the WD for aquaponics was lower than traditional agriculture and aquaculture, it was still higher than the predictive 10% of water usage of traditional agriculture that many studies support (Somerville et al 2014; Lennard & Leonard, 2006; Bainbridge 2012). This LCA indicated that aquaponics only uses 16% less water than traditional agriculture. Flourish Farms going forward should carefully track where water is being applied in the system, and look for any possible reductions. The Skretting fish food compromised 20% of the total WD, which as mentioned above, may add further reason to explore other feeds. Another possible reduction is Denver approved rainwater collection in 2016, which could be another water source the farm could utilize instead of tap water. The Bainbridge aquaponic LCA predicted that a 0 WD could be achieved in their system by relying on rainwater collection alone (2012). The farm also experienced several operational emergencies during 2014, which could have caused a need for the system to be flushed. Additional years of data and notes of future notes of high water usage may prove that 2014 was an outlier in WD for Flourish Farms.

The EU for aquaponics far outperformed hydroponics, traditional agriculture and aquaculture. This indicates that the system effectively utilizes gravity for water flow wherever possible, and the pumps and aerators are energy efficient.

Some points of consideration for this study that may contribute to the results having more variance than expected is Flourish Farms, up until recently, did not weigh their fish. The method for sale included estimating fish length at approximately 12.7 cm long or “plate size” and selling the fish for an even five USD. Typical aquaculture studies meticulously weigh the protein produced and sell the fish by weight which gives very accurate production numbers, instead of the more general estimates used in this study. The farm also experienced a dramatic die-off during the 2014 year in which 491 fish died due to a loss of electricity. In order to account for this die off, this study added these fish weights into the protein produced, even though this protein was not sold.

Additionally, the allocation methods for this study could be improved since the economic production required some estimation in regards to fish sales and vegetable sales. A method involving resource requirements or mass for each subcategory of production may generate better allocation percentages and will be considered for future research.

As this LCA is the first aquaponic study to use recorded data for a full year, instead of theoretical data, these results can be used by farmers in real situations who are considering aquaponic farming to increase their yield, and decrease their resource use and environmental impact. Ultimately, better

systems information will quantitatively address hypotheses about the relative efficiencies of aquaponic vs. other alternative farming techniques.

5. Conclusions

This study has shown that aquaponics demonstrates certain environmental efficiencies as compared to other agriculture systems which if applied on a larger scaled, could have positive environmental impacts on the food system. This production system also shows promise in international development to increase access to affordable protein when there are limited options available. This research demonstrated that there may be ways to produce high quality protein and produce, while making a profit in a way that is less environmentally wasteful and costly than traditional agriculture, hydroponics and aquaculture. Our current food system is broken and is one of the largest sources of global pollution and further investigation and alterations of alternative food systems could be a step in changing this situation for the better.

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2. Dairy Production

112. Allocation choices strongly affect technology evaluation in dairy processing

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ABSTRACT

This paper illustrates how different allocation approaches affect the assessment of energy and water saving technology options in the dairy processing sector. The choice between allocation on facility level or process level was evaluated, as well as the choice between dry matter and economic allocation in a case study of a mozzarella facility based on primary data. It was found that the carbon footprint allocated to the main product, mozzarella, is very sensitive to these methodological choices, because the dry matter in mozzarella is valued relatively highly, and would receive impact from the energy intensive whey processing under the facility level allocation approach. Economic allocation on the process level gives results that are most unambiguous and straightforward to interpret in the specific decision context of technology options evaluation.

Keywords: Mozzarella, Whey, Dairy, Global Warming Potential, Multifunctionality

1. Introduction

In the dairy processing sector, saving energy and water use through developing and integrating different innovations is an important priority. Such innovations are being piloted in the EnReMilk project (an EC Framework Programme 7 project) in a German dairy facility that produces among others skim milk powder, and an Italian mozzarella facility. To support decisions on the selection of innovations from the EnReMilk project in these distinct facilities, Life Cycle Assessment (LCA) is considered as the most appropriate methodology. Environmental footprinting, i.e. calculating LCA impacts for distinct products, is gaining popularity because it provides easy-to-understand impact indicators that can be added up (Ridoutt and Pfister 2013). Companies, consumers and governments often take a simple choice oriented comparison mindset in which it is hard to consider external consequences of this choice. In the current decision context, external consequences outside the dairy processing facility, caused via market mechanisms, are not expected.

Although LCA studies are generally done according to the ISO-standards 14040 and 14044 (ISO Technical Committee ISO/TC 207 2006a,b), these standards leave several methodological choices up to the practitioner, which can strongly affect the results of the assessment (among others Yan et al. 2011, Kim et al. 2013). Among these choices, the approach for dealing with multifunctionality is an important point of attention in dairy processing. Multifunctionality arises in cheese production in the curd-whey separation, milk-cream separation, the milk-beef farming system and the feed-seed oil production system (Feitz, Lundie, Dennien, Morain, & Jones, 2007; Thoma, Jolliet, & Wang, 2013; Thomassen, Dalgaard, Heijungs, & de Boer, 2008; Tucker et al., 2010) and is an important methodological point of attention. How multifunctionality should be treated seems to become increasingly consistent, when considering the scientific and industrial guidance (Feitz et al. 2007, Aguirre-Villegas et al. 2012, European Dairy Association et al. 2015, IDF (International Dairy Federation) 2015).

Approaches that deal with the multifunctionality problem through allocation are relevant in the case of cheese, because avoiding allocation approaches (as recommended by ISO Technical Committee ISO/TC 207 (2006b)) cannot fully solve the problem in this case: Subdivision is not possible since the curd/whey separation reflects a chemical separation of milk; and Substitution is not possible because there is no realistic market-average alternative to producing ricotta, nor can a hypothetical alternative be considered that does not originate from the multifunctional process of curdling milk (Aguirre-Villegas et al. 2012).

In this paper, it will be evaluated in which way different allocation approaches affect the results of an LCA for a specific mozzarella producing facility. These findings will be discussed in the context of the application of the LCA, and critically reviewed in a broader context.

2. Methods

Goal and Scope: Since this paper aims to illustrate the effects of different allocation approaches, the global warming potential was used as a straight-forward and suitable indicator for the goal of this paper. Because the LCAs conducted in the EnReMilk project itself have a broader focus, these evaluate all impact categories from ReCiPe 2008 (Goedkoop et al. 2009). The functional unit has been defined as “1 kg of mozzarella for pizza applications at the gate of the mozzarella facility in the baseline year 2014”. The system boundaries of the overall LCA are from the cradle to the gate of the mozzarella facility, and all material inputs, consumables and capital goods have been included except office activities, facilities overhead and supporting services because these were estimated to contribute less than 1% of the total global warming potential.

Production description: Raw Milk is stored after delivery and pasteurized, and subsequently standardized by separating a small share of the milk fat (cream) from the raw milk. The standardized milk is curdled by addition of a bacterial starter culture and rennet in the substeps of pre-ripening, coagulation and curd cutting. The resulting (sweet) whey is drained and collected, and the curd left to ripen, from which additional (acid) whey is collected. Mozzarella is shaped from the ripened curd by a process of cutting, stretching and molding. The cylinder or ball shaped mozzarella is pre-cooled with water, cooled with ice water, and packaged for subsequent storage. Ricotta is produced through heating the sweet whey. Furthermore, cream is churned into butter. A complete picture with all main product and byproduct flows is shown in Figure 1.

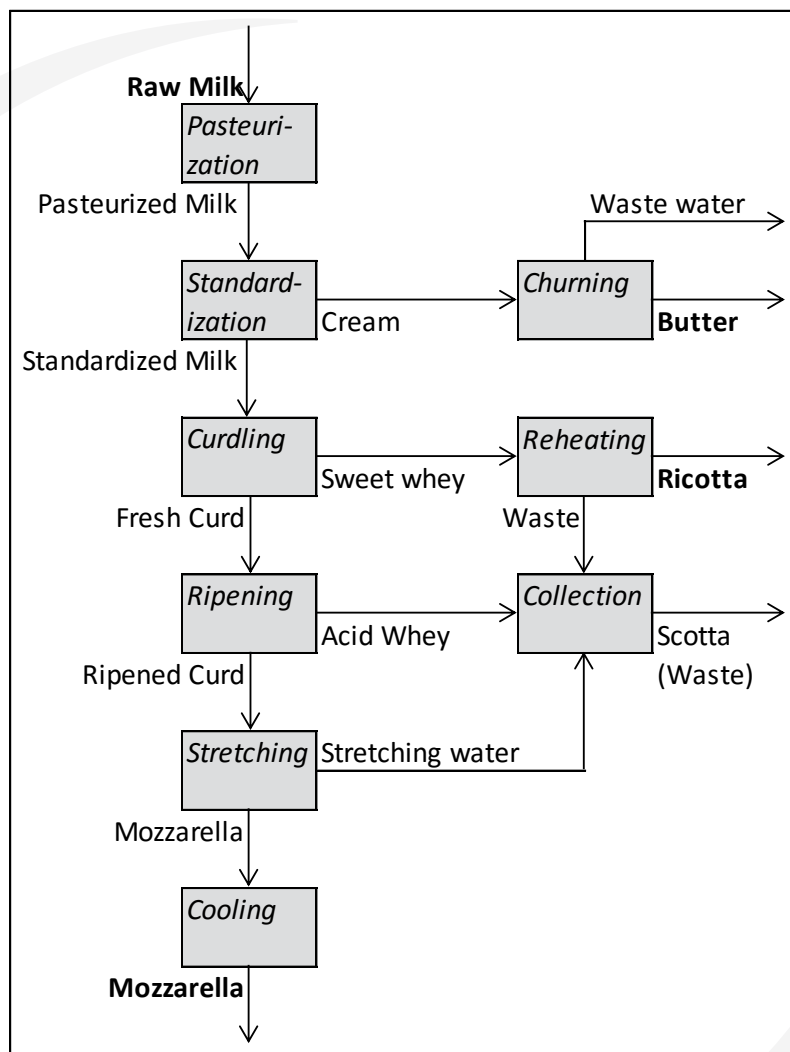


Figure 1: Overview of Mozzarella processing steps, excluding packaging

The remaining, protein-poor whey is combined with the acid whey and the waste water from mozzarella stretching. This combination of byproducts is called scotta, is sent off-site for waste treatment. Wastewater results from (pre-)cooling the mozzarella, butter production and from rinsing and cleaning equipment. Ice water cooling consumes significant amounts of electricity, whereas all other process steps consume much small amounts of electricity. Steam is consumed only during pasteurization, cleaning-in-place and mozzarella stretching, and water is consumed during pre-cooling and packaging.

Data Collection: Steam, electricity and water consumptions in the mozzarella facility were collected on the unit process level. This is possible because a monitoring system is being set up to evaluate the facility performance in different experimental technology pilots in the EnReMilk project. In addition to the baseline case of the mozzarella facility, a hypothetical scenario of moving from production of ricotta to whey powder was selected to illustrate the consequences of all four allocation approaches in an extreme case. The whey drying process was modelled using data from the skim milk powder facility in Germany, collected in the EnReMilk project. The selected facility data is representative for a typical day of production, excluding situations of intensive production or production problems. On a typical day, the mozzarella facility consumes 21 tons of milk and produces 3 tons of cow milk-based mozzarella cheese for pizza application, which is called fior di latte in Italy. As such, the facility has a limited size compared to cheese production facilities in the US (Aguirre-Villegas et al. 2012, Kim et al. 2013).

The impact of raw milk was derived from the Agri-footprint database, using economic allocation (Blonk Agri-footprint bv. 2014a, b). EcoInvent 3.2 processes (Weidema et al. 2013) were used to model the impact of electricity, steam, cleaning in place, waste water treatment and transport, augmented with specific grid mix from the International Energy Agency (IEA) (2014) and through personal interaction with the facility owner. Packaging of the final mozzarella product was included by following the draft PEFCR for Dairy (European Dairy Association et al. 2015) and including packaging raw materials from EcoInvent 3.2. SimaPro 8.2 was used for composing the model and extracting the results (Pré Consultants 2016).

Allocation: As discussed in the introduction, different allocation approaches can be identified. Firstly, the facility can be regarded as a whole with a total resource consumption (facility level, FL) or it can be subdivided into groups of unit processes that relate to all, a subset or one of the final products (process level, PL), as illustrated in Figure 2.

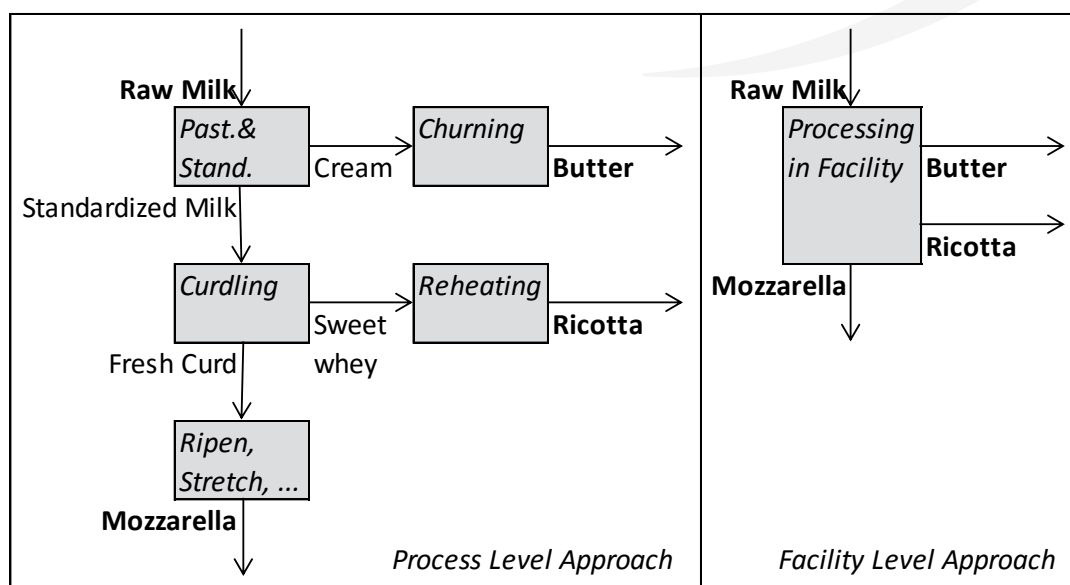


Figure 2: Two different allocation approaches for a mozzarella producing facility: on the left the process level approach, on the right the facility level approach. The distinction between dry matter allocation and economic allocation is not shown in this figure.

In the facility level approach, all impact is allocated between mozzarella, ricotta and butter. In contrast, two allocations are done in the process level approach: the impact upstream of the standardization is allocated between standardized milk and cream, and the impact upstream of curdling is allocated between sweet whey and fresh curd. Secondly, allocation between multiple flows from a process can be done according to dry matter content of the flows (dry matter allocation, DMA) or to the revenue generated with these flows (economic allocation, EA). Dry matter content data of all products were reported by the facility owner, as well as market prices of the final products. Market prices of intermediate products were derived from prices of raw milk and of the final products. These two choices lead to four allocation approaches.

3. Results

In Figure 3 the effects of the different allocation approaches can be seen for the entire cradle-to-gate assessment. It is clear that the raw milk production has the largest contribution with 61-77%, and that transport is the secondary contribution with 15-19% for all allocation approaches, when considering the baseline case (ricotta production). Figure 3 shows that less impact is allocated to mozzarella under dry matter allocation compared to economic allocation, for both facility and process level approaches, because mozzarella has a larger share in the total revenue than in the total dry matter utilized from the milk. The process level approach leads to a lower impact compared to the FL approach under DMA, because the allocation ratio between curd and sweet whey are different from the allocation ratio between mozzarella and ricotta. This is because curd and whey still include milk solids that ultimately go to waste (scotta), and could be corrected by only including the dry matter that is not wasted in the allocation factor calculation for curd and whey.

Under the dry matter facility level approach, the change in sweet whey processing (from ricotta to whey powder production) reduces the impact of mozzarella because the milk solids utilization has increased. On the other hand, under the dry matter process level approach, the mozzarella impact stays the same, because sweet whey processing is separated in the model. Under the economic allocation approaches on both levels, the change from ricotta to whey powder increases the impact of mozzarella, because less revenue is achieved by producing whey powder compared to ricotta.

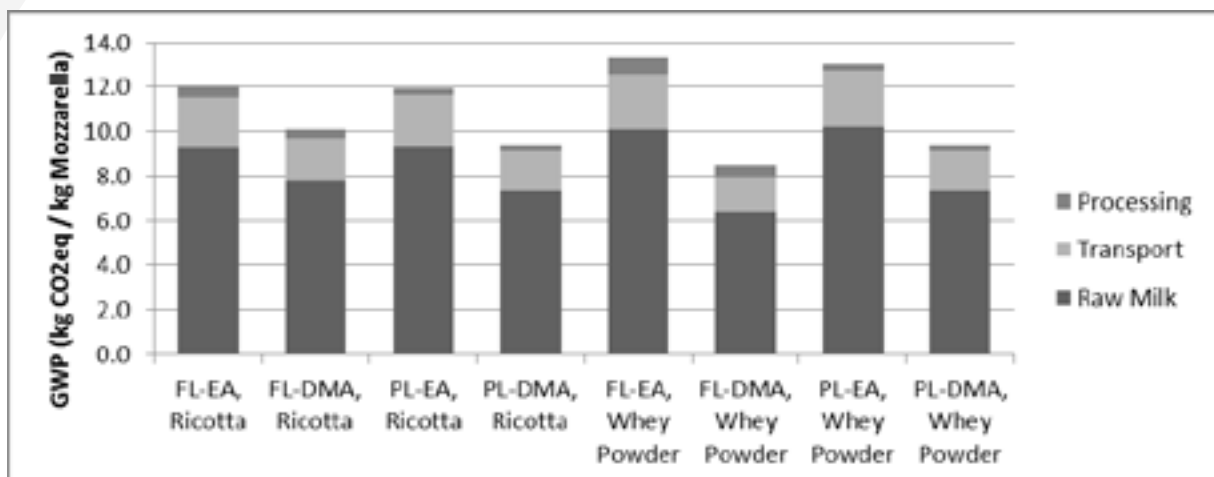


Figure 3: Cradle-to-gate carbon footprints (kg CO₂eq/kg of product) of mozzarella with contributions of raw milk, transport and processing, under different allocation approaches (FL=Facility level, PL=Process level, EA=Economic allocation, DMA=Dry matter allocation) for two scenarios: producing ricotta from sweet whey and producing whey powder from sweet whey

The contribution from the processing step is limited, compared to raw milk impacts and transport, but is affected by technological innovations within the dairy facility. For technology evaluation, it is specifically interesting how the impact of processing is distributed over the different products. As shown in Figure 4, mozzarella receives a larger share under economic allocation compared to dry matter allocation, because mozzarella has a larger share in the total revenue than in the total dry matter utilized from the milk. Mozzarella receives a much smaller share of the processing impact in

the process level approaches, because the large energy consumption in ricotta is more correctly attributed to the ricotta process, compared to the facility level approaches. The hypothetical change from ricotta to sweet whey production increases the total processing impact by 47%. Because this increase strongly affects the mozzarella contribution under the facility level approaches, the mozzarella impact is made strongly dependent on whether ricotta or whey powder is produced. The effect is most strong for economic allocation, because whey powder provides less revenue, while it increases dry matter utilization in dry matter allocation.

The effects of the trends described above translates into highly variable carbon footprints of individual products, as shown in Table 1. Mozzarella receives high impacts under facility level approaches, while whey products receive higher impacts under process level approaches, especially under dry matter allocation.

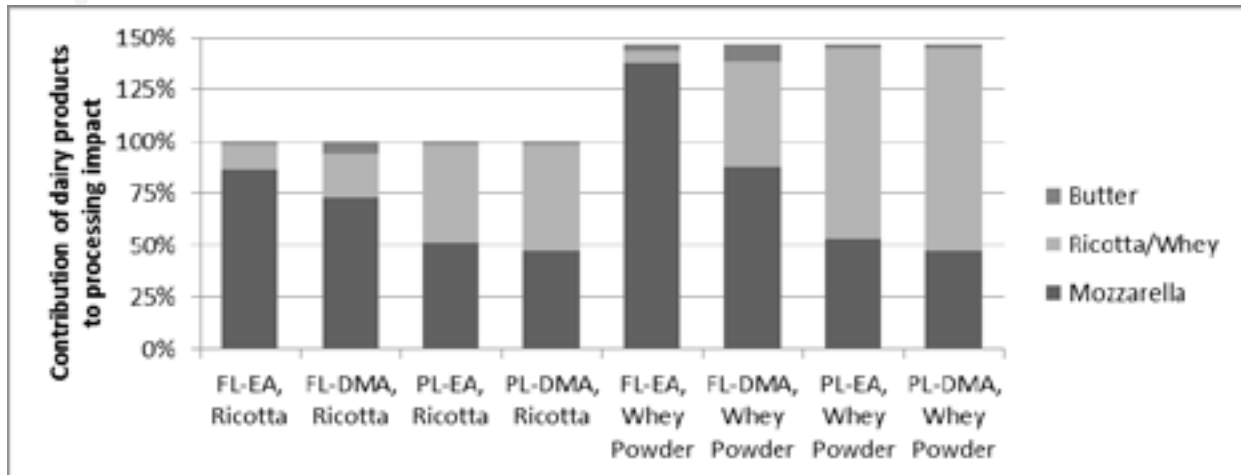


Figure 4: Percentage contributions of the products mozzarella, ricotta or whey, and butter, to the processing impact under the different allocation approaches, for the two scenarios. All data is relative to the impact of the ricotta scenario, so that the whey powder scenarios have a higher total impact.

Table 1: Carbon Footprints (kg CO₂eq/kg of product) of the processing step in the mozzarella facilities under the different allocation approaches, for the two scenarios

Allocation Approaches	Scenario: Producing Ricotta		Producing Whey Powder			
	FL	PL	FL	PL		
EA	Mozzarella	0.48	0.28	Mozzarella	0.76	0.29
	Ricotta	0.15	0.60	Whey Powder	0.12	1.91
	Butter	0.25	0.20	Butter	0.39	0.20
DMA	Mozzarella	0.40	0.26	Mozzarella	0.48	0.26
	Ricotta	0.27	0.65	Whey Powder	1.07	2.04
	Butter	0.80	0.20	Butter	0.96	0.20

4. Discussion

The results show that different allocation approaches affect both the contributions upstream of the processing facility and the impact of the processing.

The process level approach correctly separates the considerations on how much energy to invest in whey processing from the mozzarella production, since a significant change in the whey processing does not affect the mozzarella production economically or physically. The facility level approach attributes some of the impact from whey processing to mozzarella. Combined with dry matter allocation this could give the perverse incentive of moving to whey powder production, in which more energy is consumed. The process level approach gives relevant information in the decision context of technology evaluation, so that the additional detail and effort could be justified (Ekvall and Finnveden 2001). Furthermore this subdivision is recommended by the ISO standard 14040.

However, the process level approach is only possible under high data availability. Although intensive contact with facility owners and technical experts is possible in the EnReMilk project, it turned out to be challenging to be completely certain of the mass and dry matter balances that were needed to achieve the highest possible reliability. It was noted before that it can be challenging to account for all resource uses on a process level (Ekvall and Finnveden 2001), and hybrid approaches have been proposed (IDF (International Dairy Federation) 2015). The impact of the same products from facilities with different product portfolios will be most comparable if these facilities use the process level approach. However, if facilities with lower data availability are not able to follow the process level approach, all facilities should use the facility level approach, because using different allocation approaches would make results even less comparable.

For technology evaluation, the process level approach is preferred, because they give better consistency for all products and comparability between different technology alternatives. In a scientific context, the unavailability of data is a bad argument to say that the data does not need to be collected, but when footprinting products from several businesses such practical considerations play a larger role. Thus, a trade-off can be recognized between, on the one hand, the benefits of internal comparability and consistency in the process level approach, and the practicality and external comparability in the facility level approach on the other hand.

The choice between dry matter and economic allocation approaches is more fundamental. While a causal relationship between the allocation property and the inputs of the multifunctional process is recommended (ISO Technical Committee ISO/TC 207 2006b), a causal relationship between the allocation property and the incentive to produce a product is also thinkable. Examples of incentives are generating revenue, nourishing people, etc., with properties like price, nutrient content and energy value. Economic allocation is criticized because prices of dairy products are variable and would introduce variability in economic allocation factors across time and regions (Feitz et al. 2007, Aguirre-Villegas et al. 2012). The variation in prices translates to variation in production incentives, which in fact should be addressed by using market-standardized price averages over several years. Dry matter allocation approaches follow the physical flows throughout the facility, and is more practical in dairy processing, because price information is not required and dry matter tracking is common in the industry (Aguirre-Villegas et al. 2012).

Considering these prior observations as well as the decision context of the dairy facility, which is essentially economic, economic allocation is preferred. The dry matter approach is not entirely consistent in this context, because it is influenced by the share of milk solids that is wasted. Since waste is produced when it is economically unattractive to turn a process flow into a valuable product, economic considerations are introduced in the dry matter approach. Furthermore, the implied causal link between the dry matter content and the environmental impact, is only valid for the raw milk impacts, but not for the processing impacts. Economic allocation is more practical in this context because prices vary less on the Italian mozzarella market than globally. Using averaged prices also matches the allocation with the time frames of decision context for technological innovations and other production changes.

For technology option evaluation, the product perspective is useful, because a producer is most rewarded by improving the impact of the main product. The total environmental impact of the product portfolio (1kg mozzarella plus the accompanying whey product and butter) will be an additional useful perspective, because it illustrates the total change from one technology to another, and excludes the high sensitivity to allocation. Figure 4 provides an illustration of this.

The process level approach may be valuable for replication in other production systems, in which byproduct flows separate from the main product flow early in the processing facility, or require large energy use in byproduct processing. Examples are whey processing (Aguirre-Villegas et al. 2012), drying of byproducts from sugar production or from wet milling wheat grain, and drying brewers grains from beer brewing.

5. Conclusions

This paper evaluated how different allocation approaches affect the results of an LCA for a specific mozzarella producing facility. The process level approach provides useful detail that clarifies the incentives for a producer to improve processes that are specific to each coproduct: Improving

processes in mozzarella production accurately benefits the mozzarella impact, and the same is true for whey processing. Economic allocation relates incentives for production to the different coproducts while dry matter allocation also includes economic considerations through the definition of waste. The different allocation approaches may result in different technology preferences. In the evaluation of technology options in one dairy facility, process level economic allocation was found most unambiguous and straightforward to interpret. Although a growing consensus on allocation may be recognized for footprinting in the developments of industry guidelines, the goal and context of different LCA studies may best be served with different allocation approaches. The ISO standard and scientific papers can be interpreted from different angles, which allows for these different approaches. This indicates that the debate on allocation is not likely to be finished.

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228. Evaluation based on data quality of allocation methods for calculation of carbon footprint of grass-based dairy production

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Abstract

A major methodological issue for life cycle assessment (LCA), commonly used to quantify greenhouse gas (GHG) emissions from livestock systems, is allocation from multifunctional processes. When a process produces more than one output, the environmental burden has to be assigned between the outputs, such as milk and meat from a dairy cow. National and international guidelines provide different recommendations on allocation. In the absence of an objective function for allocation, a decision must be made considering a range of factors. The objective of this study was to evaluate 7 methods of allocation to calculate the global warming potential (GWP) of the economically average (€/ha) dairy farm in Ireland considering both milk and meat outputs. The methods were: economic, energy, protein, energy, mass of liveweight, mass of carcass weight and physical causality. The data quality for each method was expressed using a pedigree matrix score based on reliability of the source, completeness, temporal applicability, geographical alignment and technological appropriateness. Scenario analysis was used to compare the normalised GWP per functional unit (FU) from the different allocation methods, for the best and worst third of farms (in economic terms, €/ha) in the national farm survey. For the average farm, the allocation factors for milk ranged from 75% (physical causality) to 89% (mass of carcass weight), which in turn resulted in a GWP / FU, from 1.04 to 1.22 kg CO₂-eq/kg FPCM. Pedigree

scores ranged from 6.0 to 17.1 with protein and economic allocation having the best pedigree. It was concluded that the choice of allocation method should be based on the quality of the data available, that allocation method has a large effect on the results and that a range of allocation methods should be deployed to understand the uncertainty associated with the decision.

Introduction

With the global human population predicted to increase to over 9 billion by 2050 there will be a rise in consumption of bovine milk and meat products (FAO, 2009). Increasing primary production from large ruminant systems to meet greater demand is expected to increase greenhouse gas (GHG) emissions. To tackle this problem, EU nations have legally agreed as part of the 2020 climate and energy bill to reduce GHG emissions from the non-emission trading sector, which includes agriculture. The EU aims to reduce these emissions by 10% (20% in an Irish context) by 2020 relative to 2005 levels.

Life cycle assessment (LCA), an internationally standardized methodology (ISO14040), is the preferred method to estimate GHG emissions from agricultural systems (IDF, 2010; Thomassen and De Boer, 2005). A single impact LCA focused on GHG emissions is commonly referred to as a carbon footprint (CF). A major methodological issue of LCA is allocation between multiple outputs of a process. When a system such as a dairy farm or a process produces more than one output, the environmental burden such as GHG emission, has to be allocated between these outputs, e.g. milk and meat.

Generally LCA guidelines (BSI, 2011; IDF, 2013) recommend where achievable, allocation should be avoided, but where this is not possible guidelines differ on how to allocate, e.g. PAS2050 recommends using economic relationships while IDF (2013) recommend using physical relationships. It is well documented that for LCA studies data quality has a significant impact on the uncertainty and robustness of the results (Henriksson *et al*

2011; Rice *et al* forthcoming). While ISO standards recommend that data quality be reported, this is not that common in LCA studies. Consequently, it is typically necessary for LCA practitioners to make judgement with respect to the accuracy of LCA outcomes. While a data quality scoring/judgement matrix has been developed (Weidema and Wesnaes, 1996), the concept has never been applied in the context of allocation and the choice of method. **The goal (objective) of this study was to evaluate and assess the suitability of seven different allocation methods applied to grass based dairy production in terms of data quality.**

The methods of allocation assessed were: economic, energy, emergy (novel application), protein, mass of liveweight (LW), mass of carcass weight (CW) and physical causality. The data quality (pedigree) was assessed using (1) reliability of the source and completeness; (2) temporal correlation; (3) geographical correlation; and (4) technological correlation, which is in keeping with the data quality requirement stipulations set out by the ISO (2006).

Materials and methods

The data used for completing the LCA of grass based milk production were derived from the 2012 Irish National Farm Survey (Table 1) (NFS, Hennessey *et al* 2013) as previously described by O'Brien *et al.*, (2015). The survey was carried out on 256 dairy farms in 2012 and was weighted according to farm area to represent the national population of specialized dairy farms (15,600). All the dairy farms in the NFS used grass-based spring calving with seasonal the milk supply matched to grass growth patterns, in order to maximise grazed grass intake (Kennedy *et al* 2005).

The LCA methodology was applied according to the ISO (2006) guidelines. The goal was to evaluate 7 methods of allocation using an economically average Irish dairy (€ / ha) farm between milk and meat. The system boundary was 'cradle to farm gate', including foreground processes of milk production and background processes for production and transportation of mineral fertilizer, cultivation, processing and transportation of concentrate

feed. Infrastructure (animal housing, slurry storage facilities, and roads), machinery (tractor, milk cooling system) were not included, as these have a small influence on the GHG's from milk production (O'Brien *et al* 2014). The functional unit was kg of fat and protein corrected milk (FPCM) calculated as to 4% fat and 3.3% protein using (Clark *et al* 2001) where FPCM (kg/yr) = Production (kg/yr) × (0.1226 × Fat % + 0.0776 × True Protein % + 0.2534).

The GHG emissions, methane (CH₄), nitrous oxide (N₂O), carbon dioxide (CO₂) and halocarbons (F-gases) were calculated using the cradle to farm-gate LCA model of O'Brien *et al* (2014) that was certified by the Carbon Trust. The model used previously published algorithms and data from the NFS to calculate on and off-farm GHG emissions using Intergovernmental Panel on Climate Change (IPCC) guidelines (IPCC, 2006) and Irish GHG national inventory methods (O'Brien *et al* 2014). Within the model, the various GHG emissions were converted to CO₂-equivalents (CO₂-eq) using the IPCC (2007; O'Brien *et al* 2014) revised guidelines for GWP and summed to establish the farm CO₂-eq emissions. The GWP conversion factors for the key GHG emissions in the model were 1 for CO₂, 25 for CH₄ and 298 for N₂O, assuming a 100 year time horizon. The CF of both milk and meat were estimated by allocating the GHG emissions between milk and meat.

Energy allocation is based on the 'embodied energy' in milk and meat from culled cows and surplus calves and quantified in solar energy equivalents (seJ). Allocation by physical causality was based on the IDF (2010) guidelines and reflected the underlying use of feed energy by the dairy animals to produce milk and meat. Economic allocation was based on sales receipts for milk and animals from culled cows and surplus calves at the farm gate. Mass allocation was based on the weight of milk and weight of culled dairy cows and surplus calves. The mass of animals was calculated in terms of LW (liveweight) and CW (carcass weight). Allocation by protein was expressed in kg of protein and based on the edible protein in milk and meat from culled cows and surplus calves. Energy allocation was expressed in

joules (J) of energy and based on edible energy in milk and meat from culled cows and surplus calves.

The quality of the data was assessed by the pedigree matrix of Weidema and Wesnaes (1996) for each allocation method. The overall pedigree score was calculated for each allocation method based on the sum of the component scores, weighted by proportional contribution to the calculation where this could be assessed (e.g. proportional mass of milk and meat). The methods were then ranked based on pedigree score. For each allocation method the highest possible score was 25 and the lowest was 5 and a lower score represents a better pedigree of data.

Results and discussion

From running the activity data through the LCA model of O'Brien et al.,(2014),it was estimated that 477791.17 kg Co₂e were generated by the 'mean' group of farms .The results of this study has shown significant differences in the allocation proportion to milk (and associated meat co-product) (Table 3). For the 'Average' farm, the allocation factors ranged from 75% (physical causality) to 89% (mass of carcass weight)(Table 3), which in turn resulted in over a 17% difference in the CF values, i.e. 1.04 – 1.22 kgCo₂-eq/kg FPCM, depending on which allocation method was used. This range in allocation of emissions to milk in turn resulted in over an 11-fold difference in the CF values for meat, i.e. 0.61 – 7.49 kgCo₂-eq/kg meat. Regarding both FPCM and meat, physical causality resulted in the smallest difference i.e. 2.5% less for FPCM and 15% more for meat, compared to when economic allocation was applied. Moreover, the application of allocation by way of mass of carcass weight (CW) resulted in the greatest difference, i.e. 15% more for FPCM and 90% less for meat, compared to when economic allocation was applied. The CF's were achieved with data of widely varying pedigree (Table 4), from the simpler allocation methods (mass LW, mass CW), to the more complex methods (energy, emergy, physical causality (Table 4).With regards to FPCM, both protein content and economic allocation methods had the best

pedigree of data, with a pedigree matrix score of 6 (Table 4), whilst the energy content allocation method had the worst pedigree of data with a pedigree matrix score of 17.1 (Table 4). With regards to meat, protein content had the best pedigree of data, with a pedigree matrix score of 6.5 (Table 4), whilst both the energy content and energy allocation methods had the same pedigree matrix score of 19.4 (Table 4), indicating that they had the worst pedigree of data behind them.

Conclusion

Allocation method has a large effect on the CF result, > 11 fold difference in the case of meat. Based on pedigree score, protein content followed by the simple mass allocation methods by LW or CW were best for milk. Energy and energy were of poorest pedigree and the others fitted in between. In most cases it was only the scores for one or two indicators that dominated the final pedigree score for each method. This was also observed by Weidema and Wesnaes (1996), so if a particular method is to be used for theoretical reasons, then focused effort will be required to ensure the best possible data are available in order to justify its use from a data pedigree perspective. A further reason to be careful with the more complex methods is that they are built on a foundation of the simple methods with a cascade of additional data. This study showed the importance of using country, technology and temporally specific data so the goal and scope specification for the study should be consistent with the time that can be committed to the allocation calculations. It was also noted that when assessing meat co-products the method chosen can be used to bias the study. From the data presented here it seems that physical causality will be biased in favour of milk, and in the case of physical causality, obtaining good pedigree data to justify such an approach is difficult. A range of methods should be deployed to understand the uncertainty associated with the decision.

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Table 1. Key technical measures collected by Hennessey et al. (2013) for the mean of a sample of 221 Irish dairy farms ranked in terms of gross margin/ha.

Item	Mean
Dairy farm area, ha	35
Milking cows, number	67
Culled cows, %	17
Stocking rate, cows/ha	1.89
Soil class (1 = yes)	59%
FPCM yield, kg/cow	5181
Fat, %	3.94
Protein, %	3.4
FPCM ¹ yield, kg/ha	9776
Milk yield, kg/farm	352915.62
Concentrates, kg DM/cow	898
Grazing days	239
N fertilizer, kg/ha	196
Purchased fuel, l/ha	110
Electricity, kWh/cow	182
Gross margin, €/ha	1,758

¹ Fat and protein corrected milk, standard milk corrected to 4% fat and 3.3% protein

Table 2. Data quality Pedigree Matrix by Weidema and Wesnaes (1996)

Indicators	Indicator Score				
	1	2	3	4	5
Independent of the study in which the data are applied:					
Reliability of the source	Verified data based on measurements	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 and industrial 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 a smaller number of sites but for adequate periods	Representative data from an adequate number of sites but for shorter periods	Representative data from a smaller number of sites and shorter periods, or incomplete data from an adequate number of sites and periods	Representativeness unknown or incomplete data from a smaller number of sites and /or from shorter periods
Dependent on the goal and scope of the study:					
Temporal correlation	Less than 3 years of difference to year of study	Less than 6 years of difference to year of study	Less than 10 years of difference to year of study	Less than 15 years of difference to year of study	Age unknown or more than 15 years of difference to year of study
Geographical correlation	Data from area under study	Average data from larger area in which the area under study is included	Data from area with similar production conditions	Data from area with slightly similar production conditions	Data from an unknown area or with very different production conditions
Technological correlation	Data from enterprises, processes and	Data from processes and materials under	Data from processes and materials under	Data on related processes and materials but from	Unknown technology or data on related processes or materials



	materials under study	study but from different enterprises	study but from different technology	same technology	but from different technology
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Table 3. The allocation proportion for milk under the seven allocation methods, for the mean category of Irish dairy farms in terms of gross margin/ha

¹Greenhouse Gas measured in kg CO₂e

	Method of allocation						
	Mass of Liveweight	Mass of carcass weight	Protein content	Energy content	Emergy	Economic	Physical causality
GHG¹ allocated to FPCM	88%	89%	83%	81%	84%	77%	75%

Table 4. Application of Pedigree matrix (Weidema and Wesnaes, 1996) to the methods of allocation, with regards to milk and meat

	Method of Allocation						
	Mass of Liveweight	Mass of carcass weight	Protein content	Energy content	Emergy	Economic	Physical causality
Pedigree matrix score FPCM	6.1	6.1	6	17.1	16.9	6	9.5
Pedigree matrix score meat	6.6	6.6	6.5	19.4	19.4	8.4	9.5

Table 5. Individual pedigree component scores for contributors to allocation calculations

	Indicator							Method of allocation where used					
	Reliability	Completeness	Temporal	Geographical	Technological	Total	Mass of Liveweight	Mass of carcass weight	Protein content	Energy content	Emergy	Economic	Physical causality
Meat	Number of culled cows	1	1	1	1	1	5	x1	x	x	x	x	x
	Culled cow KgLW/hd	1	1	1	2	1	6	x	x	x	x	x	x
	Culled cow KO%	1	4	4	3	3	15	x	x	x	x	x	x
	Culled cow KgCW/hd	1	1	1	2	1	6	x	x	x	x	x	x
	Culled.cow carcass protein %	1	5	2	5	1	14	x	x	x	x	x	x
	Culled.cow carcass fat%	1	5	2	4	5	17			x	x	x	x
	Number of surplus calves	1	1	1	1	1	5	x	x	x	x	x	x
	Surplus calf kgLW/hd	2	2	3	1	1	9	x	x	x	x	x	x
	Surplus calf Ko%	1	4	4	3	4	16	x	x	x	x	x	x
	Surplus calf kgCW/hd	1	1	1	2	1	6	x	x	x	x	x	x
	Surp.calf carcass protein%	1	4	5	4	5	19	x	x	x	x	x	x
	Surp.calf carcass fat%	1	5	2	4	5	17			x	x	x	x
	Prot.Energy(J)	1	5	4	5	5	20			x	x	x	x
	Fat energy(J)	1	5	4	4	5	19			x	x	x	x
	Beef energy(σε/J)	1	5	4	4	5	19				x	x	x
	Culled cow €/kgCW	1	5	1	2	1	10			x		x	x
	Surplus male dairy calf (€/hd)	3	5	1	2	1	12					x	x
	Surplus female beef calf (€/hd)	3	5	1	2	1	12					x	x
	Milk	Parameter (Thoma 2012)	1	1	3	4	3	12					
Non corrected milk,kg		1	2	1	1	1	6	x	x	x	x	x	x
Total protein produced,kg		1	2	1	1	2	7	x	x	x	x	x	x
Total fat produced,kg		1	2	1	1	4	9			x	x	x	x
Total Lactose produced(kg)		1	3	3	3	2	12			x	x	x	x
Prot.Energy(J)		1	5	4	5	5	20			x	x	x	x

	Fat energy(J)	1	5	4	4	5	19	X	X	
	Carbohydrate energy (J)	1	5	4	4	5	19	X	X	
	Milk energy(σε/J)	1	5	4	4	5	19		X	
	Fat,kg sold	1	2	1	1	4	9			X
	Protein,kg sold	1	2	1	1	2	7			X
	Revenue milk fat(€)	1	1	1	2	2	7			X
	Revenue milk protein(€)	1	1	1	2	2	7			X
	C parameter	1	1	1	2	2	7			X

21. Environmental impact of South African milk: A comparison of Ayrshire and Holstein breeds

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ABSTRACT

In South Africa and other emerging economies, the demand for dairy products is growing rapidly. In view of the considerable environmental impacts caused by dairy production systems, environmental mitigation strategies are required. One factor affecting the environmental impact of dairy products is the breed type. In this case study, a Life Cycle Assessment (LCA) was performed for raw and processed milk from Holstein and Ayrshire cows. Primary data was provided by five South African dairy farms and two dairy plants.

The raw milk production is the decisive factor when analysing the environmental impact of dairy products. Regardless of the breed type, the carbon footprint of raw milk is dominated by direct methane and dinitrogen monoxide emissions, while the production of the feed, as well as the infrastructure and energy demand of the farms, play a major role in all other impact indicators. On farms where both Ayrshire and Holstein cows are kept, the environmental impact is lower for milk from Holstein cows. However, differences between farms outweigh differences between breeds. The implementation of best practices therefore has substantial environmental mitigation potential for livestock farming in South Africa.

Keywords: LCA; dairy; cattle; agriculture

1. Introduction

As in most emerging economies, livestock is one of the fastest growing sectors of the agricultural economy in South Africa (DAFF, 2013). Over the past ten years, milk production increased by 26%, 49% and 73% in South Africa, Brazil and India respectively (DAFF, 2015; FAO, 2015). The global dairy sector accounts for 4% of anthropogenic greenhouse gas emissions and is therefore a major contributor to climate change (FAO, 2010). Gerber et al. (2013) state that interventions in order to mitigate these emissions are largely based on technologies and practices that increase production efficiency. Consequently, the feed quality, the feeding regime and the breeding are seen as key factors in reducing the emissions of milk production (Gerber et al., 2013). The breed type can also have a significant influence on the environmental impact of dairy products. Capper & Cady (2012) show that the carbon footprint of cheese production in the US is lower for Jersey cows than for high-producing Holstein cows due to a reduced energy requirement of the Jersey herd. Given the increasing demand for dairy products in South Africa and other emerging economies, the question arises as to whether the selection of the breed type used to meet this rising demand can help to reduce the environmental impact of dairy production systems. In South Africa, Holstein, Jersey, Guernsey and Ayrshire are the major dairy breeds (DAFF, 2015). This study analyses and compares the environmental impact of raw and processed milk of Holstein and Ayrshire cows from five farms located in the South African province of KwaZulu-Natal.

2. Goal and Scope

In order to evaluate the environmental impacts of dairy farming in the South African Province of KwaZulu-Natal (KZN) and of the subsequent dairy processing, a cradle-to-gate Life Cycle Assessment (LCA) was performed. While the functional unit is 1 kg for processed dairy products at a dairy plant, the environmental impact of raw milk at the farm gate was computed for three scenarios: on a weight basis, on a fat and protein corrected milk (FPCM) weight basis, and on a price basis. This differentiation was introduced to account for different milk qualities depending on the breed: Ayrshire milk is advertised as having a superior taste by one of South Africa's largest retail stores. This is reflected in a slightly higher price for Ayrshire milk as compared to conventional milk.

The LCA covers the rearing of a female calf, the keeping of the adult dairy cow, the transport of the raw milk to a dairy factory and the milk processing. Data on milk production and processing was collected on five farms in KZN and two dairy plants in KZN and the Western Cape, respectively. The data collection took place in August 2014. The environmental impacts were assessed using six different impact indicators, namely climate change according to the Intergovernmental Panel on

Climate Change (IPCC, 2013), the cumulative non-renewable energy demand according to Hischier et al. (2010), land use according to Frischknecht et al. (2013), freshwater and marine eutrophication according to Goedkoop et al. (2009) and freshwater ecotoxicity according to Rosenbaum et al. (2008). Background data for the life cycle inventories was taken from the international ecoinvent v3.1 database using the system model “allocation, recycled content” (ecoinvent Centre, 2014). The life cycle inventories and the impact assessment were issued with the SimaPro software v8.1 (PRé Consultants, 2016).

3. Life Cycle Inventory

On average, Holstein and Ayrshire cows participating in the South African National Milk Recording Scheme produce 7441 kg and 6072 kg of milk per lactation, respectively (Ramatsoma et al., 2015). Roughly 40% of the South African herds have a milk yield below 5500 kg per cow and for 7% of the herds the yearly milk production per cow is higher than 9125 kg (Milk SA, 2015). For the herds considered in this case study, the yearly milk yield per cow ranges from 4822 kg to 9200 kg. The milk yield and other key characteristics of these herds are listed in Table 1. Furthermore, all farmers participating in this case study provided detailed information on their feeding regimes, on their electricity demand and on the water use for irrigation and other purposes. Direct methane emissions were calculated using the Tier 2 approach described by the IPCC (2006). In addition, direct and indirect nitrous oxide emissions and ammonia emissions were computed. The allocation at farm-level is based on the physiological feed requirements to produce milk and meat. This approach is recommended by the International Dairy Federation IDF (2015). Furthermore, economic allocation was used to distribute the environmental impact of meat between calves and cull dairy cows.

Table 1: Characteristics of the five dairy farms. Values in italic have been estimated due to lacking primary data. H: Holstein; A: Ayrshire

	Farm 1		Farm 2		Farm 3		Farm 4		Farm 5	
Breed	H	A	H	H	A	H	H	H	H	
Milk yield (l/a)	8300	7300	9200	8000	4822	<i>8250*</i>	7500			
Live Weight (kg)	545	600	675	<i>560**</i>	420	460	650			
Age at fist calving (months)	27	27.5	27.5	27	26	28	28			
Number of lactations	2.4	5.5	5.5	2.8	3.6	3.6	3.8			
Protein content of milk	3.20%	3.24%	3.15%	3.20%	3.34%	3.29%	3.25%			
Fat content of milk	3.40%	4.10%	3.48%	3.60%	3.80%	3.48%	3.75%			
Price for milk (ZAR/l) ***	13.5	14.5	13.5	13.5	14.5	13.5	13.5			

* The yearly milk yield specified by Farm 4 for Holstein cows is 4282 kg. However, Farm 4 indicated that all Holstein cows were in their first lactation. For that reason, the average milk yield of Holstein cows on Farm 1, 2, 3 and 5 has been assumed to be true for Holstein cows on Farm 4

** Average life weight of Holstein cows on farms 1, 2, 3 and 5

*** Prices are in South African rand (ZAR) and refer to consumer prices for conventional milk and Ayrshire milk, respectively. Prices were taken from www.woolworths.co.za (retrieved 21.4.2016)

The seven datasets created based on the information in Table 1 were aggregated to a single production mix which was used to issue the life cycle inventories of processed dairy products. The two dairy plants considered in this case study provided information on their raw milk, energy, water and chemicals input and on the production volumes of different dairy products. Also transport, infrastructure and waste water were included in the inventory. Allocation of environmental impacts between different dairy products was performed based on the dry matter content of the dairy products as recommended by the IDF (2015).

4. Life Cycle Impact Assessment

Raw milk production in KwaZulu-Natal is associated with greenhouse gas emissions of between 1.2 and 2.0 kg CO₂-eq/kg. Direct methane and dinitrogen monoxide emissions account for 66%-73% of the climate impact of raw milk (Figure 1).

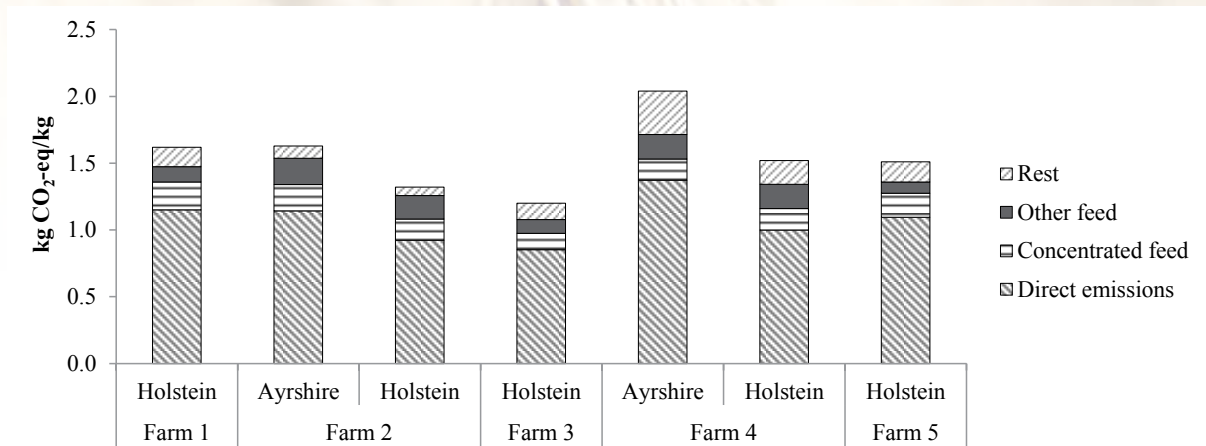


Figure 1: Greenhouse gas emissions of raw milk at five farms in KwaZulu-Natal

On farms 2 and 4, where both Ayrshire and Holstein cows are kept, the climate impact is lower for milk from Holstein cows, even when using FPCM or price as a functional unit (Figure 2). Holstein cows have a higher milk yield than Ayrshire cows and therefore a lower specific feed intake than Ayrshire cows. However, variations across farms are larger than variations across breeds.

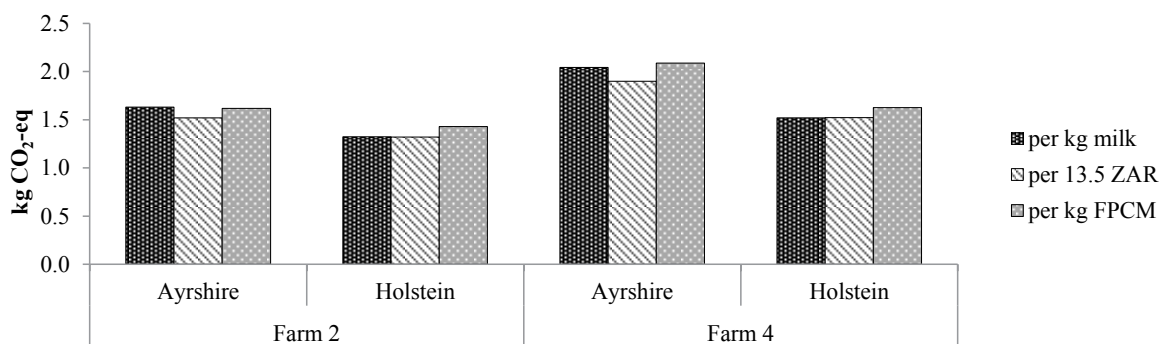


Figure 2: Greenhouse gas emissions of Ayrshire and Holstein milk at Farm 2 and Farm 4 depending on the functional unit used. 13.5 South African rand (ZAR) is the price assumed per liter of conventional milk (see Table 1).

While direct emissions play a major role in the greenhouse gas emissions of raw milk, other environmental impact indicators are dominated by the production of concentrated feed and the housing system (Figure 3). The non-renewable energy demand of raw milk depends largely on the electricity demand of the dairy farm and the use of diesel for the production of concentrate feed. The eutrophication potential can mainly be related to the use of fertilizers in the production of wheat and maize which are the main components of concentrate feed. The freshwater ecotoxicity is primarily caused by herbicide emissions from maize production. Land is predominantly used to grow wheat and maize for the production of concentrate feed.

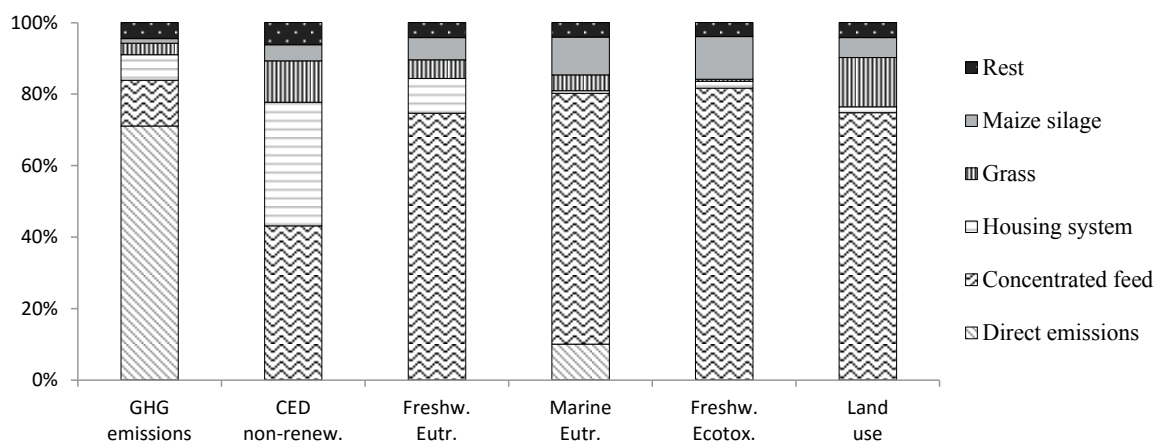


Figure 3: Contribution analysis for raw milk at Farm 1 for six impact indicators

The carbon footprint of dairy products can mainly be attributed to the production of raw milk, while milk transport as well as the infrastructure and energy demand of the dairy factory only play a minor role (Figure 4). For pasteurised milk, the greenhouse gas emissions range from 1.3 kg CO₂-eq/kg for milk with a fat content of 0.5% (dairy factory A) to 1.9 kg CO₂-eq/kg for milk from dairy factory B. Due to its high dry matter content, butter is the product with the highest climate impact (13 kg CO₂-eq/kg). In general, dairy products from dairy factory A have a lower environmental impact than products from dairy factory B. This can be attributed to a higher resource efficiency of dairy factory A as compared to dairy factory B.

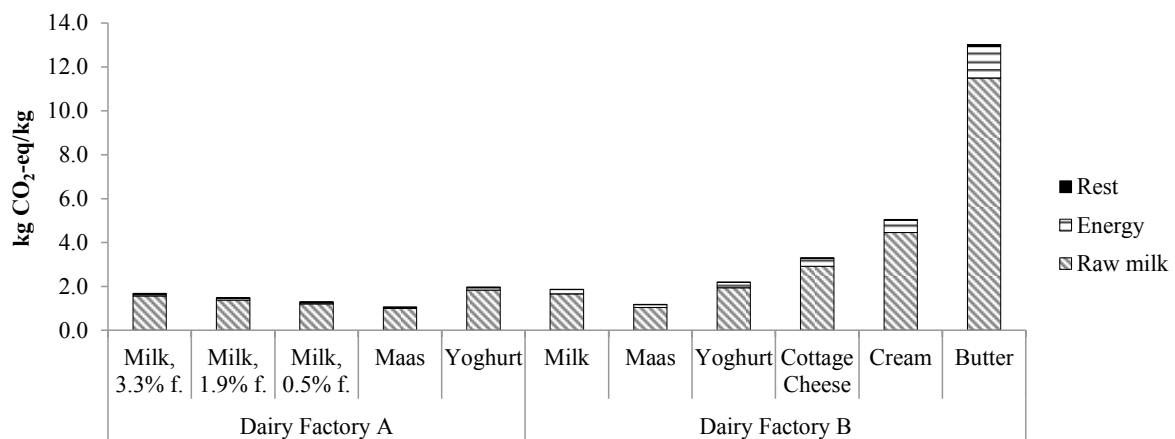


Figure 4: Contribution of the raw milk production, the energy demand of the milk processing and other processes to the carbon footprint of dairy products from two South African dairy factories. Maas: fermented milk (common South African dairy product); f.: fat.

Not only the carbon footprint but also the results for all other impact indicators are dominated by the production of raw milk. The energy demand of the dairy factory is only relevant when considering the cumulative energy demand (CED) of processed milk. In comparison with dairy factory B, dairy factory A uses 74% less energy for the processing of milk.

5. Interpretation and conclusion

Average greenhouse gas emissions from the production of raw milk in the South African province of KwaZulu-Natal amount to 1.6 kg CO₂-eq per kg (ranging from 1.2 to 2.0 kg CO₂-eq/kg). Similar results were published for other countries. The ecoinvent database reports a carbon footprint of 1.4 kg CO₂-eq/kg FPCM for Canadian raw milk (ecoinvent Centre, 2015) and the Agribalyse database shows greenhouse gas emissions amounting to 0.99 kg CO₂-eq/kg FPCM for French cow milk (ADEME, 2015).

The environmental impact of raw milk varies between farms and between breeds. Cross-farm differences can largely be attributed to differences in the resource efficiencies of farms. The environmental impact is lowest for milk from Holstein cows on farms 2 and 3 for almost all impact indicators. While milk production on farm 2 is characterized by a high energy intake of dairy cows coupled with a high milk yield, the energy intake of cows on farm 3 is low with at the same time relatively high milk yields (Figure 5).

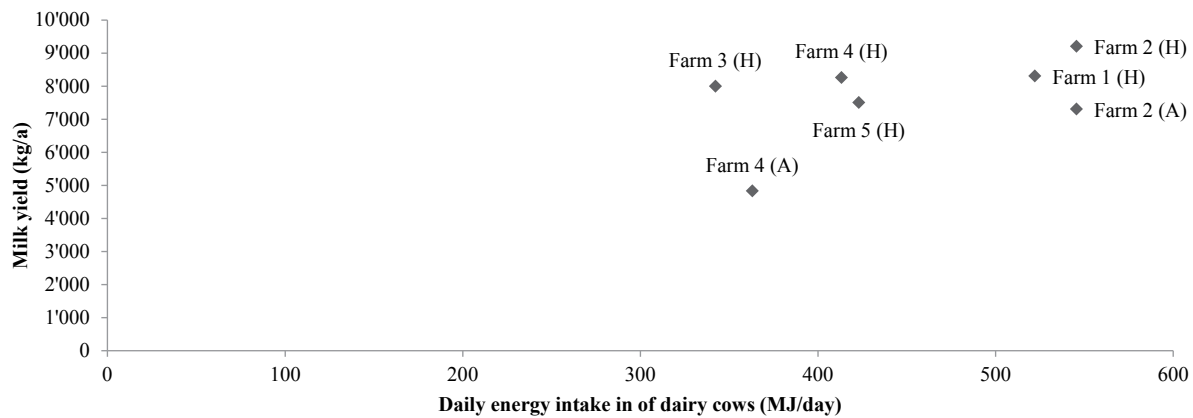


Figure 5: Relationship between daily energy intake of dairy cows and milk yield for the five farms. A: Ayrshire; H: Holstein

In contrast to Capper & Cady (2012), who compared Holstein and Jersey milk and showed that the environmental impact of dairy production is lower when using milk from a breed with a lower milk yield (Jersey), the present study found that the high-yielding Holstein breed performs better than the lower-yielding Ayrshire breed. However, differences between farms outweigh differences between breeds.

Direct methane emissions are a key factor in the carbon footprint of milk and, thus, measures to reduce emissions from enteric fermentation are decisive. According to Knapp et al. (2014) the most promising strategies to reduce enteric methane emissions combine genetic and management approaches. Effective measures include genetic selection for animals with lesser enteric methane emissions and higher production efficiency (genetic approach), as well as practices to reduce non-voluntary culling and diseases, improvements in nutrition and the reduction of stress factors such as heat (management approaches) (Knapp et al., 2014). Hristov et al. (2015) suggest using feed supplements to achieve a significant reduction in methane emissions from enteric fermentation. In an experiment with 48 Holstein cows the use of the methane inhibitor 3-nitrooxypropanol (3NOP) led to a reduction of 30% in rumen methane emissions, while milk production was not affected by the inhibitor (Hristov et al., 2015).

For impact indicators other than climate change measures to reduce the environmental burden related to the production of concentrated feed are decisive. Especially for irrigated crops, significant reductions can be achieved through the use of renewable energies in the agricultural production: Wettstein et al. (2016) show that the use of solar power for irrigation reduces the cumulative non-renewable energy demand and the freshwater eutrophication potential of irrigated South African maize by 43% and 12%, respectively.

In conclusion, a considerable variability of the environmental impact of milk from the South African province of KwaZulu-Natal can be observed. Compared to milk from Ayrshire cows, the environmental impact of milk from high-yielding Holstein cows tends to be smaller, but differences between farms are greater than differences between breeds. These findings indicate that the implementation of best practices has a substantial environmental mitigation potential for livestock farming in South Africa.

6. Acknowledgement

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307. LIFE CARBON DAIRY, the milk carbon footprint of 3,300 French dairy farms

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ABSTRACT

The French Livestock Institute (Institut de l'Élevage), in association with three partners, has launched the project LIFE CARBON DAIRY with the main objective to promote an approach allowing milk production to reduce by 20% the milk carbon footprint at farm level over 10 years. The three other leading partners are key players in the French dairy sector from the advisory services on dairy farms (Dairy advisors enterprises as ECEL and Chamber of agriculture) to dairy processors and CNIEL. To achieve the goal, project' partners developed a Life Cycle Assessment (LCA) tool named CAP'2ER® tool (Moreau et al., submitted to LCA food 2016) to assess the milk carbon footprint on dairy farms in France. Answering the LCA approach, the milk carbon footprint assessed in CAP'2ER® is covering the greenhouse gases (GHG) emissions to determine the Gross Carbon Footprint (GCF) and carbon sequestration to assess the Net Carbon Footprint (NCF). Applied on 3,316 farms representing various milk production systems in France, the project provides a good overview of the average national milk carbon footprint. In parallel, each individual evaluation gives participating farmer management factors to identify opportunities to improve farm efficiency and reach the carbon reduction target. On the 3,316 farms assessed, the average GCF is 1.04 kg CO₂e per liter Fat and Protein Corrected Milk (FPCM) and NCF, 0.93 kg CO₂e per liter FPCM. In the over way, carbon sequestration compensates 11% of GHG emissions. Variations in GCF (CV = 15%) are explained by differences in farm management. Practices with the largest impact on milk carbon footprint average are milk yield, age at first calving, quantity of concentrate, N-fertilizer used (organic and chemical) and fuel consumed. Farms with the lowest GCF (10% of farms) have an average carbon footprint of 0.85 kg CO₂e per liter FPCM and confirm that farm efficiency is a way to reduce carbon intensity.

Keywords: carbon footprint, French dairy, milk production, soil carbon sequestration.

1. Introduction

Agriculture is a contributor to global greenhouse gases (GHG) emissions, and particularly methane and nitrous oxide. In France, agriculture sector contributes to 18% of overall global GHG emissions (CITEPA, 2015) and 8% comes from ruminants taking into account animals and manure management. On the one hand, French government's targets to cut GHG emissions by 75% of 1990 levels by 2050. For agricultural sector a first target set under the Décret n°2015-1491 is to cut GHG emissions by 12% of 2015 levels by 2028. On the other hand, consumers ask for more information on the environmental impacts of products and their influence on GHG emissions. Meanwhile, retailer companies are enlarging their requirements for their suppliers to environmental impact and specifically carbon footprint.

As practices exist to reduce GHG emissions from livestock activities, we have to demonstrate their effectiveness on a widespread basis through the mobilization of business/professional and structural efforts of the entire livestock industry.

By involving a large number of farmers in six pilot regions which account for 65% of the French milk delivery and are representative of different climate conditions and feeding strategy, LIFE CARBON DAIRY represent a real opportunity to disseminate the carbon footprint on a large scale with the main objective to promote an approach allowing milk production at farm level to reduce by 20% the milk carbon footprint over 10 years.

To answer this goal, the milk carbon footprint calculation is a good way to disseminate to farmers about GHG emissions of their dairy system activity issue and how they can reach these tough environmental target. The objective of this study was to build an approach to apply Life Cycle assessment (LCA) methodology at a farm level, to determine the average

milk carbon footprint produced in France, to assess the sensitivity of the carbon footprint to farming practices and to promote a large carbon action plan in dairy farms.

2. Methods

Carbon footprint calculation

Dairy farms were assessed individually with the CAP'2ER® (environment footprint calculator and decision making for ruminants production systems) tool developed by Institut de l'Élevage (Moreau et al., submitted to LCA food 2016) for French production context. Answering to Life Cycle Assessment (LCA) standards, the system boundaries covered by CAP'2ER represents 'cradle-to-farm-gate' of the dairy unit (on-farm impacts plus embodied impacts from inputs used on the farm; Figure 1). The methodology developed to assess carbon footprint is based on international methodologies (IPCC Tiers 3, CML, LEAP guidelines). The tool also evaluates positive contribution as carbon sequestration and emissions to the environment are expressed in connection with the primary function represented by the product. The functional unit is the quantity of milk in kg Fat and Protein Corrected Milk (FPCM) leaving the farms. To standardize GHG emissions, the International Panel on Climate Change has established the global warming potential equivalence index to convert GHG to CO₂e units. In our model, the conversion factors are 25 kg of CO₂e/kg CH₄ and 298 kg CO₂e/kg N₂O (IPCC, 2007).

The GHG emissions from dairy unit are allocated between milk and meat (surplus calves and cull dairy cows) according a biophysical allocation rule based on feed energy required to produce milk and meat respectively (LEAP 2015).

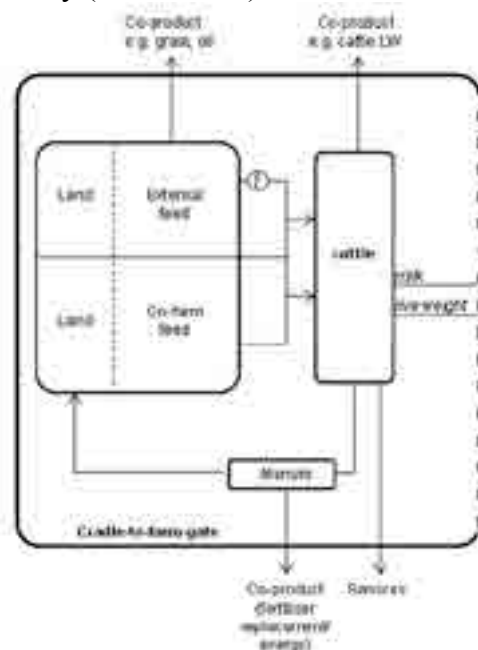


Figure 1: System boundaries (adapted from LEAP large ruminants)

Carbon sequestration

We have assumed that grassland and hedge increase the carbon content of soil every year. Respectively 570 kg carbon per year per ha and 125 kg carbon per 100 ml of hedges. On the other way, arable lands without grass in the crop rotation were considered to decrease the soil carbon content by 160 kg carbon per ha every year (Dolle et al., 2013). But including grass in the rotation cycle on arable lands can increase biomass return in soil's organic matter, and reduce disturbance to the soil through tillage. Thus, the average soil carbon balance per year for the crop rotation with grass was calculated with the assumption that crop decrease soil

carbon content by 950 kg carbon per year and ha and grass increase carbon soil content by 570 kg carbon per year and ha.

Data collection

Technical data were collected on an annual base (2013 and 2014) at the farm level and from a number of producers across six regions and three main forage systems (“Grass system” with less than 20% maize in the forage area, mixt system (“Grass-Maize system”) with grass and maize between 20 and 40% of maize in the forage area and “Maize system” with more than 40% maize in the forage area).

Calculate the milk carbon footprint need a large amount and complex farm level data. The data collection process was achieved on each farm with trained agents from ECEL companies (dairy advisors enterprises). The questionnaire consisted of 150 questions divided in six sections regarding: 1) herd demographics and milk production; 2) animal housing and manure management; 3) crop production for on farm produced feed; 4) feed rations and purchased feed; 5) energy use (fuel and electricity); 6) general information. To ensure the validity of data collected, there are checking points in the questionnaire to test consistency (e.g. comparison between animals dry matter intake need and produced and purchased feed) and the most important parameters were test within an expected range of values.

3. Results

General farm characteristics

In 2016, 3,316 farms have been assessed. General farm characteristics of this sample are different from the French average dairy farm because of an over representation of Western systems (83%), more intensive systems, using maize and producing more milk per cow than mountain grass-based systems. There is a large variation in characteristic and performance data between farms resulting from different farming conditions (climate and soil type), farmer strategies (breed and production system, size ...) and management practices (efficiency, health...). On this farm sample, the average dairy farm produces 467,000 liters of milk with 62 milking cows and 95 ha (Table 1). The stocking rate is 1.5 livestock units per hectare dedicated to the dairy herd. The age at first calving is 29 months and the replacement rate 29%. Dairy cows diet is mainly composed of maize silage with 60% of the total forage dry matter intake (DMI), grazed grass is 29%. Concentrate consumed by dairy cows represents 166 g per liter of milk produced. On farm area N fertilization is 145 kg N/ha.

Table 1: dairy farms characteristics, standard deviation (SD), coefficient of variation (CV), lower and upper 10%

Farm characteristic	Mean	SD	CV%	Lower 10%	Upper 10%
Total Farm size, ha	95	53	0.56	47	159
Area dedicated to the dairy herd, ha	64	32	0.50	35	101
Number of milking cows	62	24	0.39	38	93
Total Milk production, *1000 l	467	199	0.43	265	721
Labour productivity, *1000 l/labour unit	342	168	0.49	179	536
Stocking rate, LU/ha	1.49	0.40	0.27	1.01	1.96
Milk production standard, l FPCM/cow/year	7487	1143	0.15	5989	8808
Fat, g/l	40.1	2.0	0.05	38.0	42.3
Protein, g/l	32.4	1.3	0.04	31.1	33.8
Grazing days, days	184	37	0.20	137	226
Quantity of concentrate, g/l milk	166	56	0.34	107	236

N surplus at farm level, kgN/ha	78	40	31	31.324	125
Hedge, ml/ha	109	49	0.45	24	155
Electricity, kWh/LU	333	107	0.32	207	444
Fuel, l/ha	148	50	0.34	87	211

Milk carbon footprint

Performing a LCA study on a large number of commercial dairy farms provides an insight into the variation between milk carbon footprints that may be related to variation in farm performance and characteristics (e.g. milk yield per cow, forage system, ...).

On the 3,316 farms, the average milk Gross Carbon Footprint (GCF) is 1.04 kg CO₂e per liter FPCM (Table 2) with no significant difference between forage systems. Variations in GHG (+/- 15%) are the same whatever the forage system. On this farm sample, the carbon sequestration associated to grasslands (permanent and temporary) and hedges compensates the GCF by 11%, with variations between the forage systems. The grass system compensates till 30% of his GHG emissions, therefore, in Grass system, Net Carbon Footprint (NCF) is considerably lower than other systems.

Table 2. Milk carbon footprint of different forage systems

	Farm type – Forage system			
	Maize system	Maize/Grass system	Grass system	Total
Number of farms	1,418	1,536	362	3,316
Gross Carbon Footprint, Kg CO ₂ e/ l FPCM	1.05	1.04	1.05	1.04
<i>Coefficient of Variation</i>	14%	14%	19%	15%
Carbon sequestration, Kg CO ₂ e/ l FPCM	0.06	0.11	0.30	0.11
<i>Coefficient of Variation</i>	72%	64%	77%	106%
Net Carbon Footprint, Kg CO ₂ e/ l FPCM	0.99	0.93	0.76	0.93
<i>Coefficient of Variation</i>	15%	15%	31%	18%

A focus on the 10% of farms getting the lower milk GCF results is realized. In this group composed by 333 farms, the average GCF is 0.85 kg CO₂e per liter FPCM and the NCF is 0.75, with 12% carbon sequestration (Figure 2). The variation between total farms and this sample is 18%. These results ensure the possibility to reduce by 20% the national milk carbon footprint at farm level over 10 years.

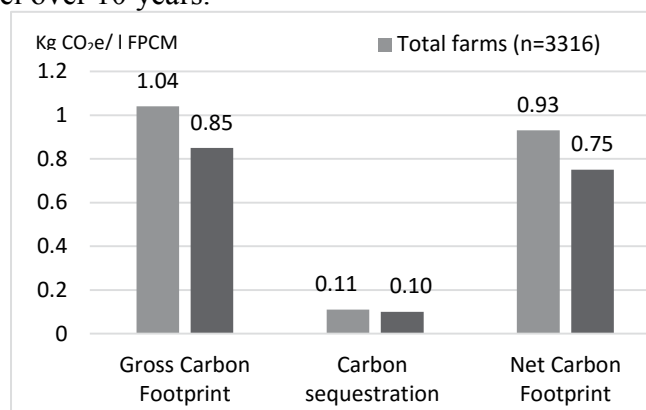


Figure 2: National and 10% lower milk carbon

The major contributor to GHG emissions is the enteric methane (49%) which average 147 kg CH₄/cow. The next largest contributor is the manure management (storage and grazing

cattle) which produce methane and nitrous oxide (18%), followed by nitrous oxide emissions from fertilizer application (11%). The remainder emissions are from inputs purchasing with fertilizer (3%), feed (12%), and direct energies (fuel and electricity, 5%).



Farm practices and carbon footprint

As the objective of the study is to identify the management practices that could be efficient to reduce the milk carbon intensity, relationship between the milk carbon footprint and farm practices and performance were tested using the correlation analysis.

Firstly, correlation is realized between milk GCF and farm practices and parameters. On the one hand, no correlation between the GCF and the herd size or the part of maize in the system was found (Table 3). On the other hand, strong linear correlations were found between the milk GCF and the milk production per cow, the quantity of concentrate, the age at first calving or the nitrogen surplus at farm level.

Table 3: Correlation between various farm performance and characteristics parameters and milk gross carbon footprint (GCF)

Parameter	Correlation with GCF	P-value
Farm size, ha	0.063	<0.001
Number of cows	-0.008	0.626
Total milk production, liter FCPM/cow/year	-0.401	<0.001
% Maize / Total area	-0.002	0.916
Age at first calving, months	0.255	<0.001
Replacement rate, %	0.082	<0.001
Concentrate rate, g/l milk	0.236	<0.001
N-fertilizer use, kg N/ha dairy herd	0.060	0.001
Fuel consumption, l/ha	-0.049	0.005
N surplus at farm level, kgN/ha	0.187	<0.001
Permanent grassland area, ha	0.085	<0.001
Carbon sequestration, kg CO ₂ e/l FPCM	0.199	<0.001

The analysis of the 10% lowest milk GCF farms show that their average GCF is 0.85 kg CO₂e/l FPCM. These farms are performant for not only one parameter accounted for the majority of variation between farm's milk GCF but for almost all. They produce more milk for the same level of concentrate and rear less heifers for the herd replacement. The carbon sequestration is quite the same than average with 0.10 kg CO₂e/l FPCM.

Secondly, the same analysis is realized between the milk NCF and farm practices and parameters. Results are different from the previous ones. Age at first calving, milk production per cow or quantity of concentrate are not correlated with milk NCF whereas part of maize in the system, use of fertilizer or fuel used are strongly correlated (Table 4). The strong correlation between permanent grassland area and NCF shows that carbon sequestration is an important parameter composing the NCF.

Table 4: Correlation between various farm performance and characteristics parameters and milk net carbon footprint (NCF)

Parameter	Correlation with NCF	P-value
Farm size, ha	- 0.062	<0.001
Number of cows	0.040	0.022
Total milk production, liter FCPM/cow/year	- 0.043	0.013
% Maize / Total area	0.415	<0.001
Age at first calving, months	- 0.025	0.150
Replacement rate, %	0.058	0.001
Concentrate rate, g/l milk	0.165	<0.001

N-fertilizer use, kg N/ha dairy herd	0.385	<0.001
Fuel consumption, l/ha	0.248	<0.001
N surplus at farm level, kgN/ha	0.365	<0.001
Permanent grassland area, ha	- 0.291	<0.001
Carbon sequestration, kg CO ₂ e/l FPCM	- 0.493	<0.001

4. Discussion

The LCA results obtained for this average dairy farm sample appeared to be consistent with other LCA studies of milk production obtained on the same perimeter, except for Irish dairy systems based on extensive grass dairy farms with a lower milk GCF (Table 5). The FAO report gives an average milk GCF of 1 kg CO₂e/kg FPCM (Gerber et al., 2010).

Table 5: Comparison of the milk GCF

Countries	Milk GCF (kg CO ₂ eq/kg FPCM)	Publication
Ireland	0.74	O'Brien et al (2014)
New Zealand	0.93	Basset-Mens et al (2009)
	0.78	Dollé et al (2016)
France	1.06	Dollé et al (2016)
	0.95	Dollé et al (2015)
Sweden	1.16	Flysjö et al (2011)
World	1.0	Gerber et al (2010)

Our results give a 10% higher milk GCF than found with previous French study (Dolle et al., 2015), but the carbon sequestration is similar. A smaller sample and the profile of the farm with a greater efficiency could explain the lower milk GCF in the previous study.

Our results show that milk GCF is similar between the three forage systems. This means that low environmental impacts can be reached in every dairy system and farm. The grass-based system farms have better NCF results, but GCF, reflects of management efficiency is the same whatever the system. It is possible to say that, in order to reach low milk carbon footprints, two main strategies exist:

- on the one hand, the correlation between farm performance and milk GCF shows that dairy producer can mitigate GHG emissions with practices to improve efficiency. Farms increasing milk production per cow and reducing the number of replacement animals emit less methane enteric per unit of milk. And then, if associated with an efficient use of inputs (fertilizer, feedstuffs and fuel) that reduce direct and indirect CO₂ emissions, farms present a low net carbon footprint, even if their carbon sequestration is lower than average result. To perform better, this type of system can increase their grassland area in order to store more carbon,
- on the other hand, the grass-based system farms that seem to be less performant on the milk production per cow and replacement herd, present a higher carbon sequestration, because of a larger part of grassland storing carbon, and can compensate their methane enteric emissions. This management is a second way to mitigate milk carbon footprint. The most efficient farmers in grass-based system are efficient in the use of their input too. Their milk carbon footprint can be very close to 0.0 kg CO₂e/kg FPCM.

It should be mentioned that it is more appropriate to explore the milk GCF than the milk NCF to analyze the system efficiency. Indeed, in some cases, farms are not efficient and GHG emissions are very high, but these results can be hidden thanks to an important carbon sequestration mitigating their milk NCF.

To finish, LCA tool as CAP'2ER® gives an interesting opportunity to assess a large number of farm and to involve farmers in decreasing GHG emissions from dairy sector. Given the wide range in milk GCF of 18%, significant opportunities exist for most farms to reduce their milk carbon footprint.

5. Conclusions

The project LIFE CARBON DAIRY represents the opportunity to determine carbon footprint of a first sample of 3,316 dairy farms. The project is up going for two more years with the objective to create a national dynamic and involve 600 news farms in the disposal. These first results are satisfactory and ensure our objective to reduce by 20% the milk carbon footprint at farm level over 10 years. Focused on the relationship between dairy performance and milk carbon footprint, our investigations concern also the other environmental burdens as air and water quality, energy consumption, biodiversity,

The CAP'2ER® tool calculates these others indicators as eutrophication, acidification, total energy use and biodiversity to quantify wider environmental impacts and positive contributions of milk production. Future analyses will be carry out on the carbon dairy database to get the global impact of milk production and to test if the carbon mitigation strategies identified don't cause any undesirable changes in other aspects of environmental performance.

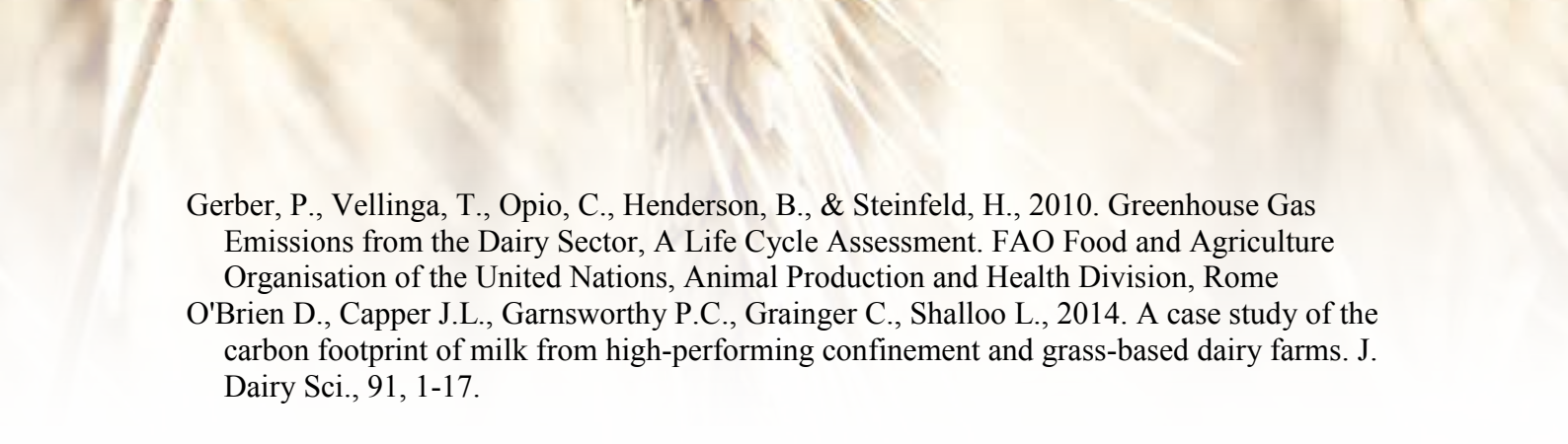
The dissemination of these assessments on farm is permitting to implement a national carbon and environmental action plan to increase dairy sustainability and communicate with stakeholders on the progress done by the dairy sector to reduce environmental impact.

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265. The effect of field drainage on productivity and environmental impact of grass based dairy production systems

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ABSTRACT

The aim of this study was to quantify the greenhouse gas (GHG) emissions of a typical Irish dairy farm. The study considered two farms on poorly drained soil with and without drainage. System simulation and life cycle assessment methodology were combined to calculate the emissions. GHG emissions were quantified per unit of energy corrected milk. The GHG emissions for the farm with field drainage was 1.15 kg CO₂ equiv. per unit functional unit while for the farm without field drainage was 1.18 kg CO₂ equiv. per unit functional unit. It can be concluded that field drainage can improve the productivity of the farm thus reducing the GHG emissions.

Keywords: life cycle assessment, system simulation, dairy production, artificial drainage

1. Introduction

Globally, dairy production is an important source of greenhouse gas (GHG) emissions and accounts for 3% of global emissions (O'Brien *et al.* 2014). In Ireland, grass based dairy production is a key agricultural enterprise (O'Brien *et al.* 2014). The Irish dairy industry is expected to expand by 50% with the abolishment of EU milk quota (DAFF 2010). The Irish Environmental Protection Agency has estimated an increase of 12% in the GHG emission as consequence of expected increases in production (EPA 2012). Many farms in north-western Europe suffer from both poorly drained soil and high precipitation levels (Tuohy *et al.* 2016). Dairy production systems on well and poorly-drained soils differ in system properties and management with poorly drained sites having lower productivity (Fitzgerald *et al.* 2008). The trafficability and workability for field operations depend on soil moisture status (Earl 1997). The profitability of milk production systems on well-drained soil is generally greater than on heavy, wet soils (Shalloo *et al.* 2004). Effective land drainage can improve the output of marginal lands (Tuohy *et al.* 2016), for instance subsurface tile drains have been widely installed in the US Midwest to facilitate improved crop production (Williams *et al.* 2015). Shallow drainage techniques are used for low permeability soils to drain excess rainwater in grassland systems (Tuohy *et al.* 2016).

Life cycle assessment (LCA) is widely used to evaluate the GHG emissions from milk production (O'Brien *et al.* 2014); (Chen *et al.* 2016). LCA of agricultural systems typically evaluate environmental impacts up to the farm gate instead of end of life (O'Brien *et al.* 2012). *Dairy_sim* was developed to model pasture-based spring milk production for different climate and soil types (Fitzgerald *et al.* 2008). *Dairy_sim* simulates an optimum farm in which the feed demand of the herd is met by herbage produced, limited concentrate input and minimum housing days (Fitzgerald *et al.* 2008). The three components of *Dairy_sim* are: a herbage growth model, an intake and grazing model and a nutritional energy demand model, combined with a framework for operational management. The output of *Dairy_sim* can be used as part of the inventory for LCA. Fitzgerald *et al.* (2008), simulated grass based dairy production on poorly drained soil in Ireland using *Dairy_sim* where the optimized system was considered to be the equivalent of having field drains installed.

Subsurface drainage of imperfectly drained soils removes surplus soil water and a significant amount of water contaminants (Monaghan *et al.* 2016), (Williams *et al.* 2015). It is also confirmed that sub-surface drains transport N, especially nitrate to streams and open drainage ditches (Williams *et al.* 2015). The Intergovernmental Panel on Climate Change (IPCC) have specified different emission factor for the managed/drainage soil to calculate direct and indirect N₂O emissions.

The objective of the study was to evaluate GHG emissions of an optimized Irish pasture-based system on poorly drained soil type without and with field drains.

2. Methods

Dairy_sim was used to find optimum management practices for a theoretical 20 ha dairy unit supporting 2 dairy LU (live unit) ha⁻¹, yielding 5191 kg cow⁻¹ yr⁻¹ on poorly-drained soil before and after field drain installation. Thirty years (1981-2010) weather data for Fermoy, Co. Cork, Ireland were used for the simulations. Management decisions related to silage were taken from the National Dairy Blueprint described by O'Loughlin *et al.* (2001) and related to fertilizer were taken from Humphreys *et al.* (2008). *Dairy_sim* was initialized using Fitzgerald *et al.* (2008), with stocking rate 1.9 cow ha⁻¹, yielding 5871 kg cow⁻¹ yr⁻¹ dairy farm on poorly drained soil at Kilmaley, Co. Clare, Ireland.

The goal of the study was to quantify GHG emissions from a grass based dairy system in Ireland with and without artificial drainage to compare the emissions. The LCA model was developed using GaBi 6 (thinkstep 2014) software. The system produced milk from spring calving dairy cows. Only dairy cows were considered on the farm. The system boundary was from cradle to farm gate. On farm process included the milking unit, manure storage, cattle housing, manure application, grassland (grass and silage), fertilizer application, electricity and diesel use. Off farm processes include manufacturing and transportation of concentrate feed and fertilizer. The functional unit was defined as 1kg of ECM with the reference flow of the herd output in 1 year, where ECM = milk delivered * (0.25 + 0.122 * %fat + 0.077 * %protein) (Yan *et al.* 2013). The fat and protein percentage were assumed to be 3.94 and 3.4, respectively (Hennessy *et al.* 2013). The allocation between milk (88%) and meat (12%) was based on energy and protein requirement (O'Brien *et al.* 2012).

Partial inventory for the LCA was derived from *Dairy_sim* outputs. Energy use for grassland management, fertilizer and slurry spreading was taken from Yan *et al.* (2013). The data for fertilizer processing, cattle housing, manure storage and spreading were taken from Ecoinvent (2014). Concentrate feed mix was taken from O'Brien *et al.* (2012).

Methane (CH₄) emission factors for enteric fermentation (106.2 kg CH₄ head⁻¹ yr⁻¹) and manure management (15.9 kg CH₄ head⁻¹ yr⁻¹) were taken from O'Mara (2006). CH₄ emission from slurry spreading (autumn slurry: 6.8 g CH₄ m⁻³; spring slurry: 12 g CH₄ m⁻³) was taken from Chadwick *et al.* (2000). Direct and indirect nitrous oxide (N₂O) emission factors for manure storage (Direct: slurry: 0.002 N₂O-N (kg N)⁻¹; manure: 0.005 N₂O-N (kg N)⁻¹; Indirect: 0.01 kg N₂O-N), grassland (direct and indirect: 0.01 kg N₂O-N) were taken from IPCC (2006). N₂O emissions from grassland include emissions from manure excretion, fertilizer and manure spreading. Emission factor for direct N₂O emission from field drainage were taken as 1.6 kg N₂O-N ha⁻¹ (Hiraishi *et al.* 2014). Indirect N₂O leaching emission from field drainage were 0.0025 kg N₂O-N kg⁻¹ mineral N whereas without field drains were 0.0075 kg N₂O-N kg⁻¹ mineral N (IPCC 2006). The emission factor for indirect N₂O emissions from field drains was taken as 0.0025 instead of 0.015 kg N₂O-N kg⁻¹ mineral N, as the latter is considered to be very high (IPCC 2006). Ammonia (NH₃) emission factors for manure storage (94 g m⁻² over 30 days) were taken from Duffy *et al.* (2011). NH₃ emission factor for manure spreading (solid: 0.81; autumn slurry: 0.37; spring slurry: 0.60 of TAN applied) were taken from Webb and Misselbrook (2004).

3. Results

Comparison of initial parameterization compared to Shalloo *et al.* (2004), Fitzgerald *et al.* (2008) (Table 1) indicated minor differences, mainly cause by the different weather data because of climate period used. The differences in the simulated optimum management practices of the dairy unit on poorly-drained soil before and after installation of field drains (Table 2) indicated that with field drains the dairy unit can sustain 1.95 cows ha⁻¹ whereas with no drains it can only sustain 1.55 cows ha⁻¹. Housing time was 44 days more for the poorly drained unit without field drains. The amount of N

fertilizer input were largely unchanged. Figure 1 shows the GHG emissions for the dairy farms with poorly drained soil and field drains. For poorly drained soil with no drains the emissions was 1.18 kg CO₂ equiv. per kg ECM whereas with field drains installed it was 1.15 kg CO₂ equiv. per kg ECM.

Table 1: Comparison of *Dairy_sim* parameterization results with previous studies

Property	Unit	Poorly drained soil Shalloo <i>et al.</i> (2004)	Poorly drained soil Fitzgerald <i>et al.</i> (2008)	Dairy_sim simulated farm
Stocking rate	cows ha ⁻¹	1.89	1.90	1.90
Fertilizer	kg ha ⁻¹	238	236	232
Concentrate	kg cow ⁻¹ yr ⁻¹	759	455	464
Grass intake	kg cow ⁻¹ yr ⁻¹	2121	1949	2332
Silage intake	kg cow ⁻¹ yr ⁻¹	2375	2355	2225
Grazing time	days	149	177	187

Table 2: Management practices derived from *Dairy_sim* for a dairy farm on poorly drained soil with and without drains installed assuming national average milk production

Property	Unit	Poorly drained soil	Field drains installed
Cows	numbers	31	39
Stocking rate	cows ha ⁻¹	1.55	1.95
Milk output	kg cow ⁻¹ yr ⁻¹	5191	5191
Fertilizer	kg ha ⁻¹	195	197
Concentrate	kg cow ⁻¹ yr ⁻¹	471	545
Grass intake	kg cow ⁻¹ yr ⁻¹	1953	1996
Silage intake	kg cow ⁻¹ yr ⁻¹	2502	2272
Housing time	days	205	161

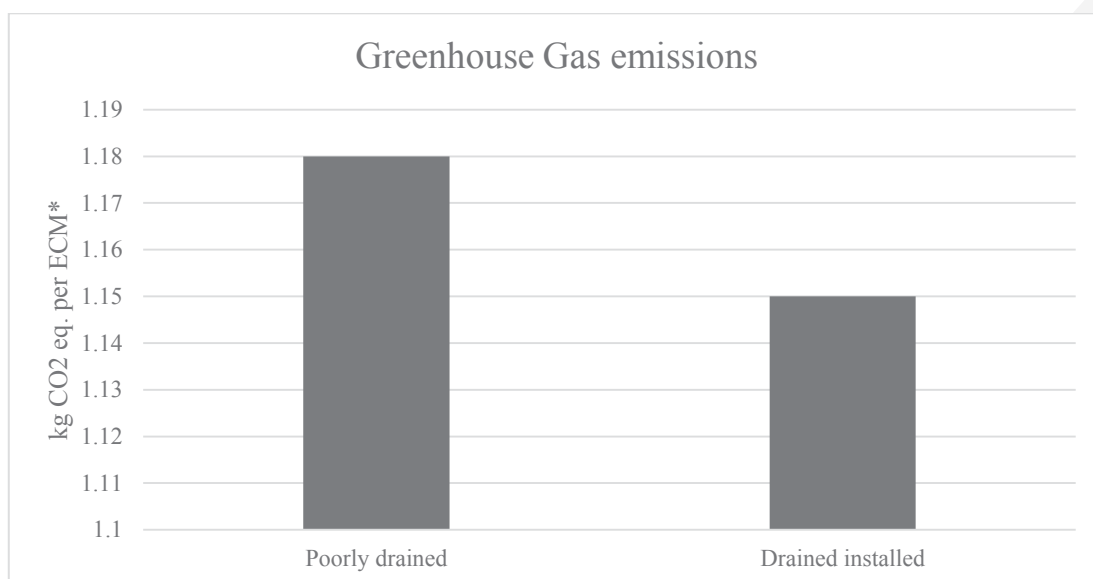


Figure 1: Greenhouse gas emissions of dairy farm on poorly-drained soil and with field drains installed

*ECM energy corrected milk

4. Discussion

The *Dairy_sim* simulation model predicted differences in management practices for the dairy unit following installation of field drains. The dairy unit with field drains could sustain a 20% increase in stocking rate and the housing days were reduced by 21%. The GHG emission values were in range

with Casey and Holden (2005b) and O'Brien *et al.* (2014). Due to difference in scope, assumptions and inventories the results cannot be compared directly with previous studies. The GHG emissions were less by 2.5% per kg ECM for an optimally managed farm following field drains installation. The decrease in emissions for drained farms were mainly due to the increase in productivity. Enteric methane was the major contributor to GHG emission for both situations, as found by Casey and Holden (2005a) and Yan *et al.* (2013). The other main sources of GHG emissions were N₂O from grassland and solid/liquid manure storage. The N₂O emissions from drained farms were higher as than with no drains by 4%.

The percentage of fat and protein assumed in the study can have effect on total emission. If the values of %Fat and %Protein (%F:4.31; %P:3.49; (O'Brien *et al.* 2012) increase the total ECM will increase thus reducing the total emissions. Regional emission factor for enteric and manure management methane were assumed instead of national average (108.81 kg CH₄ head⁻¹ yr⁻¹ and 20.53 kg CH₄ head⁻¹ yr⁻¹) from O'Mara (2006), would result higher methane emissions. Emission factor assumed to calculate indirect N₂O emissions result in lower N₂O emissions from artificial drainage, as the default IPCC value 0.015 is considered to be overestimation of losses (Sawamoto *et al.* 2005); (Reay *et al.* 2003). Similarly, the direct N₂O emissions would increase drastically, if the IPCC default value (8 kg N₂O–N ha⁻¹) is considered, these are now supplement by Hiraishi *et al.* (2014). However, animal live weight and stocking rates will not affect the soil physical properties and herbage production but higher stocking density grazing on poorly drained grassland will have greater poaching damage (Tuohy *et al.* 2014).

5. Conclusions

The result of simulating a dairy unit using *Dairy_sim* before and after drain installation indicated that management practices would change. LCA predicted greater GHG emissions for poorly drained soil compared with the situation once field drains were installed. It was concluded that field drainage can theoretically improve the productivity of optimally managed dairy farms on poorly-drained soil and this was reflected in lower GHG emissions per unit output. Further work is required to evaluate this result for different agroclimatic regions, target milk outputs and other environmental impacts.

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213. Spatial variation of secondary PM_{2.5} exposure and health impact from milk production

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ABSTRACT

Secondary PM_{2.5} human health impacts in life cycle assessment (LCA) are based on linear and simplified assumptions that may lead to a potential double counting. We investigate secondary PM_{2.5} intake fractions (iF) spatial variability due to milk production PM-related emissions in Wisconsin (WI), New Jersey (NJ), and New York (NY) using the Intervention Model for Air Pollution (InMAP). iFs are coupled with epidemiology-based dose-response to provide region-specific characterization factors (CF) for secondary PM_{2.5} and are then tested using a large dairy farm with 2436 animals, performing a sensitivity analysis with location and how impacts compare with nutritional benefits. Our findings suggest that there is substantial spatial variation of iF_{PM_{2.5}} in the U.S. linked to population density and atmospheric chemistry. Although WI emissions can result in PM_{2.5} with a travel distance of about 1500 km, the resulting magnitude of exposure is lower by about a factor of 10 compared to NJ emissions that have the highest exposure estimate with shorter PM_{2.5} travel distance. In regards on our case study, milk production in highly populated NH₃-limited regions, such as NJ, has substantially higher PM_{2.5}-related health impacts to populations downwind than production in agricultural regions where NH₃ is in abundance, such as WI. This research contributes to spatially-explicit CFs for the agricultural sector.

Keywords: agriculture, particulate matter, ammonia, intake fraction, milk

1. Introduction

Agriculture- and food-related processes are usually associated with high ammonia (NH₃) emissions that contribute to the formation of secondary inorganic PM_{2.5}, particles with an aerodynamic diameter of <2.5 μm that are formed in the atmosphere from through photochemical reactions and oxidation processes. Ambient particulate matter (PM_{2.5}) is an important environmental risk factor according to the global burden of disease (GBD) with a burden of about 3 million annual deaths globally (IHME, 2015).

Up to date, secondary inorganic PM_{2.5} human health impacts in life cycle assessment (LCA) have been treated based on linear (Hofstetter, 1998) and simplified (Van Zelm et al., 2008) assumptions related to exposure characterization that do not fully capture the complex relationship between precursors and secondary PM formation and may lead to a potential double counting of corresponding health impact estimates (Fantke et al., 2015). It has been found that precursor availability, and more specifically NH₃ ambient background concentrations, can substantially affect the magnitude of secondary PM_{2.5} exposure (Paulot and Jacob, 2014). The first aim of this study is to provide spatial intake fractions (iF) for secondary inorganic PM_{2.5} for the U.S. and to identify potential factors of influence. The second aim is to estimate PM-related health impacts that may result in from an increase in milk production so as to meet the Dietary Guidelines for Americans. The final aim of this study is to quantify and compare the overall environmental and nutritional effects linked to the addition of one serving milk to the average U.S consumption.

2. Methods

2.1. Intake fraction

Intake fraction (iF) is a metric that links environmental emissions to human exposure (Bennett et al., 2002). To estimate iF we use the following equation by Greco et al., (2007):

$$iF_i = \frac{\sum_i (P_i \Delta C_{ij}) BR}{Q_j} \quad (\text{Equation 1})$$

where P_i is the population in the region of impact, ΔC_{ij} is the change in ambient PM_{2.5} concentration in region of impact i measured in $\mu\text{g m}^{-3}$ due to precursor emissions (PM_{2.5}, NH₃, NO_x, SO₂)

indicating the , Q_j is the precursor emissions in the region of emissions j , and BR is the population breathing rate set at $13 \text{ m}^3\text{day}^{-1}$.

2.2. Emissions - Concentration Model

The (In)tervention (M)odel for (A)ir (P)ollution (InMAP) is a multi-scale emissions-to-health impact model that can operate as an alternative to comprehensive air quality models for marginal emission changes (Tessum et al., 2015). This model estimates primary and secondary $\text{PM}_{2.5}$ concentrations and corresponding health exposure and impacts that result in from annual changes in precursor emissions. It operates on annual-average input parameters using transport, deposition, and reaction rates estimates from the chemical transport model (WRF-Chem) within an Eulerian modeling framework. The model allows exposure-dependent resolution with higher resolution (1 km grids) for urban areas and a lower resolution (48 km grids) for remote areas. In addition the model allows for a low computational cost. Finally, although the model currently covers the greater region of North America (U.S., Southern Canada, Northern Mexico, etc.), it has the potential of being extended from a regional to a global scale.

2.3. Study locations

Spatial $\text{PM}_{2.5}$ concentration estimates are determined for three distinct locations in the U.S. in Wisconsin (WI), New Jersey (NJ), and New York (NY). The three locations have been selected as to reflect various population density and precursor limiting conditions (Table 1). The WI location represents a region with abundance in NH_3 but limited in NO_x and SO_2 and low population around the source. The NJ location represents a region with abundance in NO_x and SO_2 but limited in NH_3 and high population around the source. The NY locations has no dominantly limiting conditions and hence represents the point of reference.

Table 1. Population data and atmospheric conditions around source locations

	State	County	Pop _{50 km}	Pop _{500 km}	Atm. conditions			
			(Million)	(Million)	NH_3	SO_2	NO_x	$\text{PM}_{2.5}$
1	WI	Clark	0.02	28.1	Medium	Low	Low	Medium
2	NY	Onondaga	0.5	84.8	Medium	Medium	Medium	Medium
3	NJ	Union	3.2	68.4	Low	High	High	High

2.4. Characterization factors

Having the approach of Humbert et al, (2011) as a foundation, characterization factors (CF) for primary and secondary $\text{PM}_{2.5}$ will be calculated to reflect the human health impact in DALYs per unit of precursor emitted ($\text{PM}_{2.5}$, NH_3 , NO_x , SO_2). CF can be estimated using the equation (Humbert, et al., 2011):

$$CF_i = iF_i \times ERF \times SF \quad (\text{Equation 2})$$

where i represents one of the precursors $\text{PM}_{2.5}$, NH_3 , NO_x , SO_2 , CF_i indicates the human health impact per mass precursor i emitted (DALYs/kgemitted), iF_i is intake fractions of $\text{PM}_{2.5}$ from precursor i emissions that is inhaled by the exposed population, ERF is in deaths/ unit mass concentration and indicate the $\text{PM}_{2.5}$ exposure-response factor, and SF is in DALYs/death indicating the severity factor. For ERF and SF are utilize the work by Gronlund et al., (2015) with an effect factor ($EF=ERF \times SF$) estimate of $78 \text{ DALY}/\text{kg}_{\text{PM}_{2.5} \text{ inhaled}}$.

2.5. Case study: milk consumption in the U.S.

These CFs are then applied to a milk case study that investigates the potential environmental and nutritional effects associated with a one serving increase in milk consumption so as to meet dietary guidelines (USDHHS and USDA 2015). For a comprehensive assessment of this potential dietary change, we employ the Combined Nutritional and Environmental Life Cycle Assessment (CONELCA) framework that consistently evaluates and compares environmental and nutritional effects of foods or diets (Stylianou et al., 2016). For this case study, the environmental assessment, emissions associated with one milk serving are determined based on a U.S. specific milk LCA (Henderson et al., 2013) and are then linked to human health impact in regards to global warming and PM_{2.5} following a traditional LCA approach. For the nutritional assessment, impacts and benefits associated with one serving milk increase over the current average consumption, such as colorectal cancer, stroke, and prostate cancer are estimated based on epidemiological studies, starting with the GBD, as described in Stylianou et al., 2016. As a sensitivity study, we assume that milk production occurs in our three study locations.

3. Results

3.1. Intake fraction summary

Figure 1 illustrates the spatial variations of human exposure as iFs resulting from NH₃ emissions in NY (A), WI (B), and NJ (C). In all locations, highest estimates are found as expected around the source of emission. iFs in WI show the greatest dispersion of exposure that expands from the northern Midwest to the eastern coastal regions of the U.S with an exposure travel distance of more than 1500 km. However, iF absolute values due to the WI emissions are substantially lower by about 2 orders magnitude when compared to iF due to NJ emissions. In NJ, a considerably smaller region is affected (150 km) by emissions since particles are transported off-land, but iFs are substantially higher than the other two regions, with estimates ranging between the order of 10⁻⁶ and 10⁻⁷ kg_{in}/kg_{emitted} close to the source where there is a high population density.

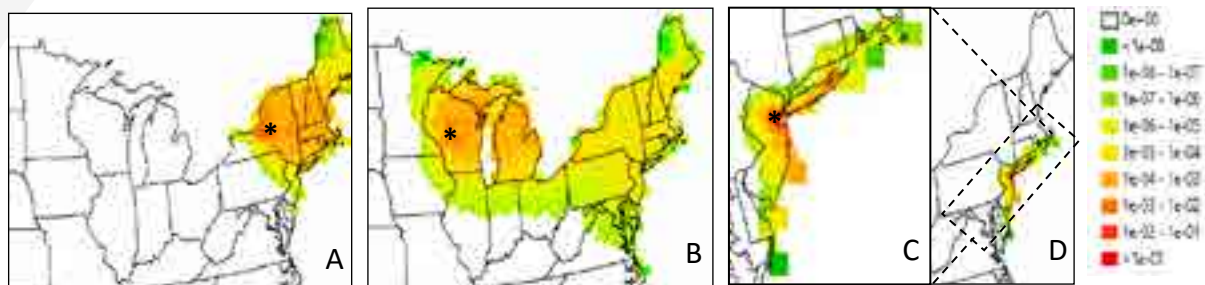


Figure 1. PM_{2.5} exposure estimates resulting from NH₃ emissions in NY (A), WI (B), and NJ (C & D) presented as iF_{PM_{2.5},NH₃} in parts per million (ppm) per grid cell. C is a close-up of D.

* indicates the location of the source of emission

Cumulative iFs are summarized in Figure 2 for each location of emission and precursor. There is a considerable spatial variation of exposure per precursor between locations, due to main factors of influence, i.e. population around the source and background precursor's concentrations (limiting conditions). NJ shows the highest iFs followed by WI and NY, reflecting population around sources. Of the total PM_{2.5} population exposure, nearly all (90%) happens on average within 20 km for NJ, 350 km for NY, and 1500 km for WI. Differences in atmospheric chemistry are also reflected by the fact that the iF_{PM_{2.5},NH₃} is 18 times higher than iF_{PM_{2.5},SO₂} in NJ (where NO_x and SO₂ are in abundance), but only 2 times higher in WI (where NH₃ is in abundance), suggesting that PM_{2.5} population exposures resulting from adding 1kg NH₃ are higher in NH₃-limited regions compared to NH₃-abundant regions.

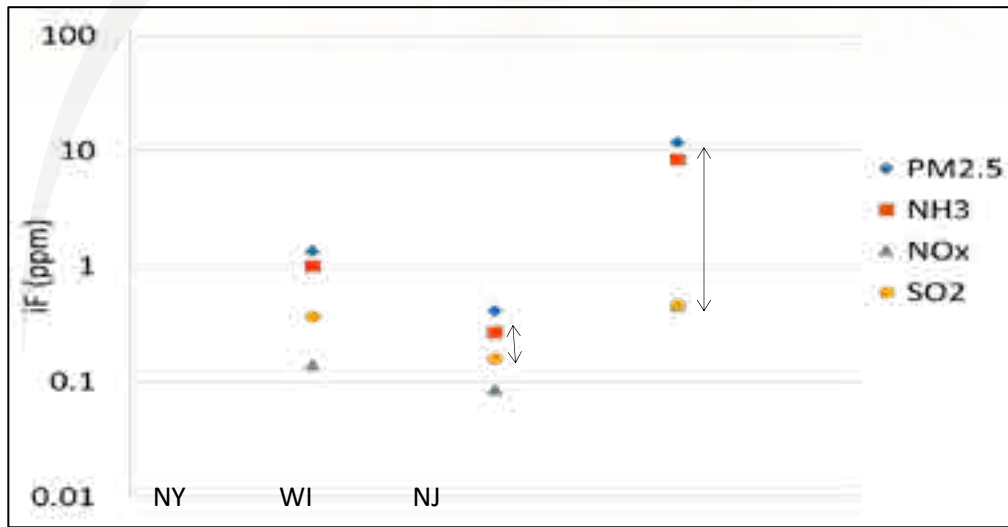


Figure 2. Primary and secondary PM_{2.5} Intake Fractions by precursor emissions in NY, WI, and NJ.

Combining the iF values reported here with an exposure-response (78 DALY/kg_{PM_{2.5} inhaled}) results in CFs (units: 10⁻⁵ DALY/kg precursor emitted) of 0.7–7.8 (NH₃), 0.6-3.5 (NO_x), 0.1-3.1 (PM_{2.5}), and 1.2-3.5 (SO₂).

3.2. Case study: Overall comparison

Figure 3 represents the overall comparison of environmental and nutritional effects associated with adding one serving of milk to the average U.S. milk consumption for the three locations of production. In regards to PM_{2.5}-related health impacts, about 63-73% of the impact is related to NH₃ emissions, with the highest contribution in NJ where milk production induces the highest total PM_{2.5}-related health impacts (9.1 μDALY/kg_{milk}). Overall, this dietary change leads to an overall health benefit due to nutritional benefits if production was to occur in NY or WI. However, if the corresponding production was to take place in NJ, PM-related health effects become substantial and comparable to nutritional benefits.

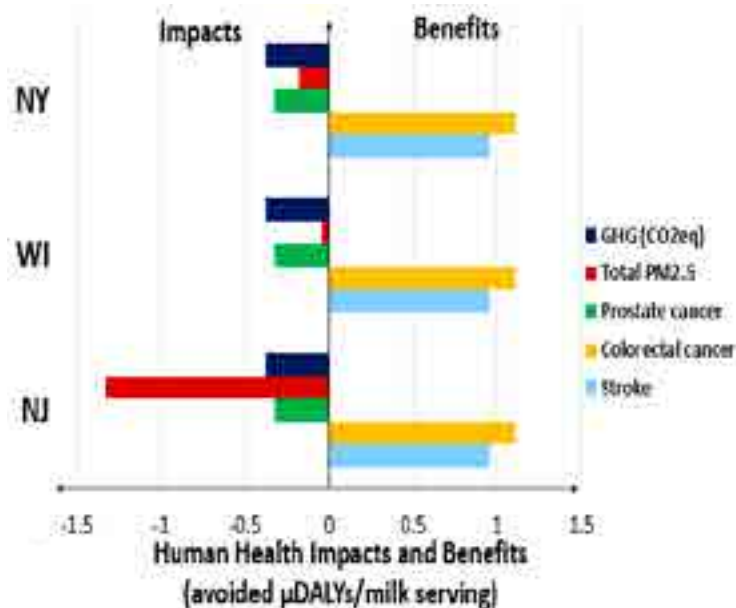


Figure 3. Comprehensive human health assessment for one milk serving increase per production locations.

4. Discussion

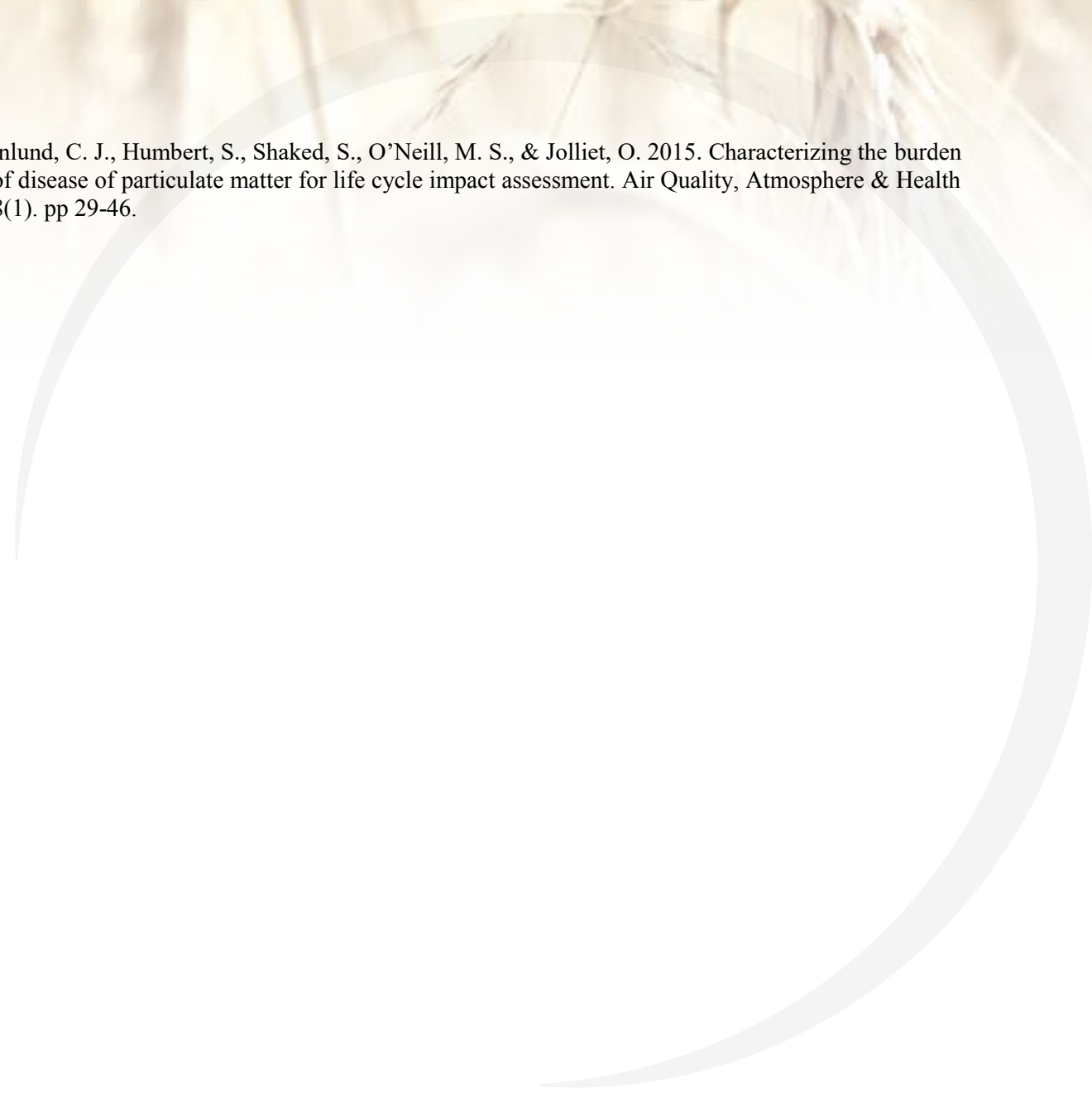
In this paper, we use the InMAP model to estimate spatially-explicit PM_{2.5} iF for PM-related precursors (PM_{2.5}, NH₃, NO_x, SO₂) in three distinct locations. This model allows us for resolution according to exposure and low computational intensity. Although we limited our analysis to three locations, WI, NY, NJ, our preliminary results support spatial variation of exposure that is linked primarily to population density. However, there are indication that atmospheric conditions, and in particular, NH₃ limiting conditions, influencing exposure estimates. Finally, our spatial estimates for PM_{2.5}, SO₂ and NO_x are in agreement with previous estimates in the literature (Humbert et al., 2011) while for NH₃ results suggest a higher maximum and wider range of estimates.

5. Conclusions

Our preliminary results support a spatial variation of secondary inorganic PM_{2.5} exposure in the U.S. linked to population density and atmospheric chemistry. Milk production in highly populated NH₃-limited regions has substantially higher PM_{2.5}-related health impacts to populations downwind than production in agricultural regions where NH₃ is in abundance. These findings is especially important for food-related decision makers since potential emission relocations might have considerable health effect to populations downwind. This research contributes to spatially-explicit CFs for the agricultural sector.

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73. The impact of subclinical ketosis in dairy cows on greenhouse gas emissions of milk production

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ABSTRACT

This study aimed to estimate the impact of subclinical ketosis (SCK) and related diseases in dairy cows on greenhouse gas (GHG) emissions of milk production. A dynamic stochastic Monte Carlo simulation model was developed and combined with life cycle assessment (LCA) to quantify the impact of SCK and related diseases on GHG emissions per ton fat-and-protein-corrected milk (kg CO₂e/t FPCM). The model simulates on cow level and the impact on GHG emissions was assessed from cradle-to-farm gate for the Dutch situation. Emissions of GHGs were increased on average by 18.4 kg CO₂e/t FPCM per case of SCK. Our study showed that LCA is a useful method to estimate the impact of diseases on GHG emissions and showed that reducing SCK and related diseases will reduce GHG emissions of milk production.

Keywords: disease, life cycle assessment, environment

1. Introduction

Subclinical ketosis (SCK) in dairy cows is a metabolic disorder that occurs around the calving period. In this period, the energy requirement of the cow can exceed her energy intake, resulting in a negative energy balance (NEB) (Grummer 1995). An NEB results in an increase of non-esterified fatty acids and beta-hydroxybutyrate levels in the blood. A cow has SCK when the beta-hydroxybutyrate level is higher than 1.2-1.4 mmol/l blood, but has no clinical signs (Raboisson et al. 2014). SCK results in reduced milk production and reproduction. Moreover, SCK increases the risk on related diseases, e.g. displaced abomasum, metritis, mastitis, lameness and clinical ketosis (Berge & Vertenten 2014, Raboisson et al. 2014). These related diseases also have an impact on milk production and reproduction and, in addition, may result in discarded milk and an increased risk on culling and dying. The prevalence of SCK in European dairy cows varies between 11 and 49% (Berge & Vertenten 2014, Suthar et al. 2013).

Diseases result in inefficient production, and, therefore, have an impact on the economic and environmental performance of dairy farming. Economic costs per case of SCK have been estimated at €257 (Raboisson et al. 2015) and \$289 (McArt et al., 2015). The impact of SCK on the environmental impact of dairy production, however, has not been analysed.

An important environmental problem is climate change. The livestock sector is responsible for about 14.5% of human induced greenhouse gas (GHG) emissions (Gerber et al. 2013), of which 30% is emitted by the dairy sector. With an expected increase in milk consumption of 58% in 2050 (FAO 2011, OECD/FAO 2013), reducing GHG emissions from the dairy sector becomes more important.

This study aims to combine the dynamics and consequences of diseases with life cycle assessment (LCA) to estimate the impact of SCK on GHG emissions of milk production. The method proposed can support decision making to improve health of dairy cows while reducing the environmental impact of food production.

2. Methods

A dynamic stochastic Monte Carlo simulation model was developed and combined with LCA to quantify the impact of SCK and related diseases on GHG emissions. The model simulates on cow level and the impact on GHG emissions was assessed from cradle-to-farm gate, based on parameters representing a Dutch dairy farm with day grazing. The model consists of four parts: parameters of the cow, dynamics of diseases, losses because of diseases, and estimation of GHG emissions. The model was developed in R (R_Core_Team 2013) and ran with 100,000 iterations.

Parameters of the cow

Each cow in the model received a parity (1-5+), based on an average herd composition. Based on the parity, a milk production, a body weight, and calving interval were attributed to each cow (CRV 2014, CVB 2012). The calving interval included a 60 days dry period. A lactation curve was utilised to estimate the daily milk production (Wood 1967). Subsequently, energy requirement for maintenance, growth, pregnancy and fat-and-protein-corrected milk (FPCM) were estimated and summed per cow per lactation (CVB 2012).

Dynamics of diseases

Diseases that occur in the first 30 days after calving were included. First, cows had a probability to get SCK, which was dependent on her parity. Second, cows with SCK had an additional probability to get one related disease: mastitis, metritis, displaced abomasum, lameness or clinical ketosis. The additional probability was the difference between the probability of a cow with and a cow without SCK on getting the disease (Berge & Vertenten 2014). Thus, a cow could have no disease, SCK only, or SCK and a related disease. Cows without SCK (and related diseases) were excluded from further analyses. All probabilities were based on Berge and Vertenten (2014). Total disease incidence among the 5 parities in the first 30 days was, 25.0% for SCK, 12.5% for mastitis, 10.2% for metritis, 4.0 % for displaced abomasum, 4.5% for lameness, and 1.6% for clinical ketosis (Berge & Vertenten 2014, Bruijnis et al. 2010, Raboisson et al. 2014). Third, cows with SCK and a related disease had a probability on culling and dying (Bar et al. 2008), together called removal. The events of SCK, a related disease or getting removed were determined with discrete distribution functions.

Losses because of diseases

Cows with SCK (and related diseases) had milk losses during one lactation because of reduced milk production, discarded milk, and removal of the cow. Milk production of cows with SCK (and related diseases) during one lactation increased because of an extended calving interval.

Cows with SCK (and related diseases) had a reduced milk production. The reduction of milk production (%/d) and duration of reduced milk production were disease specific. Cows with SCK only had a reduced milk production during the first 30 days, whereas cows with SCK and a related disease also had a reduced milk production after day 30 (Gröhn et al. 2003, McArt et al. 2012, Raizman et al. 2002, Seegers et al. 2003). Cows with SCK and a related disease, except for lameness, were always treated with antibiotics for a disease specific period and milk was discarded during treatment and the withdrawal period. The calving interval was extended for cows with SCK (and related diseases). The extension of calving interval (in days) was disease specific. A cow was removed at day 30 and replaced with a heifer with average production parameters.

Subsequently, net energy requirement and the FPCM per lactation of cows with SCK (and related diseases) were estimated.

Estimation of GHG emissions

Emissions of carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) were estimated for processes along the dairy production that were affected by the consequences of SCK, including feed production, enteric fermentation, and manure management. Emissions of GHGs were estimated for a cow with SCK (and related diseases) and without SCK for one lactation and were expressed as the sum of kg CO₂ equivalents (100 years' time horizon) (Myhre 2013) per ton of FPCM (kg CO₂e/t FPCM). The difference between a cow with SCK (and related diseases) and without SCK was the impact of SCK (and related diseases) on GHG emissions.

A diet for the summer and winter period was composed of concentrates and roughages (grass, grass silage, maize silage) (CBS 2014, Nevedi 2012, 2013, 2014, 2015). Roughages were produced on the farm and concentrates were purchased. Feed intake (kg DM/ cow) was estimated based on the weighted average energy content of the diet (MJ/ kg DM) and the energy requirements of the cow (MJ/ cow). First, the emissions related to feed production (kg CO₂e/ kg DM) were estimated (transport, crop cultivation, processing, feed mill) (Vellinga et al. 2013). Emissions from land use and land use change (LULUC) related to feed production were included based on Vellinga et al. (2013). Second, emissions related to enteric fermentation were estimated based on feed specific emission factors (kg CH₄/ kg DM) (Vellinga et al. 2013). Third, emissions from manure were estimated. Direct and indirect N₂O (i.e., N₂O derived from volatilization of ammonia (NH₃) and nitrogen oxides (NO_x) and from leaching of nitrate (NO₃)) and CH₄ emissions from manure in stables, storage, and on grass were included and based on national inventory reports (De Mol 2003, De Vries 2011, Oenema 2001, Schils 2006) and IPCC (2006). Finally, all emissions were summed and divided by the total amount of FPCM in tons.

System expansion was applied to account for the production of meat from removed cows. The method has been shown to be a solid method for the evaluation of environmental impacts related to changes in

milk production (Zehetmeier et al. 2012). Meat from dairy cows, except for cows that died, was assumed to replace meat from chicken, pork or beef. Emissions of GHGs related to the production of meat from chicken, pork or beef were estimated per kg edible product (Van Middelaar et al. 2014a) and weighted to an average emission factor based on the average consumption of pork, chicken and beef in OECD countries (OECD 2015).

Cows that were removed before the end of parity 5 were assumed to be removed to early, resulting in additional GHG emissions for raising extra heifers. First, we calculated the emissions of GHGs from raising a heifer based on system expansion, to determine how much of these emissions were related to milk production. Emission were estimated as the difference between emissions of raising a heifer and emissions of the meat to be replaced by meat from that heifer. Second, emissions related to raising were depreciated over the total amount of milk that was produced by the cow at the moment of removal. For cows that were removed at parity 2+, average milk production levels were assumed for all previous parities.

3. Results

The emissions of GHGs increased on average by 18.4 (median 8.5) kg CO₂e/t FPCM per case of SCK, but variation was large. The increase in emissions was lowest for cows that had SCK only (≤ 11.4 kg CO₂e/t FPCM) followed by cows with SCK and a related disease (16.8-55.5 kg CO₂e/t FPCM). Cows that were removed showed the highest increase in emissions per case of SCK (≥ 56.7 kg CO₂e/t FPCM) (Figure 1).

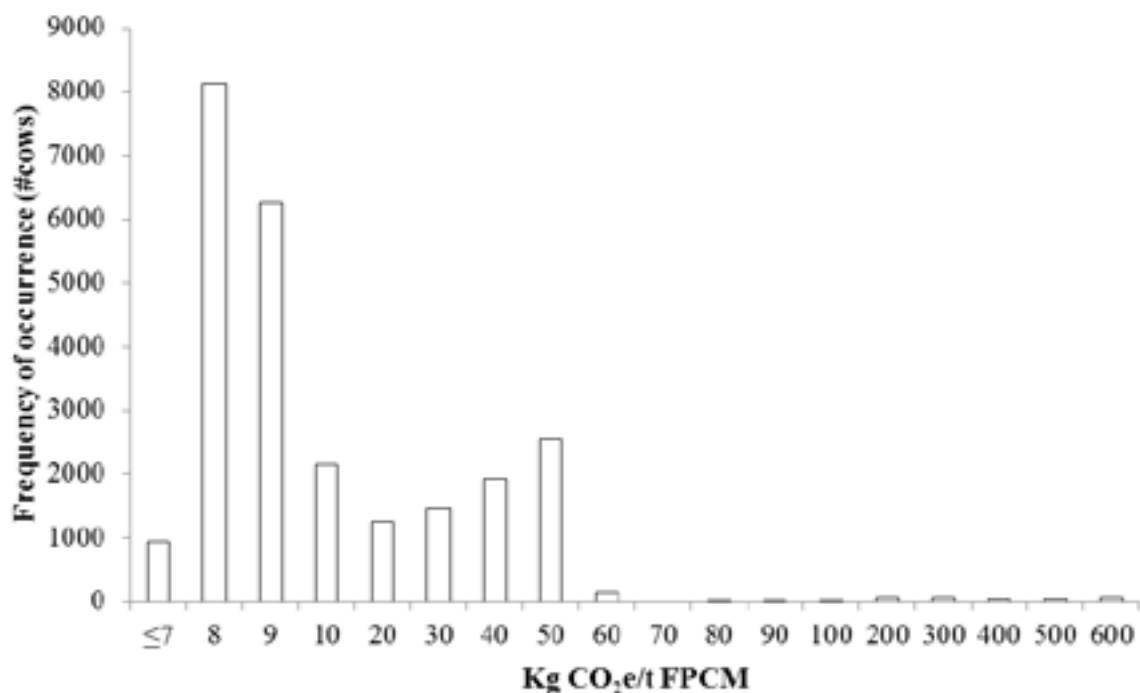


Figure 1. Extra kg carbon dioxide equivalents (CO₂e) per ton fat-and-protein-corrected milk (t FPCM) per case of subclinical ketosis (SCK), and frequency of occurrence. Average increase in emissions was ≤ 11.4 kg CO₂e/t FPCM for cows with SCK only, 16.8-55.5 kg CO₂e/t FPCM for cows with SCK and a related disease, and ≥ 56.7 kg CO₂e/t FPCM for cows that get culled or died.

4. Discussion

The average impact of SCK (and related diseases) on GHG emissions was 18.4 kg CO₂e/t FPCM per case, but showed large variation. Most cows with SCK (and related diseases), however, had SCK only, which had a lower increase in GHG emissions than the average cow with SCK (and related diseases). Cows with SCK and a related disease and cows that were removed had a higher increase in GHG emissions and had an important impact on the average increase of GHG emissions per case of SCK.

Cows with SCK only had a reduced milk production and an extended calving interval, whereas cows with SCK and a related disease had higher reduced milk production, a longer extended calving interval and also had discarded milk. Cows that were removed had a lower milk production and additional emissions for breeding replacement heifers, which were not needed if the dairy cows were not removed. Literature shows a huge variation in disease probabilities, relation of SCK with other diseases, removal probabilities, losses related to diseases, and GHG emissions of flows. Therefore, the impact of cows with SCK (and related diseases) on GHG emissions might differ between farms.

Most LCA studies use economic allocation (De Vries & de Boer 2010) to allocate emissions to the different outputs in case of a multiple output system. Estimating the economic allocation factor of milk based on one lactation requires additional assumptions about e.g. longevity of the cow. In this study, therefore, we applied system expansion. We assumed that milk production was the main purpose of dairy farming and that the meat of dairy cows substitute the production of beef, pork or chicken. Assumptions regarding the type of meat that is replaced can have an important impact on the result (Van Middelaar et al. 2014a). Therefore, our GHG calculations were based on the average consumption pattern of meat products in OECD countries. Future studies can include multiple types of meat products in order to show the potential range in results.

We estimated the impact of SCK (and related diseases) on GHG emissions for one lactation, because there was no data available for the incidence of diseases over multiple lactations. Estimating the impact of diseases during one lactation only, however, is a common method to estimate the economic costs of diseases (Bruijnijis et al. 2010, Huijps & Hogeveen 2007, Inchaisri et al. 2010).

Based on an average emission factor of 1,000 kg CO₂e/t FPCM (De Vries & de Boer 2010), a case of SCK increased GHG emissions on average by 1.8%. The impact of SCK on GHG emissions at herd level depends on the disease incidence of the herd. Complete eradication of SCK in a herd might not be achievable, but a minimum incidence of 10% of SCK at herd level might (Raboisson et al. 2015). Reducing the incidence of SCK from 25% to 10% at herd level, therefore, might have, on average, a minor impact on GHG emissions. On dairy farms with a higher incidence of SCK and related diseases and a higher removal rate of cows, however, reducing SCK and related diseases might have a higher impact on GHG emissions. In addition, our study only estimated the impact of SCK and the additional impact of related diseases in the first 30 days after calving. Including more diseases during the whole lactation might result in a higher impact of diseases on GHG emissions.

Examples of other mitigation options to reduce GHG emissions in the dairy sector are feeding and breeding strategies. Different feeding strategies showed on average a higher reduction in GHG emissions (9-32 kg CO₂e/t FPCM) (Van Middelaar et al. 2014b) than reducing SCK. These strategies, however, reduced the income of the farmer, whereas reducing SCK will increase the income of the farmer (McArt et al. 2015, Raboisson et al. 2015). In addition, farmers might prefer a reduction of SCK above a different feeding strategy, because this may be easier from a management perspective. Increasing the milk yield with 698 kg/year per cow and the longevity with 270 days per cow showed a reduction of 27 and 23 kg CO₂e/t FPCM (Van Middelaar et al. 2014a). Achieving this production by breeding, however, might take several years, whereas reducing SCK might not.

This study combined the dynamics and consequences of diseases with LCA to estimate the impact of SCK on GHG emissions of milk production. The method proposed can also be used to evaluate the impact of other diseases and can be extended to other environmental impact categories.

5. Conclusions

The average increase of GHG emissions per case of SCK was 18.4 kg CO₂e/t FPCM. The increase in emissions varied from ≤ 11.4 for cows that had SCK only, to ≥ 56.7 for cows that were removed. Our

study showed that LCA is a useful method to estimate the impact of diseases on GHG emissions and that reducing SCK and related diseases will reduce GHG emissions of milk production.

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215. Use of process-based models to quantify life cycle nutrient flows and greenhouse gas impacts of milk production: influence of beneficial management practices and climate change scenarios

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Assessing and improving the sustainability of dairy production systems is essential to secure future food production. This requires a holistic approach that reveals trade-offs between emissions of the different greenhouse gases (GHG) and nutrient-based pollutants and ensures that interactions between farm components are taken into account. Process-based models are essential to support whole-farm mass balance accounting, however, variation between process-based model results may be large and there is a need to compare and better understand the strengths and limitations of various models. Here, we use a whole-farm mass-balance approach to compare five process-based models in terms of major nutrient (N, P) flows and greenhouse gas (GHG) emissions associated with milk production at the animal, farm and field-scale (figure 1). Results of these models are then used as input for a farm Life Cycle assessment of a US farm of located in NY state with 2436 animals. Results show that predicted whole-farm, global warming impacts were very similar for the two whole farm models with a predicted global warming impact of approximately $1.1 \cdot 10^7$ kg CO₂eq./year for both models and a dominant contribution of enteric CH₄ emissions (figure 2). Model predictions were also highly comparable, i.e. within a factor of 1.5, for most nutrient flows related to the animal, barn and manure management system, including enteric CH₄ emissions, and NH₃ emissions from the barn. In contrast, predicted field emissions of N₂O and NH₃ to air, and N and P losses to the hydrosphere, were very variable across models. This indicates that there is a need to further our understanding of soil and crop nutrient flows and that measurement data on nutrient emissions are particularly needed for the field. A systematic analysis of Beneficial Management Practices (BMPs) and climate change scenarios will be presented, including comparison between feed rations scenarios (incl. high forage with Neutral Detergent Fiber Digestibility), between manure processing and storage systems (incl. digester vs lagoon), and crop & soil management practices (incl. all manure applied in Spring or in season, no till, all manure injected) and 15 climate change scenarios including RCP 2.5, 4.5, 6.0 and 8.5 (figure 3). The whole-farm mass-balance approach is advocated as an essential tool to assess and improve the sustainability of dairy production systems.

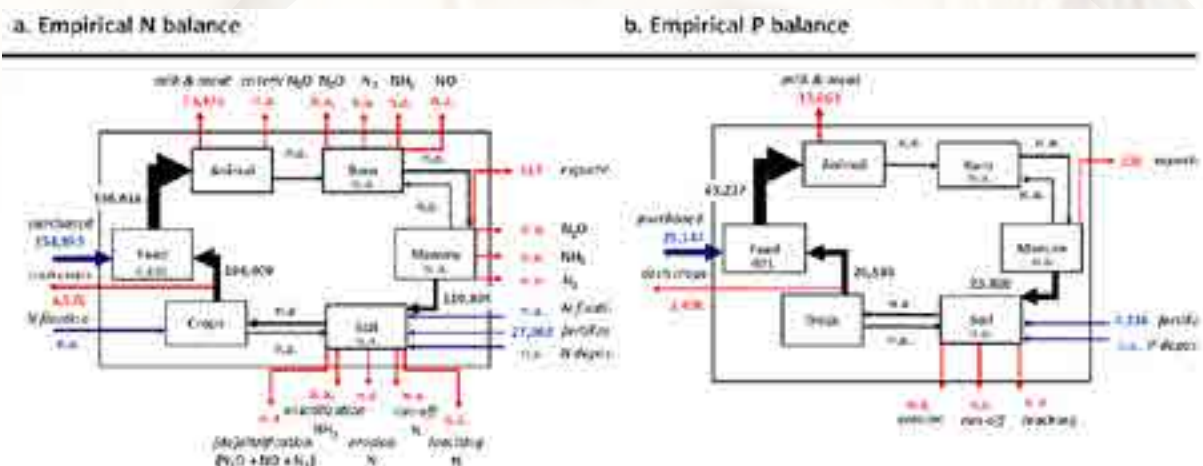


Figure 1. Whole-farm nutrient mass-balance: predictions from IFSM for N- and P-balance for the farm case study.

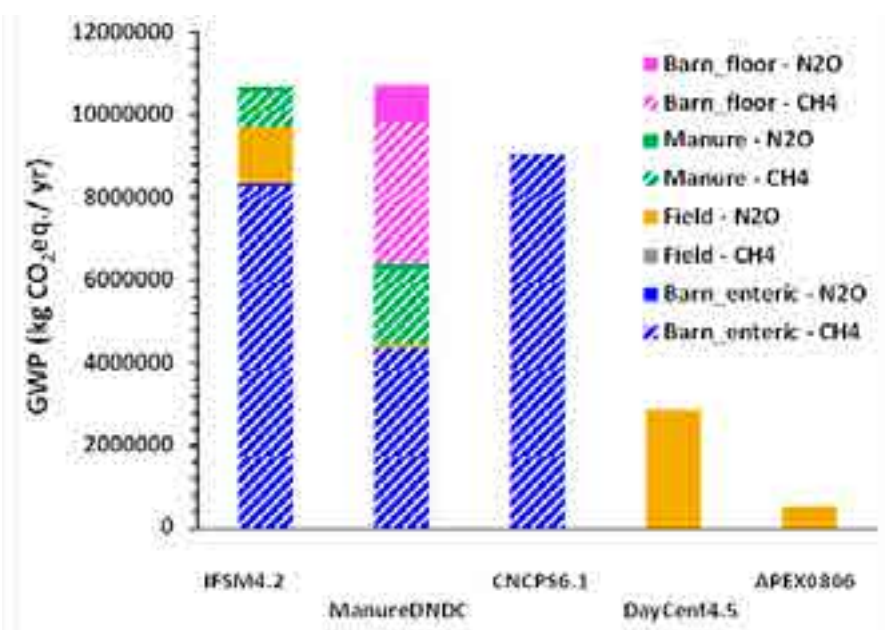


Figure 2. Predicted global warming potentials per farm component (in kg CO₂ eq./year)

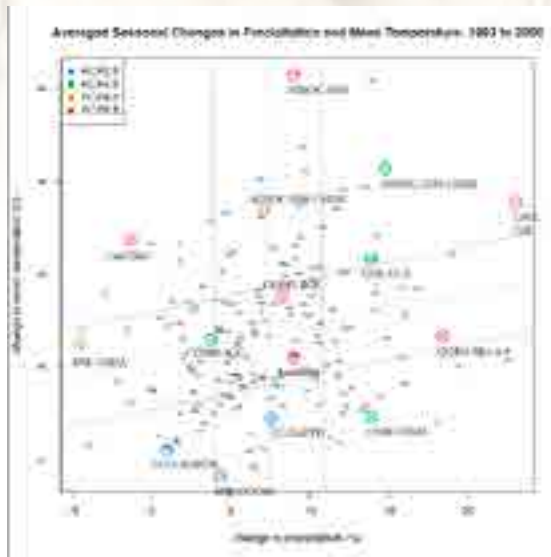


Figure 3. Change in mean temperature and in precipitation for various climate change scenarios

3. Crops and Fruits.

24. South African maize production: Mitigating environmental impacts through solar powered irrigation

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ABSTRACT

Agriculture is among the largest contributors to global greenhouse gas emissions. Clean technologies, such as renewable energies, have the potential to significantly reduce these environmental repercussions of agriculture. Countries like South Africa have a coal intensive electricity mix, as well as high solar irradiation and a dry climate which is why agricultural crops are produced under fossil energy intensive irrigation. At the same time, the high solar irradiation could be used for the generation of photovoltaic electricity as a renewable power supply for irrigation. A joint research project between the University of Cape Town and the Zurich University of Applied Sciences quantified the environmental impacts of South African maize production (*Zea mays*) and the improvement potential of maize irrigation with photovoltaic electricity by means of life cycle assessment (LCA).

The LCA includes the whole value chain of maize production from cultivation to storage in a silo for six months, respectively with a functional unit of one kilogram of maize at silo storage produced either on dry land or under irrigation. Electricity consumption for irrigation was identified as an environmental hotspot in the impacts related to greenhouse gas emissions from maize production. Therefore, clean electricity would be the starting point to reduce the carbon footprint of South Africa's maize. We calculated that replacement of South African electricity mix with photovoltaic electricity in the maize irrigation can reduce environmental impacts by up to 48 %. The calculated greenhouse gas emissions per kilogram of maize on dry land without irrigation, under irrigation and under irrigation using photovoltaic electricity, are 0.51 kg CO₂-eq. and 0.81 kg CO₂-eq. and 0.56 kg CO₂-eq., respectively, with a potential reduction of 31 % if the electricity is supplied from photovoltaics compared to the conventional fossil electricity mix. The analysis of further indicators reveals a reduction for freshwater ecotoxicity and human toxicity of carcinogenic substances. The irrigation of a maize field of one hectare consumes 1'900 kWh of electricity per year, which, in turn, requires a solar power plant with an area of 9 m². We computed that a total area of 199 ha of solar panels would suffice to produce the total electricity requirement of the current maize production area under irrigation. This corresponds to more than approximately 100'000 t CO₂-eq. saved per year.

Compared to data representing maize production in the United States and in Switzerland, South African maize production has a higher global warming potential per kilogram of maize due to lower yields in South Africa.

The replacement of the South African electricity mix in the irrigation with electricity from photovoltaics has proven to be an effective clean technology to reduce environmental impacts associated with maize production in South Africa. Compared to the irrigated field area, land use for PV panels is almost negligible and is therefore no limiting factor in the implementation of irrigation using photovoltaic electricity.

Keywords: Photovoltaic, greenhouse gas emissions, crop production, emerging economy

1. Introduction

Maize is the major feed grain and the most important staple food for the majority of South Africa's population. South African maize production is largely dependent on sufficient and timeous summer rainfall and can reach 12 million tonnes per year, planted in an area of over 2.6 million hectares (Grain SA, 2014). Maize is produced throughout South Africa. The provinces Free State, Mpumalanga and North West are the largest producers, accounting for approximately 83 % of the total production.

A distinction is made between maize production for feed and for human diet. White grain maize is primarily produced for human consumption and is on average 60 % of the total maize production in South Africa. Yellow maize is mostly used for animal feed and comprises about 40 % of the total South African maize production (DAFF, 2014).

In 2013 there was a shift towards higher maize production for animal feeding at the expense of maize production for human diet. The production volume of maize for feeding was 5'933'100 t spread over an area of 1'164'000 ha. The production of white grain maize for human diet is estimated to 5'580'300 t planted on 1'617'200 ha (Grain SA, 2014).

Agriculture is among the largest contributors to global anthropogenic non-CO₂ greenhouse gas emissions, accounting for 56 % of emissions in 2005 (IPCC, 2014). While animal production contributes most by methane emissions, arable production is associated with dinitrogen monoxide emissions (Johnson *et al.*, 2007).

In South Africa crop irrigation is typically operated with fossil fuel based energy. Using green and clean technologies, such as renewable energies, some of the environmental impacts of agriculture can be reduced.

In a joint research project between the University of Cape Town and the Zurich University of Applied Sciences the environmental impact, as well as the potential for improvement through the use of clean technologies, was assessed for South Africans maize production.

2. Goal and Scope

In order to define optimization strategies through the use of clean technologies, a Life Cycle Assessment (LCA) of the status quo of the grain maize production in South Africa for human diet was performed, following the ISO 14040 and 14044 standards (ISO, 2006a; 2006b). The system boundaries of the LCA include the whole value chain of grain maize production, from seed bed preparation, to maize cultivation and harvesting, to storage of the harvested maize in a concrete silo for approximately six months. The LCA considers the production and application of fertilizers and pesticides, as well as the particular use of irrigation water. Land use, direct field emissions, the production and use of tractors and agricultural machines and consequential diesel consumption of tractors in use as well as transport are also taken into account. The most relevant production data, including production area, application of fertilizers and pesticides, diesel consumption and yield, were taken from planning models provided by Grain SA, the national representing and consulting institution for grain producers in South Africa (Grain SA, 2014). The planning models, which give a detailed compilation of any costs in the maize production, cover the circumstances of grain maize production in three different regions of South Africa: Eastern Highveld, North West, and Central and Northern Free State. As a functional unit one kilogram of grain maize at silo storage produced either on dry land or under irrigation for human diet was defined.

South Africa has a high level of solar irradiation and a dry climate, which leads to an agricultural crop production under irrigation. Electricity supply for irrigation is a coal intensive electricity mix, leading to high environmental impacts, which are associated with irrigation. The goal of this study was to quantify the reduction potential by using solar irradiation for the generation of photovoltaic electricity as a renewable power supply for irrigation.

The environmental impacts were assessed using five different impact indicators, namely climate change according to IPCC (2013), the cumulative non-renewable energy demand according to Hischier *et al.* (2010), acidification and freshwater eutrophication according to the European Commission (2011), and human toxicity (cancer) according to Rosenbaum *et al.* (2008).

3. Life Cycle Inventory

In 2013, 82'000 ha with grain maize for human diet were irrigated in South Africa. Under irrigation yields are higher but also the use of fuel and need for fertilizers increase. An overview of the key inventory data for South African maize production for human diet on dry land and under irrigation is given in Table 1. Transport distances are estimated using online distance calculators for sea and land routes.

Field emissions to air as nitrous oxides (N₂O) and ammonia (NH₃) and leaching of nitrate to ground water (short and long term) are calculated according to Meier *et al.* (2012; 2014). Phosphate emissions to ground water through leaching and to surface waters through run-off and water erosion, as well as emissions of heavy metals to soil are modelled according to Nemecek *et al.* (2007). In addition, all pesticides applied for maize production were assumed to end up as emissions to the soil (Nemecek *et al.*, 2007). Background data for the life cycle inventories were taken from the international ecoinvent v3.2 database using the system model "allocation, recycled content" (ecoinvent Centre, 2015).

Seed, pesticides and fertilizers are transported by lorry from retailers to the farm (650 km) and harvested maize is transported from the field to the farm and further to a silo co-operation by tractor (5 km and 40 km, respectively). Tractor and agricultural machines are imported from the United States, Canada, Europe and Japan, whereas 1'700 km and 650 km are assumed representative for inland transportation by lorry in the export land and in South Africa, respectively. Overseas transport

is an average distance from the mentioned export countries above to South Africa, accounting for 14'400 km by a transoceanic ship. The life cycle inventories and the impact assessment were modelled with the SimaPro software v8.2 (PRé Consultants, 2016).

Table 1: Summary of life cycle inventory of grain maize for human diet produced on dry land and under irrigation in South Africa (ZA), representing maize production in 2013

	Unit	Grain maize, dry land	Grain maize, irrigated
Production area	ha	1'519'557	85'924
Yield	kg/ha	3'770	8'134
Seed	kg/ha	10.0	23.1
Fertilizers			
lime	t/ha	0.9	1.0
NPK	kg/ha	84.7	282
manure	t/ha	2.5	2.5
Pesticides			
herbicides	kg/ha	0.5	0.0001
insecticides, fungicides	L/ha	7.7	2.2
Diesel consumption	L/ha	71.9	79.7
Irrigation			
water	m ³ /ha	0	7'000
electricity	kWh/ha	0	1'900
Transports	km/t	281	179

South Africa, as an emerging economy, is used to irrigating its crops with fossil fuel based energy. Although only 10 % of the maize crop is produced under irrigation (DAFF, 2014), the potential to reduce environmental impacts by using green and clean technologies for irrigation is worth considering in more detail.

The main irrigation system in South Africa is a Centre Pivot system. In ecoinvent, inventory data for Centre Pivot irrigation systems are not available, therefore a new dataset was generated according the South African conditions concerning electricity supply and water use, based on personal communication with Jan Coetzee, extension officer at The South African Breweries.

A new inventory was also established for photovoltaic electricity. The dataset is based on a 570 kWp open ground installation with multi crystalline silicon panels. Annual yield is adapted to the main growing regions for maize (North West and Free State), and accounts for 1'770 kWh/kWp (European Commission, 2012). According to the IEA PVPS Methodology Guidelines, a life time of 30 years and an annual degradation of 0.7 % have been assumed (Fthenakis *et al.*, 2011).

4. Life Cycle Impact Assessment

Production of fertilizers, direct field emissions, diesel consumption and (if present) electricity consumption for irrigation were identified as environmental hotspots in the South African grain maize production (Figure 3). The global warming potential (GWP) of irrigated grain maize in South Africa amounts to 0.82 kg CO₂-eq. per kilogram of grain maize and is 39 % higher than the global warming potential of grain maize produced on dry land (0.50 kg CO₂-eq. per kilogram grain maize). The higher yields of irrigated maize cannot compensate for the additional electricity and diesel consumption for irrigation (Figure 3). If irrigation is supplied by the South African electricity mix, the contribution of irrigation to the overall GWP accounts for 36 %. The replacement of the South African electricity mix in the irrigation with electricity from photovoltaics results in a reduction of 0.27 kg CO₂-eq. per kilogram of grain maize, which is equivalent to 33 % (Figure 3). The GWP of irrigated grain maize using photovoltaic electricity is similar to grain maize production on dry land.

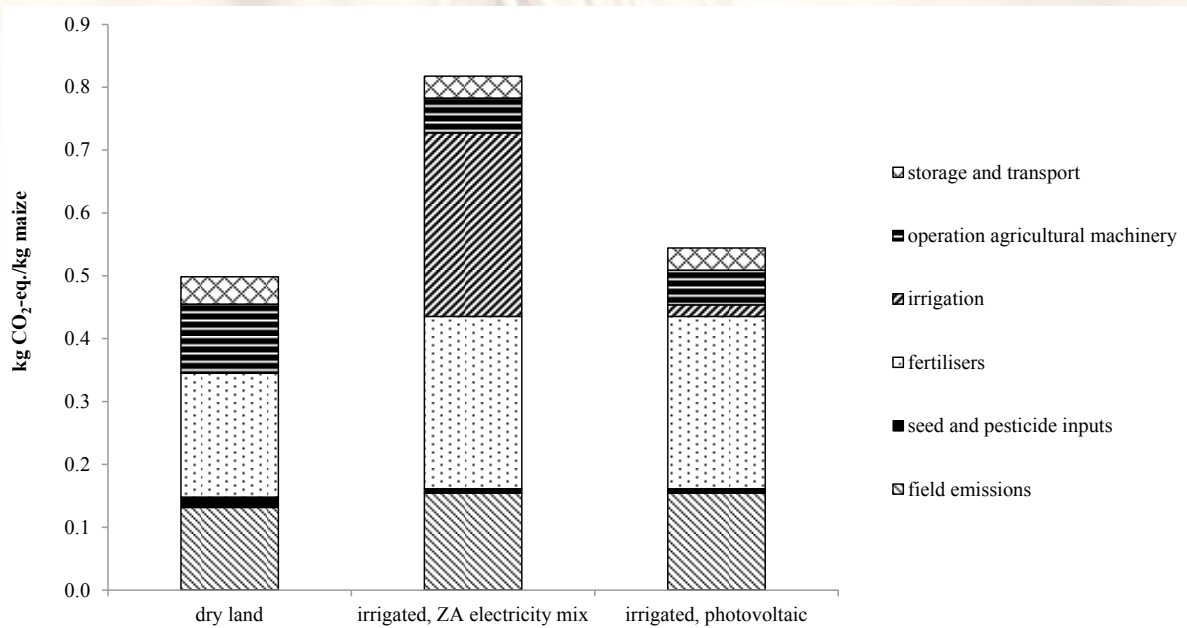


Figure 3: Global warming potential in kg CO₂-eq. of the production of 1 kg of grain maize on dry land (left-hand column) and under irrigation. Irrigation is either supplied by the South African electricity mix (central column) or by electricity from photovoltaics (right-hand column).

The analysis of further indicators reveals a significant reduction for non-renewable energy demand (fossil and nuclear) of 47%. Acidification, human toxicity of carcinogenic substances and freshwater eutrophication are reduced by 21%, 19% and 13%, respectively (Figure 4). However, not all environmental and human domains are equally affected. Freshwater ecotoxicity, marine eutrophication, land use and human toxicity of non-carcinogenic substances remain almost unaffected by the change of electricity supply.

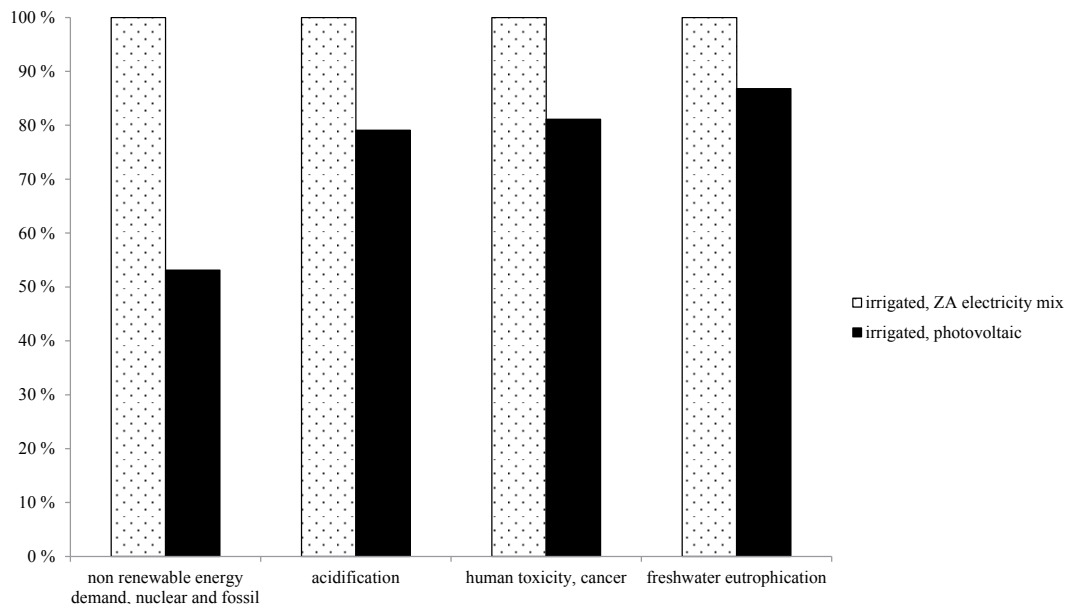


Figure 4: Comparison of selected environmental impacts for irrigated grain maize for human diet in South Africa using the South African electricity mix or electricity from photovoltaics for irrigation. All results are normalized by the results for maize irrigated using the national grid mix.

The high environmental impacts of the South African electricity mix are due to its composition: 88 % of the electricity is supplied by hard coal power plants and 5 % by nuclear power plants (ecoinvent Centre, 2015). By eliminating contributions of electricity with high environmental impacts, overall environmental impacts can be considerably reduced.

The irrigation of a maize field of one hectare consumes 1'900 kWh of electricity per year, which, in turn, requires a solar power plant with an area of 9 m². This means that in order to supply the power used for the irrigation of a field by means of photovoltaic panels, an area of only 0.09 % of the irrigated maize field is required.

5. Interpretation

We estimated that a total area of 76 ha of solar panels would be needed to produce the electricity to supply the current grain maize production area for human diet under irrigation (85'924 ha). This corresponds to about 190'900 t CO₂-eq. saved per year. Including the maize production under irrigation for feed, which covers a production area of 139'964 ha in South Africa, a total of more than 502'000 t CO₂-eq. could be saved per year (additional 311'200 t CO₂-eq. from feed). The required solar panel area to supply the total current maize production area under irrigation in South Africa, including maize for human diet and for feed, would increase up to 199 ha or 0.09 % of the irrigated maize area. The calculations about land use revealed that the installation area of PV panels is almost negligible compared to the irrigated production area. Consequently, land use is no limiting factor in the implementation of photovoltaics to irrigate the whole maize production throughout South Africa.

Compared to inventory data in ecoinvent, the modelled South African grain maize inventory has a higher global warming potential (0.82 kg CO₂-eq./kg maize) than maize produced in the United States (0.54 kg CO₂-eq./kg maize), in Switzerland (0.51 kg CO₂-eq./kg maize) or interpolated in global maize production (0.60 kg CO₂-eq./kg maize). System boundaries of the data inventories in ecoinvent are comparable to the maize inventory in the present study, including inputs of seeds, fertilizers, pesticides and irrigation water, as well as machine operations, field emissions and transport, and are therefore not crucial for the discrepancies regarding global warming potential. In contrast to our study, drying of grains at the farm is included, but not storage in a concrete silo. In South Africa, yields of 8'134 kg per hectare are lower than the yields of 9'315 kg per hectare gained in the United States, in Switzerland or in global maize production, leading to the higher greenhouse gas emissions per kilogram of maize, as mentioned above.

A further clean technology process, which is not yet widely used, is wireless sensor irrigation networks (WSIN). WSIN involve soil moisture sensors, specialized software interfaces and decision-supporting tools, which allows a more efficient and precise 'water on demand' irrigation. Water-saving technological processes are very important, especially where water is scarce and yield is highly dependent on proper irrigation, as is the case in South Africa. Majsztzik *et al.* (2013) show a decline in average water consumption of approximately 50 % compared to traditional irrigation in ornamental plant production in the USA. A reduction in fertilizer application, nutrient runoff and related greenhouse gas emissions can be attributed to the implementation of wireless sensor irrigation networks in horticulture. Further study is required to estimate the reduction potential through the implementation of WSIN in agronomic crops such as maize in open field production. By applying a combination of WSIN and renewable energy, the potential for mitigating environmental impacts could possibly be maximized.

6. Conclusion

The replacement of the South African electricity mix in irrigation with photovoltaic electricity has proven to be an effective clean technology to reduce environmental impacts associated with irrigated

maize production. As the calculations showed, land use is no limiting factor for installing PV panels in order to generate solar energy for the large scale irrigation of maize fields in South Africa.

Depending on the impact indicator, up to 47 % of the environmental impacts can be saved with irrigation supplied by photovoltaic electricity compared to energy supply by fossils. The environmental benefit would be even higher if renewable energy were expanded to further irrigated crops and additional clean technology processes like WSIN were implemented in South Africa.

Acknowledgement: This research project was carried out within the framework of the Swiss-South Africa Joint Research Programme and was funded by the Swiss National Science Foundation and the South African National Research Foundation.

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259. An evaluation of energy use within organic farming systems

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This review investigated the energy efficiency of a range of organic farming systems through a structured review of 35 LCA-based studies. Comparisons were made in relation to the amount of fossil energy required per unit of product and per unit of land. Energy output/input ratios were also compared for each product category.

Organic systems were found to use less fossil-fuel energy on a unit of land area basis for most crop and livestock products, although the energy use associated with imported compost could lead to higher inputs per hectare for some horticultural crops (Figure 1). Results were more variable per unit of product where lower yields and higher energy requirements for weed control could make some organic cropping systems perform worse (e.g. potatoes and tomatoes under glass, Figure 2). In addition, higher feed conversion ratios and mortality rates make some organic poultry systems less efficient per unit of output. For most grazing systems, organic farming resulted in lower energy use per unit area or weight of product. This results from using clover and other forage legumes in leys, which leads to more efficient forage production compared to the conventional Haber-Bosch based practice. Organic dairy production also tended to require less energy use per litre of milk produced, due to greater energy efficiency in the production of forage and reduced reliance on concentrates.

Lower levels of inputs were found to lead to higher output/input ratios in most organic systems, although with some exceptions (e.g. organic top fruit production and stockless arable systems, Figure 3). In many cases, organic farmers' diesel requirements were comparable to conventional.

Overall, the review found that organic farming systems have potential to contribute towards more energy efficient agriculture, but with lower yields. The review also highlighted that organic systems do not offer a radical alternative, as they are still reliant on fossil fuel sources and the differences in energy use per unit of product were often marginal. Organic methods could still be applied to increase the efficiency of the agriculture sector as a whole, although

energy use is only one aspect of sustainability and trade-offs may occur (e.g. between fossil energy and water use) for some products.

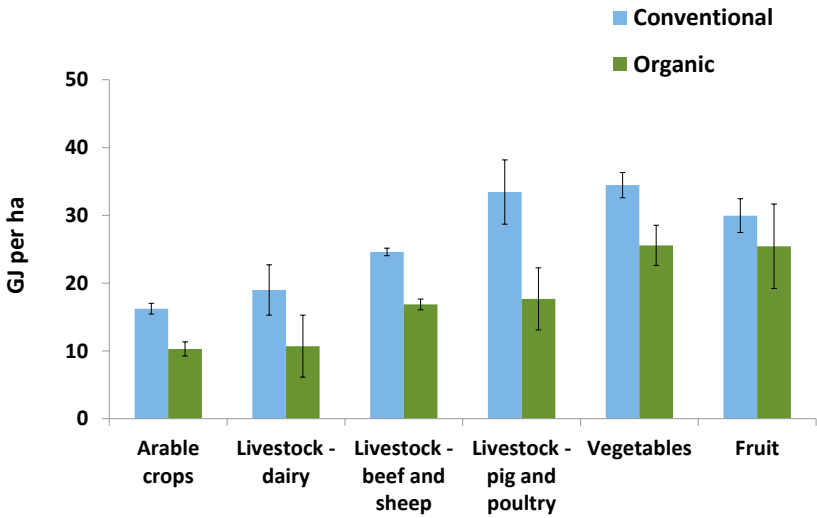


Figure 5: Average of all studies in gigajoules (GJ) per hectare with standard error

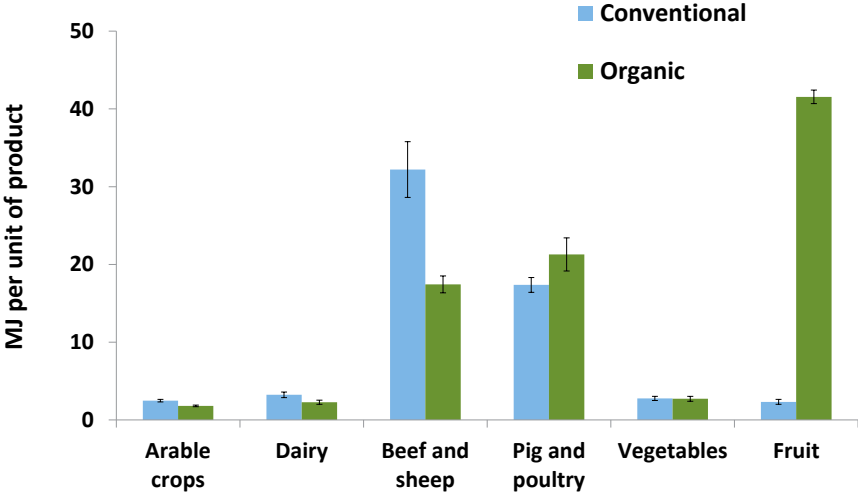


Figure 6: Average of all studies in megajoules (MJ) per unit of product with standard error

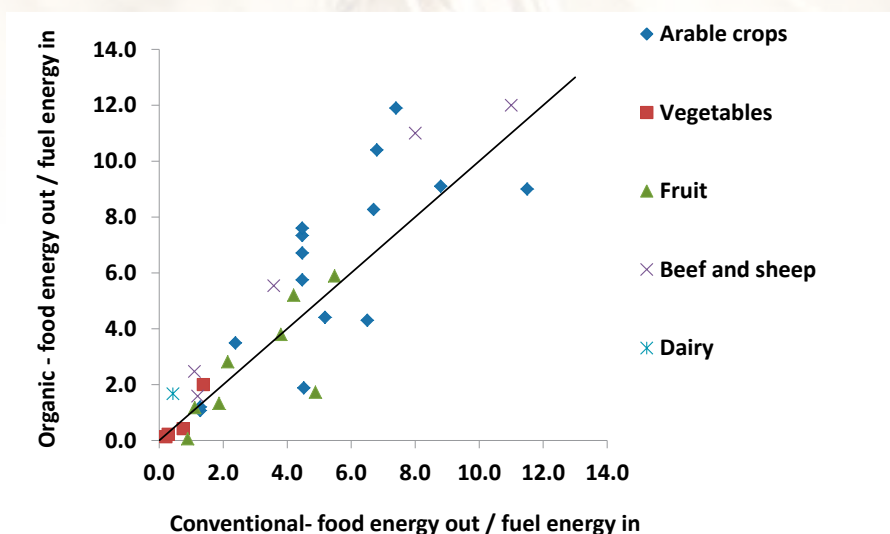


Figure 7: Average of energy ratios (energy output divided by input)

292.Life Cycle Assessment of Thai Organic Hom Mali Rice Farming to Support Policy Decision on Area Expansion

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ABSTRACT

Thailand has the strategic national policy to promote organic Hom Mali (Jasmine) rice farming in its quest to become the regional hub for organic agricultural products. The target for organic rice is set to be increased from a modest 0.18% to an ambitious 20% of the total rice production area in 2021. Different scenarios based on varying cultivation area ratios of organic rice farming in the North and North East regions of Thailand were proposed and their potential environmental impacts were evaluated by using life cycle assessment (LCA). The impact categories of interest included Climate change, Terrestrial acidification, Freshwater eutrophication, Terrestrial ecotoxicity, and Freshwater ecotoxicity and the impact assessment method applied was ReCiPe. In general, the rice yield in the North was higher than that in the North East. In terms of inputs, the fertilizer application rate and the amount of water use in the North were lower. The GHG emissions from the rice fields in the North East were higher ranging between 1.97 – 3.17 kgCO₂e/kg; those in the North ranged between 1.15 – 1.70 kgCO₂e/kg. It was revealed that the potential impacts on all the impact categories considered per kg of paddy rice were generally lower in the North. The expansion of organic rice farming in the North for 100% performed the best with the lowest impacts in all categories. The quantitative environmental performance of different areas could be useful for supporting the policy decisions on the area expansion.

Keywords: Life Cycle Assessment, Greenhouse gas emission, Hom Mali rice, Organic Rice Farming, Thailand

1. Introduction

Jasmine rice is a key product significantly contributing to socio-economic benefits for Thailand. At the same time, rice cultivation activities are highlighted as a major source of greenhouse gas (GHG) emissions from the agricultural sector, which ranks after energy production and industry operation, respectively. Water consumption for rice cultivation, especially for “Hom Mali” (Jasmine) rice, is also raised as a concern as it requires a wetland system maintained at least one month during a production cycle mostly through irrigation or rainfall. The potential impact on biodiversity losses are additional issues of concern for environmental sustainability of rice production systems.

Organic farming is seen as an alternative system for more sustainable rice production due to lower risks from chemical use, increasing biodiversity, lower production costs, and higher price. At present, the proportion of organic rice is only 0.18% (19,994 ha) of the total area of rice production at the national level. However, it is targeted to be increased to 20% in 2021 as stated in the national strategic plan of organic agricultural production with the aim to become the regional hub for organic agricultural products (MOAC, 2016). At this stage, it is not clear which areas should be promoted for increased organic rice farming for achieving maximum environmental benefits.

2. Methods

Life cycle assessment (LCA) was conducted to evaluate the environmental performances of different potential paddy farming areas to provide supporting information for policy decisions. The life cycle inventory data of organic rice farming were collected by interviewing 184 farmers covering about 4% (887 ha) of the total production in the North and 208 farmers covering about 1% (290 ha) of the total production in the North East of Thailand. The direct greenhouse gas (GHG) emissions from rice fields were obtained from the literature based on the field measurement and supplemented by the default values as defined in the national Product Category Rules (PCR) of rice products when necessary (TGO, 2013).

Different scenarios based on varying cultivation area ratios of organic rice farming were proposed in the North and North East regions, and their potential environmental impacts were evaluated. The impact categories of interest included Climate change, Terrestrial acidification, Freshwater eutrophication, Terrestrial ecotoxicity, and Freshwater ecotoxicity. The impact assessment method used was ReCiPe (Goedkoop et al., 2008).

3. Results

Table 1 shows the average gate-to-gate inventory data of organic rice farming over the crop period of 120 days. It was found that the productivity in the North was higher than that in the North East. A higher fertilizer application rate was being used in the North East due to the lower fertility of soil. Most of the rice farms in the North were irrigated, while those in the North East were mainly rain-fed. The water use was higher in the Northeast, due to the climate conditions that are rather dry.

Table 1: The average gate-to-gate inventory data of organic rice farming (the crop period was 120 days)

Region	Inputs		Outputs	
	Item (unit)	Quantity	Item (unit)	Quantity
North	land (ha)	0.16	paddy rice (kg)	525
	seed (kg)	43	GHG emission	27.65
	water (m ³)	3,928	(kgCH ₄)	
	diesel (L)	42.8	GHG emission	0.0001
	manure (kg)	147.85	(kgN ₂ O)	
	organic fertilizer (pellet) (kg)	56.23	water emissions	
	compost (kg)	46.62	- nitrogen (kg)	0.0048
	organic fertilizer liquid (L)	9.56	- phosphorus (kg)	0.0001
North East		0.16		
	land (ha)	43	paddy rice (kg)	482
	seed (kg)	4,550	GHG emission	42.26
	water (m ³)	42.8	(kgCH ₄)	
	diesel (L)	96.08	GHG emission	0.0001
	manure (kg)	13.06	(kgN ₂ O)	
	organic fertilizer (pellet) (kg)	142.23	water emissions	
	compost (kg)	5.81	- nitrogen (kg)	0.0044
organic fertilizer liquid (L)		- phosphorus (kg)	0.0001	

4. Discussion

Table 2 indicates that the life cycle environmental impacts per kg of paddy rice cultivated in the North were generally lower in all impact categories than that of the North East. The life cycle impact assessment (LCIA) results indicated that Sukhothai, Chiang Mai, Chiang Rai, Lampang and Phrae provinces in the North offered the lowest impact on climate change while the other impacts were not significantly different. In the same manner, Amnat Chareon, Yasothon, Mukdahan, Beng Kan, and Udon Thaini provinces in the North East performed the best with the lowest impact on climate change. Table 3 shows the results from the LCIA associated with different scenarios. It was revealed that the expansion of organic rice farming 100% in the North performed better, with the lowest impacts in all categories considered.

Table 2: Life cycle environmental impacts per kg of paddy rice cultivated in the North and North East

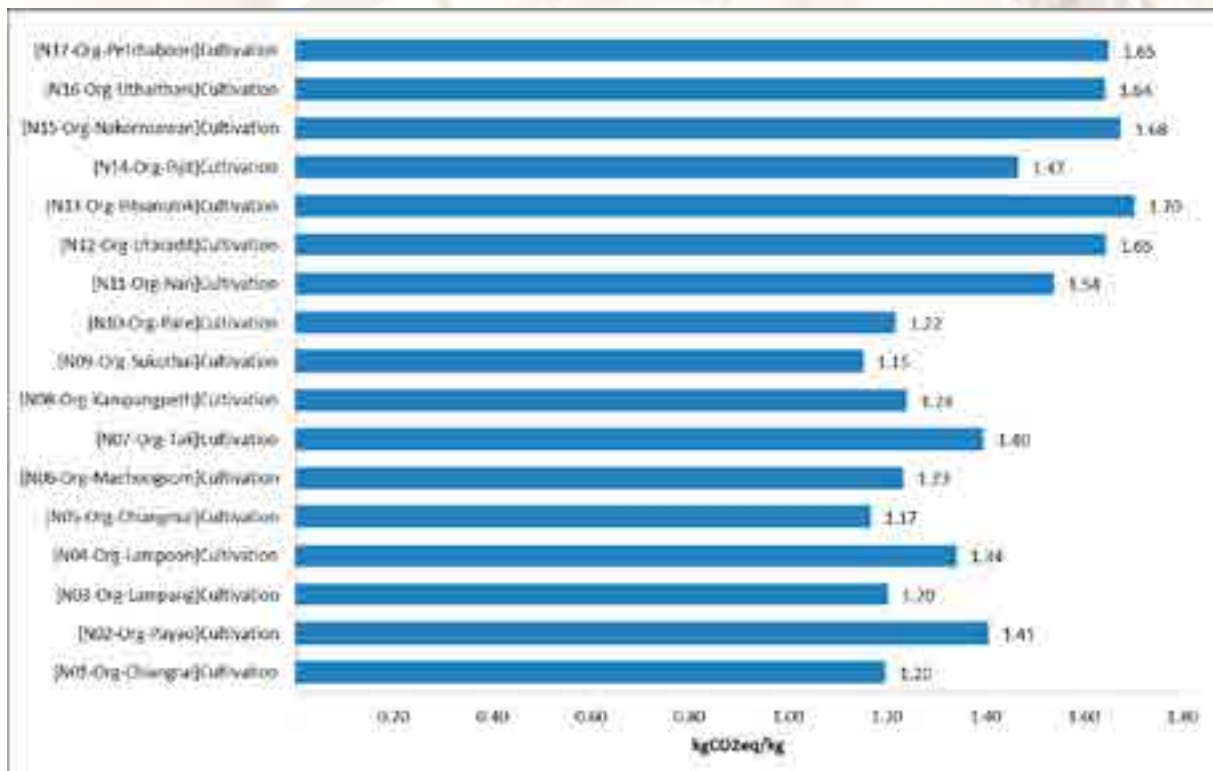
Impact category	Unit	North East	North
Climate change	kg CO ₂ eq	1.52E+00	1.09E+00
Terrestrial acidification	kg SO ₂ eq	2.71E-02	2.36E-02
Freshwater eutrophication	kg P eq	8.78E-06	7.50E-06
Terrestrial ecotoxicity	kg 1,4-DB eq	4.89E-06	4.16E-06
Freshwater ecotoxicity	kg 1,4-DB eq	1.12E-04	9.56E-05

Table 3: Results from the life cycle impact assessment

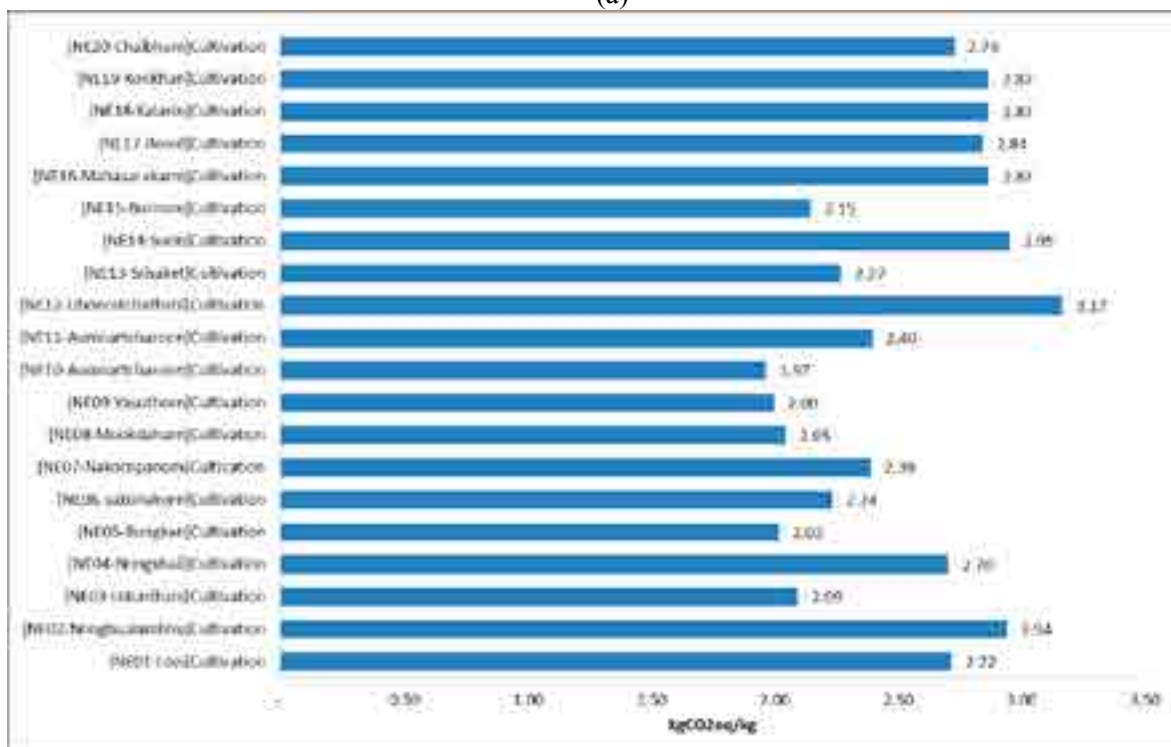
Scenario	Climate change (kg CO ₂ eq)	Terrestrial acidification (kg SO ₂ eq)	Freshwater eutrophication (kg P eq)	Terrestrial ecotoxicity kg (1,4-DB eq)	Freshwater ecotoxicity kg (1,4-DB eq)
N100%	46,116,390	998,345	317	176	4,046
N90%+NE10%	47,403,985	1,003,867	320	177	4,079
N80%+NE20%	48,691,581	1,009,389	322	179	4,111
N70%+NE30%	49,979,176	1,014,911	325	180	4,143
N60%+NE40%	51,266,771	1,020,434	327	182	4,176
N50%+NE50%	52,554,366	1,025,956	329	183	4,208
N40%+NE60%	53,841,961	1,031,478	332	185	4,241
N30%+NE70%	55,129,557	1,037,001	334	186	4,273
N20%+NE80%	56,417,152	1,042,523	337	187	4,305
N10%+NE90%	57,704,747	1,048,045	339	189	4,338
NE100%	58,992,342	1,053,567	341	190	4,370

Note: N – North, NE, North East (percentages indicate the areas planted in the North and North East)

The GHG emissions from rice fields in the North ranged between 1.15 – 1.70 kgCO₂e/kg whereas that in the North East between 1.97 – 3.17 kgCO₂e/kg (Figure 1). Sukhothai and Chiang Mai provinces performed the best in terms of climate change, due to lower fertilizer application rate and higher yield. In the North East, the lowest impact on climate change was found in Amnat Chareon, followed by Yasothon. It is worth noting here that the GHG emissions in Surin, Ubon Ratchathani, Khon Kaen, Phrae, and Chiang Mai provinces were based on the field measurements informed in technical reports (NSTDA, 2014). It was observed that the field measurements resulted in higher values of GHG emissions compared to the default values (using the IPCC tier-1 method).



(a)



(b)

Figure 1: GHG emissions per kg of rice produced in the North (a) and North East (b)

5. Conclusions

To make policy decisions on which areas should be expanded for the organic Hom Mali rice, the environmental performances of rice farming in different provinces should be taken into account. LCA offers quantitative environmental performances that could be useful for comparing the environmental performances among different areas. It was revealed that the expansion of organic rice farming 100% in the North was associated with the lowest impacts.

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89. Assessing methods to attribute soil greenhouse gas emissions to a crop in life cycle assessment of cropping systems

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ABSTRACT

There is an increased demand for a reduction in greenhouse gas (GHG) emissions and LCA has been widely applied to agricultural systems to estimate their Global Warming Potential (GWP). However, no consensus has been found on how to attribute soil GHG emissions in agricultural LCA. In this study, the objectives were: (i) to compare methods of attribution (year period, planting to planting, harvest to harvest), and (ii) to assess advantages and disadvantages of each method when used to attribute CO₂ and N₂O emissions to a crop in the LCA of cropping system with no winter crops. Soil CO₂ and N₂O emissions were estimated over 28 years using the biogeochemical DNDC model for 4 different scenarios based on a field experiment in Manitoba, Canada. Model results were used in the Crop.LCA tool to estimate global warming potential (GWP) on ha basis. Results showed no significant differences among methods when considering the full cropping system, however large differences were found on a year basis. Inter-annual variability was found to be higher than the difference across methods. Larger differences among methods were found for the cropping system where residues remained on the field. Thus a multimethod approach is suggested together with a long term LCA assessment to assess this system. The choice of methods to employ is a compromise between accuracy and applicability with regards to the LCA objectives.

Keywords: LCA, temporal variability, soil GHG emissions, cropping systems, DNDC

1. Introduction

There is an increasing demand for a reduction of greenhouse gas (GHG) to face the climate change challenge (Ingrao et al., 2016). Agriculture is responsible for 10-14% of the direct anthropogenic GHG emissions and an additional 12%–17% is due to land cover change, including deforestation (Paustian et al., 2016). Greenhouse gas emissions associated with agricultural activities can be highly variable because they depend on weather and climatic conditions, soil type, and agricultural practices (Miller et al., 2006).

Soil CO₂ emissions are caused by the decomposition of plant residues, mineralization of soil organic matter, decomposition of lime and urea hydrolysis. Different factors affect soil CO₂ emissions

including soil temperature, soil moisture, type of residues and tillage (Brady and Weil, 2002; Paustian et al., 2016). Improved crop management often leads to an increase in soil C content which can largely contribute towards the reduction of atmospheric CO₂ concentration (Paustian et al., 2016). Management changes will tend to result in an initial rapid rate of soil C change slowly decreasing in time until a new equilibrium is reached, unless further management changes occur. After reaching this equilibrium, soil C content is affected mainly by climate variability, without any clear trend in net soil C change (Hutchinson et al., 2007; Paustian et al., 2016). This equilibrium can take a long time (e.g., 20-100 years) to reach depending on soil and climatic conditions (Goglio et al., 2015; Petersen et al., 2013; Smith et al., 2010).

In contrast to soil CO₂ emissions, soil N₂O emissions occur mostly through denitrification and nitrification, which have a shorter time frame (i.e., days, weeks) (Goglio et al., 2013; Lehuger et al., 2007). The applications of nitrogen fertiliser and animal manure, soil tillage and crop type can all affect N₂O emissions. However, the effects of these crop management practices are largely dependent on soil type and weather conditions (Saggar, 2010). Along with soil C, N₂O emissions have a large spatial and temporal variability (Goglio et al., 2013; Uzoma et al., 2015). Often the emissions remain low throughout the year with the exception of peaks where nearly 90% of N₂O emissions occur (Abdalla et al., 2009). These peaks generally occur after rainfall events, fertiliser and manure applications, during thawing and the decomposition of high N content residues (Goglio et al., 2013; Saggar, 2010; Wagner-Riddle et al., 2007). Several models have been used to assess soil N₂O and CO₂ emissions (Goglio et al., 2013; Kim et al., 2009; Zaher et al., 2013).

Life cycle assessment (LCA) is undertaken to account for all GHG emissions (e.g. CO₂ and N₂O) and the environmental impacts of agricultural systems. In cropping systems, it has been observed that the impact of a crop is significantly affected by the previous crop (Brankatschk and Finkbeiner, 2015; Knudsen et al., 2014; Nemecek et al., 2015). Several studies have proposed different approaches to account for N₂O and CO₂ emissions (Goglio et al. 2015; Topp et al. 2011), including their temporal variability. However, no consensus has been found on how to attribute the timing of soil GHG emissions in agricultural LCA.

In this study, the objectives were: (i) to compare methods of attribution (calendar year, planting to planting, harvest to harvest), (ii) and to assess the advantages and disadvantages of each method when used to attribute CO₂ and N₂O emissions to a crop in the LCA of a cropping system with no winter crops.

2. Methods

The three methods of attribution considered were: the calendar year (Y), the period from planting to planting (PP), and the period from harvest to harvest (HH) (Figure 1). These methods were applied to 4 different cropping system scenarios. The Crop.LCA tool (Goglio et al., in preparation) was used to carry out a scenario assessment for different cropping systems on the basis of a field experiment described by Glenn et al. (2010, 2011, 2012) and Maas et al. (2013), located at the Glenlea Research Station (49.64°N, 97.16°W; height, 235 m, <2% slope) close to Winnipeg, Manitoba, Canada. The SOC content (0–0.2 m) was about 3.2% at the start of the study. The particle size distribution was 60% clay, 35% silt, and 5% sand.

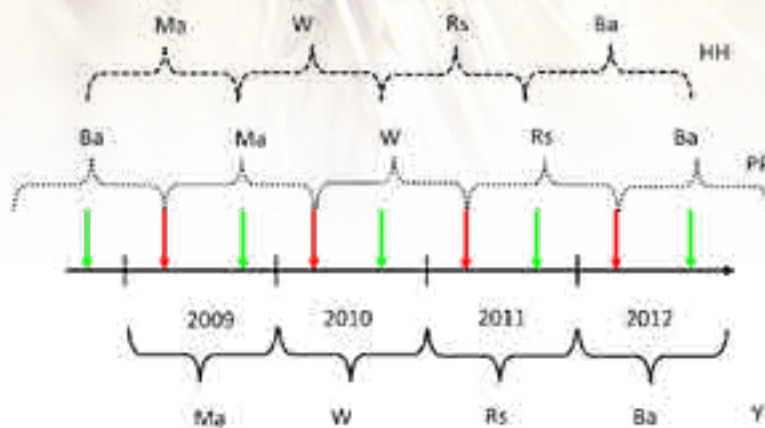


Figure 8 Description of the different methods to attribute soil GHG emissions to a crop (red arrows: planting date; green harrows: harvest date; Rs: rapeseed; Ba: barley; Ma: maize; W:wheat)

Crop management for the conventional (CONV) cropping system (Table 1) was similar to the annual cropping system described by Glenn et al., (2010) and Uzoma et al., (2015). The CONV, NT, and RES systems employed the same crop sequence, while in the legume (L) system, faba bean (*Vicia faba* var. minor L.) was substituted for maize (*Zea mays* L.) (Table 1). In the NT system, no tillage was carried out. Finally, in the RES cropping system, both straw and corn stover were left in the field (Table 1).

The scenario assessment used 28 years of climate data (1985–2012) to drive the estimates of the DNDC (Denitrification and Decomposition) model using site management and soils data from which the model was previously validated for estimating N₂O emissions (Uzoma et al., 2015). DNDC results for GHG emissions, grain and residue yield, and ammonia volatilization were then used as input to the Crop.LCA tool to perform the LCA (Goglio et al., in preparation) with 1 ha of land as the functional unit, in agreement with previous research (Goglio et al., 2014; Nemecek et al., 2011, 2015). One impact category was considered: 100 year horizon global warming potential (GWP), using the IPCC factors from the 5th assessment report (Myhre et al., 2013).

The system included the agricultural phase, all the upstream processes and farm transport (i.e. machinery production, transport, maintenance and repairs; production and delivery of fertilizers, pesticides, seeds and fuels and fuel consumption).

Data were collected from the field experiment, integrated with statistical data, expert opinion, databases, a survey of machinery manufacture, and products suppliers as in Goglio et al. (2014). For the HH method, it was assumed that both baling and collecting were carried out at harvest time.

GWP results on a ha basis were statistically assessed for the Y, PP and HH methods for each cropping system scenario separately, using R software (R Development Core Team, 2005). Each sample series was tested for normality. If the normality conditions were not met, the Friedman test was carried out together with paired nonparametric comparisons (Galili, 2013; Rosner, 2011). The standard deviation was computed for each cropping system x method combination; while the average difference between the various methods was computed for each cropping system.

Table 2 Summary of the average characteristics of the cropping systems considered in the scenario assessment (Note: Bold indicates differences between systems)

Cropping System		CONV	NT	L	RES
Crop sequence ^a		maize- spring wheat- rapeseed-spring barley	Maize- spring wheat- rapeseed-spring barley	faba bean^b- spring wheat- rapeseed- spring barley	maize- spring wheat- rapeseed- spring barley
Tillage		Yes	No tillage	Yes	Yes
Fertiliser	N (kg ha ⁻¹ y ⁻¹)	112	112	72.2	112
	P (kg ha ⁻¹ y ⁻¹)	16.6	16.6	6.6	16.6

K (kg ha ⁻¹ y ⁻¹)	18	18	0	18
S (kg ha ⁻¹ y ⁻¹)	18	18	0	18
Pesticide treatment number per year	1.5	2.5	1	1.5
Residue management	Straw and stover collected	Straw & stover collected	Straw and stover collected	Straw and stover left on the field

^amaize (*Zea mays* L.), spring wheat (*Triticum aestivum* L.), rapeseed (*Brassica napus* L.), spring barley (*Hordeum vulgare* L.), and faba bean (*Vicia Faba* var. *minor* L.); ^bfaba bean received no N fertiliser

3. Results

For each method, several advantages and limitations were assessed (Table 2). The applicability was assessed in agreement with Goglio et al. (2015) and (JRC, 2011). The Y methods has several advantages including the ease of use; however PP better represents soil C dynamics, while the HH method better considers crop management prior to planting (Brankatschk and Finkbeiner, 2015; Goglio et al., 2015; Nemecek et al., 2015). However, a higher level of applicability was attributed to the Y due to its ease of use since many biophysical models and emission factor accounting procedures provide output on an annual basis.

Figure 2 shows the overall results for GWP, soil CO₂ and soil N₂O emissions. It is clear that there is large variability between years and across the different cropping systems. This high inter-annual variability resulted in a statistical analysis that showed no significant differences (p<0.05) for the three methods across the entire time period. However, for GWP on a per year basis, the differences across the methods can reach up to 10,800 kg CO₂eq ha⁻¹y⁻¹ with higher values for the RES system; while the average differences among methods ranged between -430 and 41 kg CO₂eq ha⁻¹y⁻¹, representing at most 18% of the standard deviation (1130-4120 kg CO₂eq ha⁻¹y⁻¹) for all the methods and cropping systems. The estimated GWP per ha ranged from -3870 to 12100 kg of CO₂eq ha⁻¹ y⁻¹ (Figure 2a).

A similar pattern was found for soil CO₂ emissions, with no significant difference among the methods tested (Figure 2b). However, on average the difference among methods reached up to 450 kg CO₂ ha⁻¹y⁻¹, while the maximum difference reached 10,600 kg CO₂eq ha⁻¹y⁻¹ within the RES system. The average difference represented up to 27% of the standard deviation which ranged between 884-

Table 2 Summary of the main observations for the 3 different methods compared (Y: year method; PP planting to planting method; HH: harvest to harvest method)

Methods	Y	PP	HH
Advantages	Ease of use; In Canadian conditions no management activity occurs early or late in the year due to frozen ground	It includes residue incorporation due to tillage in the autumn or spring C inputs from residues to the appropriate crop.	Impacts for spring tillage operations including seedbed preparation are reflective of the following crop
Disadvantages	Highly affected by carry over from previous crops; autumn tillage carried out for the following spring crop are attributed to the previous crop	The impacts of the tillage operations carried out prior to seeding are attributed to the previous crop	It attributes residue incorporation due fall tillage and also decomposition to the following crop
Applicability	High	Medium	Medium

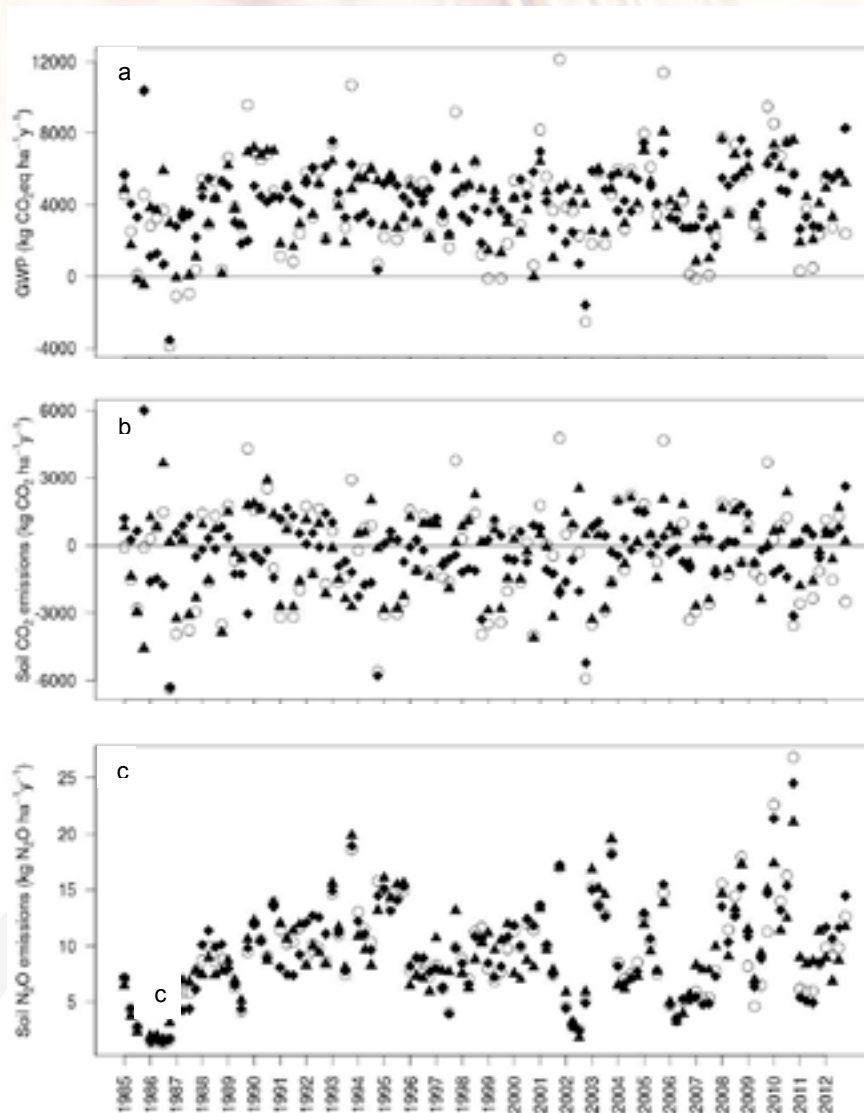


Figure 9 GWP (a), soil CO₂ emissions (b), soil N₂O emissions (c) for the four cropping systems on year basis with the 3 different methods tested (○ Year period (Y), ▲ planting to planting (PP), ◆ Harvest to harvest (HH))

3200 kg CO₂ ha⁻¹y⁻¹. Soil CO₂ emissions on a per ha basis were estimated between -6350 and 6000 kg CO₂ ha⁻¹ y⁻¹.

There was also no significant difference between the three methods for soil N₂O emissions (Figure 2c). However, the values on a yearly basis were clearly more similar than results for CO₂ and in contrast to GWP and soil CO₂ emissions, N₂O emissions for certain crop and cropping system combinations were found to be similar between the methods tested (Figure 2c). The inter-annual variability in N₂O emissions was very high for all methods, which was not surprising considering the site specific variability was also high (Uzoma et al., 2015). The standard deviation for each method (by plot) ranged between 3.1 and 5.2 kg N₂O ha⁻¹y⁻¹, at least 48 times larger than the average difference among methods (-0.082 to 0.084 kg of N₂O ha⁻¹y⁻¹). The simulated soil N₂O emissions ranged from 1.3 to 27 kg of N₂O ha⁻¹ y⁻¹.

4. Discussion

The main objectives were to compare different methods to attribute the timing of GHG emissions in agricultural LCA. The main advantages of the yearly method is the ease of use, since some methods

for estimating GHG emissions only operate on an annual basis (such as Tier I and Tier II IPCC methodology, (Paustian et al., 2006)). Instead, the PP method had the advantage of including residues dynamics of the planted crop, and the HH method considered the effect of the associated crop management prior to planting. In contrast, the latter two methods had lower applicability, while the yearly method was affected by carry over effect from previous crops and attributes autumn tillage effects to the previous crop. A previous experiment was carried out by Topp et al. (2011) with soil N₂O emissions while here soil N₂O emissions together with soil CO₂ emissions were assessed. Furthermore, Topp et al. (2011) assessed both spring and winter crops, while here just spring crops were evaluated.

The results obtained show that over the long term all the methods are equivalent. The differences between methods was overshadowed by high inter-annual variability, however on a year to year basis large differences between methods were obtained for both GWP (up to 10,800 kg of CO₂eq ha⁻¹y⁻¹), soil CO₂ emissions (up to 10,600 kg of CO₂ ha⁻¹y⁻¹) and N₂O emissions (up to 5.8 kg of N₂O ha⁻¹y⁻¹). This suggests that an important accounting error may occur when only one year is assessed. Therefore, long term assessment should be performed to avoid uncertainties related to GHG attribution to different crops. This was in agreement with previous findings which highlighted the importance of carrying out both LCA of crops and cropping systems at the same time to better consider the carry over effects over long periods (Goglio et al., 2014; Nemecek et al., 2015).

The present results suggest that the residue retained system is more strongly influenced by the various methods on a year to year basis, thus an appropriate multimethod comparison should be carried out. Indeed soil C dynamics are slow and a long period of time is necessary to detect potential changes in crop management affecting the overall soil C content (Goglio et al., 2015; Paustian et al., 2016). This is generally better captured by the PP method since in the RES systems crop residues are tilled before planting time the next year. Alternative approaches suggested by Knudsen et al. (2014) consider residue decay, but they are more complex than the methods presented here. Thus, the choice of methods results in a compromise between the applicability and accuracy of the methods with regards to the objectives of the LCA, as previously discussed (Garrigues et al., 2012; Goglio et al., 2015). The present recommendations could be considered valid and reliable for spring crops in temperate/continental conditions. Due to the complexity of cropping system interactions and the dependency of GHG emissions on regional conditions (Brankatschk and Finkbeiner, 2015; Goglio et al., 2014; MacWilliam et al., 2014, 2014; Nemecek et al., 2015), the present recommendations cannot be extended for other cropping systems in different climates.

GWP values obtained here were larger in range (-3870 to 12,100 kg CO₂eq ha⁻¹y⁻¹) in comparison to previous research (-6277 to 5190 kg CO₂eq ha⁻¹y⁻¹) (Camargo et al., 2013; Dendooven et al., 2012; Dyer et al., 2010; Goglio et al., 2014; Kim et al., 2009; Pelletier et al., 2008) who assessed several crops including wheat, rapeseed and maize in North American conditions using either agroecosystem models or direct measurements for GHG emissions. Present CO₂ emissions had a lower range (-6350 to 6000 kg of CO₂ ha⁻¹ y⁻¹) than several studies for North American conditions (-17,100 to 2270 kg of CO₂ ha⁻¹y⁻¹) (Dendooven et al., 2012; Ellert and Janzen, 2008; Hernandez-Ramirez et al., 2011). However, the soil N₂O emissions from the study site for which DNDC was validated (Uzoma et al., 2015) were highly variable but the modelled results were considered to be reasonably accurate in comparison with observations (Uzoma et al., 2015). Further perspectives would include the complete assessment of methods to account for soil GHG emissions together with smaller secondary N₂O emissions (i.e, NH₃ volatilisation and nitrate leaching).

5. Conclusions

The present research compared different methods to attribute the timing of soil GHG emissions in agricultural LCA of spring crops in a continental climate. The long term differences between the methods were negligible for spring crops in continental Canadian conditions. In contrast, on an annual basis, the methods showed large differences with regards to GWP and soil CO₂, particularly in systems with no residues harvested. In this situation, a multimethod comparison is recommended to avoid erroneous conclusions. In the choice of the methods, a compromise should be found between accuracy and applicability in relation to the objectives of the study. Present findings can be applied in similar

climatic areas and for similar crops. Further perspectives include carrying out this LCA comparison with winter crops and in other climates, including secondary N₂O emissions.

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206. Life-Cycle Assessment of Table Grape Production in Eastern Austria

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ABSTRACT

In Austria, a domestic table-grape producing sector is emerging, with production being pioneered by small, family-owned vineyards. While life-cycle assessment (LCA) results are readily available for wine grapes, table grape LCAs have not been found. The objective of this work was the quantification of selected environmental impacts of table grape production in three small case study vineyards in Eastern Austria, enhanced by a comparison with a hypothetical reference vineyard and by an identification of optimization measures. The method is a cradle-to-gate, attributive life-cycle assessment (LCA) with a functional unit of one kilogram of table grapes at the first point of sale. Results demonstrate that impacts can vary substantially between vineyards. Climate change impacts (GWP100) range from 0.30 to 1.05 kg CO₂-eq/kg grape, mainly due to machinery operations, the production of packaging materials, and fertiliser application. Freshwater eutrophication impacts range from 0.09 to 0.18 g P-eq/kg grape, terrestrial ecotoxicity impacts range from 0.09 to 1.76 g 1,4-DCB-eq/kg, and human toxicity impacts range from 0.13 to 0.28 kg 1,4-DCB-eq/kg. Large uncertainties that preclude any differentiation between vineyards were found for eutrophication and human toxicity impacts. A much higher assumed grape yield in the hypothetical reference vineyard results in substantially lower impacts, but with the yield assumption adjusted to match those in the example vineyards, the reference impacts also become comparable. Options to reduce impacts in the example vineyards include the use of modern, efficient tractors with fewer cultivation steps and a less material-intensive packaging system. To the authors' knowledge, the study for the first time presents LCA results on table grape production in small Austrian vineyards, and its findings may help to lower future impacts by pointing out effective improvement measures.

Keywords: Table grapes, attributive LCA, hot spots, sensitivity analysis.

1. Introduction

Table grapes consumption in Austria is dominated by imports (Statistik Austria 2016), but a domestic table-grape producing sector is emerging, with production being pioneered by small, family-owned vineyards. Potential advantages of local table grape production include a streamlined supply chain with more disease-resistant varieties, less storage and shorter transport distances, allowing for ripening at the vine and thus enhanced flavors (Ollig 2010). Table grape varieties are distinct from wine grapes; while the environmental impacts of wine production have been extensively studied (e.g. Bosco et al. 2011, Christ and Burritt 2013, Rugani et al. 2013), similar work on table grapes is lacking. The objective of this work was a cradle-to-gate, attributive life-cycle assessment (LCA) of table grape production in three small vineyards in Eastern Austria. Specific sub-objectives were (a) to quantify four potential environmental impacts (global warming potential with a 100-year horizon, GWP100; freshwater eutrophication potential, FEP; terrestrial ecotoxicity potential, TETP; and human toxicity potential, HTP), to compare these impacts to those of a hypothetical reference vineyard, and to identify options to reduce the potential environmental impacts in this emerging agricultural sector.

2. Methods

The three case study vineyards are situated in established wine production regions in the two eastern provinces of Austria, Lower Austria and Burgenland. Their characteristics are summarized in Table 1. In all three vineyards, the vines are vertically trained to a bilateral cordon, but other characteristics differ. Vineyard A is partially situated on a steep slope with sandy soil that is low in potassium and phosphorus. On a total area of one hectare, 2000 vines are planted. The area comprises three smaller vineyards, all at a distance of approximately 1.5 km from the farm. The expected yield of approximately 7.0 tons of grapes per hectare and year is an estimate since one of the three smaller vineyards had just been planted at the time of data collection. Fertilisation is currently limited to green manure, a perennial grass-clover mix (approximately one quarter grass). Additionally, all woody and leafy residues are mulched into the soil three times a year. The vineyard currently limits pesticide use to a 80% wettable sulfur fungicide dispersion. Tillage, mulching, and other cultivation requires 53 hours of tractor operation annually, and harvest requires 40 h/a with an old (1965) tractor model. Grapes are pruned and harvested manually. In this and the other vineyards studied, all grapes are assumed to be stored after harvest in chilled storage halls, and no grapes of lesser quality are used as co-products for juice production. Forty percent of vineyard A's production is sold at local farmers' markets, the remainder is sold directly on the farm to mostly local customers living within 10 km of the farm. The packaging consists of cardboard trays of 0.5 or 1.0 kg capacity.

Table 1: Key characteristics of the three studied vineyards A-C and the hypothetical reference vineyard R. Data for vineyards A-C are based on questionnaires completed by vineyard owners; data for vineyard R are based on Richter (2010), unless noted otherwise.

Characteristic	Vineyard A	Vineyard B	Vineyard C	Vineyard R
Planted area, ha	1 ^a	8	0.66	1 ^b
Yield, kg/ha/a	7000 ^a	6500 ^a	7500 ^a	14000 (Ziegler 2011)
Management system	“Natural” (owner’s specification)	Organic	Organic	Conventional
Fertiliser	Green manure, mulched 3x per year, no tillage	Green manure, mulched 4x per year, with tillage	Green manure, mulched 3x per year, with tillage	Mineral fertiliser ^c , green manure mulched 1x per year
Pesticides (kg/ha/a)	Sulfur (1.6)	Sulfur (11.7), copper (0.17), others ^{d,e}	Sulfur (15), copper (2), others ^e	Sulfur (12), phosphonic acid (3), copper (1) (Redl 2013)
Tractor operation (year produced, rated power in	Tractor 1 (2010, 66, 40); Tractor 2 (1965, 48,	Tractor 1 (2006, 63, 9.8); Tractor 2 (2006, 52,	Tractor 1 (1987, 52, 45.5); Tractor 2 (2004, 55,	Tractor 1 (2002, 45, 27) ^{b,f}

kW, hours/ha/a)	53)	8.2)	30.3)	
Packaging (kg grapes per container)	Cardboard trays (0.5 kg or 1.0 kg)	PET trays (0.5 kg) in corrugated cardboard one-way boxes (5kg)	PET trays (0.5 kg) in reusable PP crates (5-7kg)	LDPE bags (0.5 kg) in reusable PP crates (5-7kg) ^b
Transport to first sales point	60% sold through direct marketing (10 km by diesel car); 40% sold at farmer's markets (40%, 13 km by diesel car)	To supermarkets (150 km by a 3.5-7.5 ton lorry)	To supermarket (90 km by van < 3.5ton)	To supermarket (30km by van < 3.5 ton) ^e

^a approximate value; ^b authors' assumptions; ^c mass per ha and year: 40 kg N, 20 kg P₂O₅, 70 kg K₂O, 25 kg MgO (Ziegler 2011); ^d includes ethanol, potassium bicarbonate, tensides, pheromones (not modelled); ^e algae-based phytoestimulants (not modelled); ^f model tractor from the ecoinvent v2.2 (ecoinvent Centre, 2010) database

Vineyard B is an organic vineyard situated at an approximate distance of 0.5 km from the farm on gently sloped sandy loess. The total area of 8 ha is split into 2 four-hectare sections, with 2800 vines per hectare yielding 6.5 t/ha/a. Green manure cover (80% legumes) is maintained except during the main vine growth period between May and July, with mulching four times a year. The vineyard applies pesticides and preventative plant protection chemicals, including wetting sulfur, copper, potassium bicarbonate, a wetting agent, and a foliar fertiliser. An algae-based plant strengthener and a pheromone agent could not be included in the LCA model due to a lack of inventory data. Two 2006 tractors with 63 kW and 52 kW are operated for a total of 18 hours per hectare and year. Grapes are manually harvested with 5-kg multi-use PP crates and packaged in 0.5 kg PET trays that are contained in 5-kg corrugated cardboard crates as requested by the customer, a distributor located at a distance of 150 km from the vineyard.

Vineyard C is an organic 0.66-ha vineyard situated at an approximate distance of 4 km from the farm on gently sloped sandy loam. 2244 vines yield 4950 kg grapes per year, or 7.5 tons per hectare and year. Green manure is re-seeded every other year and mulched three times a year, and woody and leafy residues are incorporated into the soil as well. Plant protection consists of wetting sulfur, copper, and an algae-based plant strengthener that could not be modelled due to a lack of data. Mulching and other cultivation, as well as part of the harvest are conducted with an older model tractor (1987) for a total of 45 hours, and the other part of the harvest, as well as tillage operations are conducted during 30 h/a with a newer (2004) tractor model. Grapes are manually harvested directly into PP crates that contain 0.5-kg PET trays and shipped in a small van to the sole customer, a supermarket at a distance of 90 km from the vineyard.

A hypothetical, one-hectare conventional reference vineyard (vineyard R) was assembled from literature data, mainly from a German model cost calculation by Richter (2010). The vineyard is assumed to be located at a 5-km distance from the farm; it grows 4000 vines with a yield of 14 t/ha/a (Ziegler 2011). Permanent green manure is mulched once per year, as are all woody and leafy residues. This is in addition to mineral fertilisers (ammonium nitrate, diammonium phosphate, magnesium oxide, potassium chloride, and Boron). It was further assumed that the vineyard applies fungicides in the form of sulfur, copper, and phosphonic acid in the course of six applications a year. Tractor operation was modelled at 27 operating hours per hectare and year, with ecoinvent data that describe a 2002 tractor. Harvest is done manually into reusable PP crates that are also used for storage and transport. Grapes are assumed to be sold to a supermarket at a distance of 30 km, packaged in 0.5-kg LDPE-bags. This hypothetical vineyard differs from the three case study vineyards mainly in the much higher grape yield and in the use of mineral fertiliser. The influence of the higher yield assumption on the results is demonstrated below as part of a sensitivity analysis.

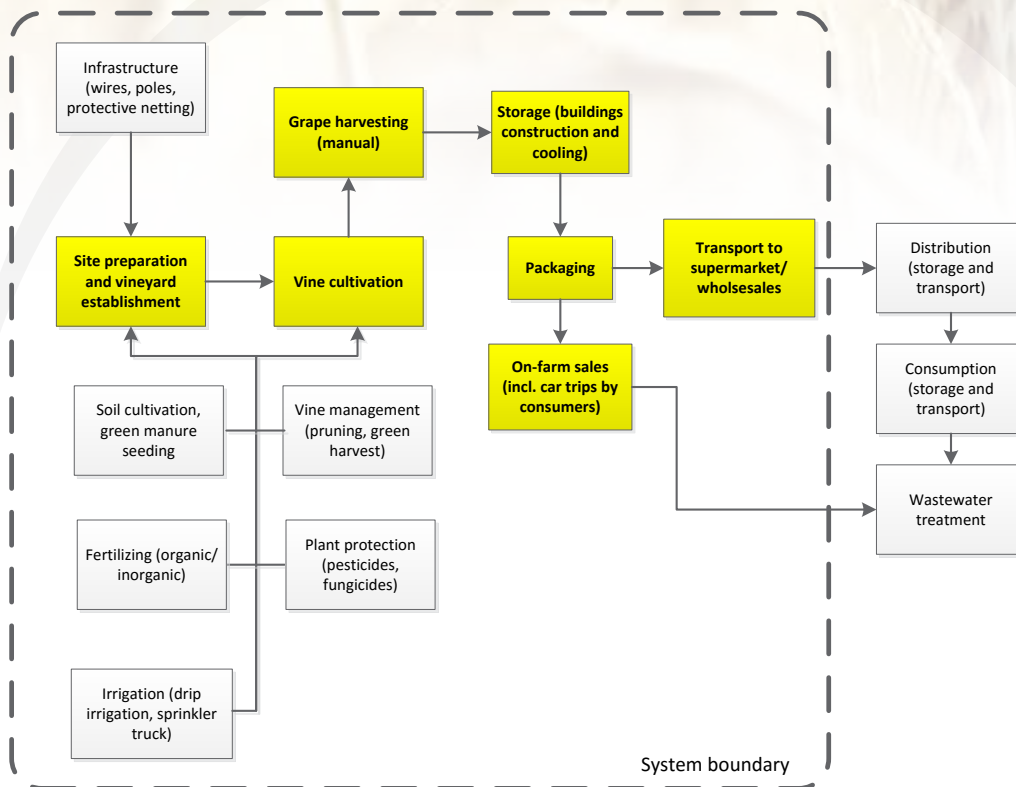


Figure 1: Simplified system diagram of table grape production in Eastern Austria

The life-cycle inventory model for the table grapes shows technical system boundaries (Figure 1) that model the cradle-to-gate process chain of table grape production. The models include the full production chains, from establishing the vineyard, to grape storage (2 days in chilled rooms) and packaging, to the transport to the first point of sale. An attributive LCA approach was deemed suitable since the focus was on assessing emerging local table grape production rather than on comparing such production with that of the currently prevalent imported table grapes. The functional unit was one kilogram of table grapes at the first point of sale, either to private customers or to a distributor/supermarket. The LCA model was implemented with the openLCA software (version 1.4.2; Green Delta GmbH, Berlin, Germany) combined with the ReCiPe(H) impact assessment method (Goedkoop et al. 2013). Four of ReCiPe(H)'s impact categories were chosen as representing anticipated environmental impacts: the 100-year global warming potential (GWP100), the freshwater eutrophication potential with infinite time horizon (FEP), the freshwater ecotoxicity potential with infinite time horizon (FETP), and the human toxicity potential with infinite time horizon (HTP). Primary data were collected from interviews with vineyard owners, secondary data were obtained from literature and from the ecoinvent v2.2 (ecoinvent Centre, 2010) database. Parameter uncertainties (e.g. vineyard poles, wires, and nets; hours of tillage; N₂O emissions from fertilisation; and transportation distances), were propagated to total impact uncertainties using Monte-Carlo simulations with 10,000 runs for the vineyard total impacts, and 1,000 runs for the contribution analyses.

3. Results

A comparison of overall results between the four vineyards (Figure 2) shows that the climate change impact is higher for all three case study vineyards than that of the hypothetical reference vineyard R, with a GWP100 of 0.49 to 1.05 kg CO₂-eq/kg grape for vineyards A to C compared to 0.30 kg CO₂-eq/kg for vineyard R. This is largely due to the above-mentioned higher yield assumed for vineyard R that distributes impacts over a larger amount of grapes. Other reasons for higher GWP100 of the case study vineyards include the operation of older, less efficient tractors for more hours per year in vineyards A and C, which contributes 69% and 40.0% to the total GWP100,

respectively, and relatively material-intensive packaging (PET polymer trays) by vineyards B and C (see Table 1).

Vineyard R impacts are lowest also with respect to the eutrophication potential FEP and the human toxicity potential HTP, with the exception of vineyard A's eutrophication potential (Figure 2), again due to the high yield assumption for the reference vineyard. However, large uncertainties blur the differences between vineyards in both impact categories.

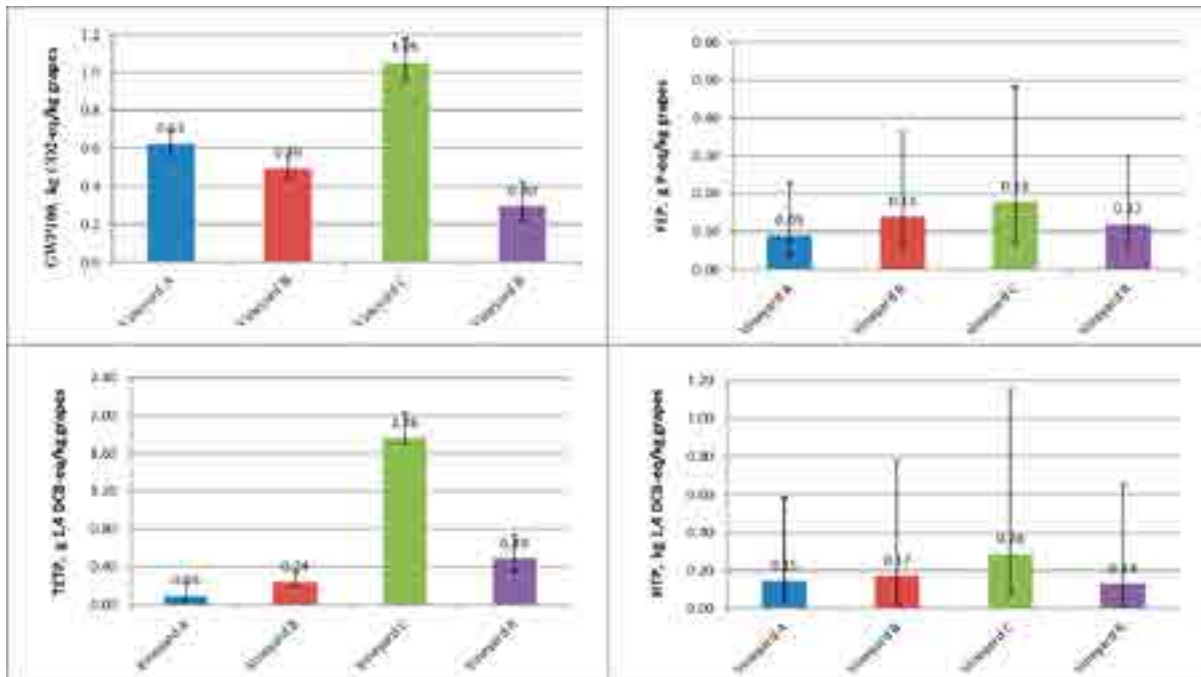


Figure 2: Potential impacts per kg table grapes for four selected impact categories of the three studied vineyards A to C and the hypothetical reference vineyard R. GWP = global warming potential, FEP = freshwater eutrophication potential, TETP = terrestrial ecotoxicity potential, HTP = human toxicity potential (error bars = 5th and 95th percentiles, Monte-Carlo simulations, n=10,000)

The terrestrial ecotoxicity impact of table grape production varies widely between vineyards, with vineyard C showing by far the highest result, almost twenty times that of vineyard A. This is explained by the main contribution to the TETP, the application of copper as a fungicide; Vineyard C applies the highest copper amounts, while vineyard A reported that it does not apply any copper (s. Table 1). However, the use of copper is considered essential by Austrian experts; an assumed copper use by vineyard A of 1 kg/ha/a would increase its total TETP more than tenfold, from 0.09 to 0.98 g 1,4-DCB-eq/kg grapes.

The contribution analysis shows that dominant processes are vineyard- and impact-specific. For the climate change impact, vineyards A (Figure 3) and C (data not shown) are dominated by tractor operations (69% and 46% of the total impact, respectively) due to their high machinery operation hours and use of old tractors for some processes (see Table 1). For vineyards B and R (data not shown), potential contributions to climate change are more evenly spread, but diverse. In vineyard B, the largest contribution to climate change is the manufacturing of PET packaging trays (27% of total GWP100), while in vineyard R the highest contribution is due to the application of mineral fertiliser (21% of total GWP100), both from nitrous oxide emissions after field application and from ammonium nitrate production emissions.

Freshwater eutrophication is dominated by infrastructure manufacturing processes for all vineyards (machinery in vineyard A, materials such as steel support poles and wires in vineyards C and R), as well as by phosphate emissions from plant protection and from PET manufacturing for packaging vineyard B grapes. The terrestrial ecotoxicity potential is dominated by emissions of copper fungicide

in all vineyards, with the exception of vineyard A, which does not use copper. There, 32% of a relatively low total TETP is due to hydrocarbons emitted during operation of the very old (1965) tractor, followed by machinery and diesel production emissions. Emissions from this tractor are also the largest contribution (28% of total impacts) to the human toxicity potential impacts of vineyard A. Vineyard materials manufacturing, the second-largest contribution to HTP, dominates the impacts in the other vineyards, with 40%, 32%, and 56%, of total impacts for vineyards B, C, and R, respectively.

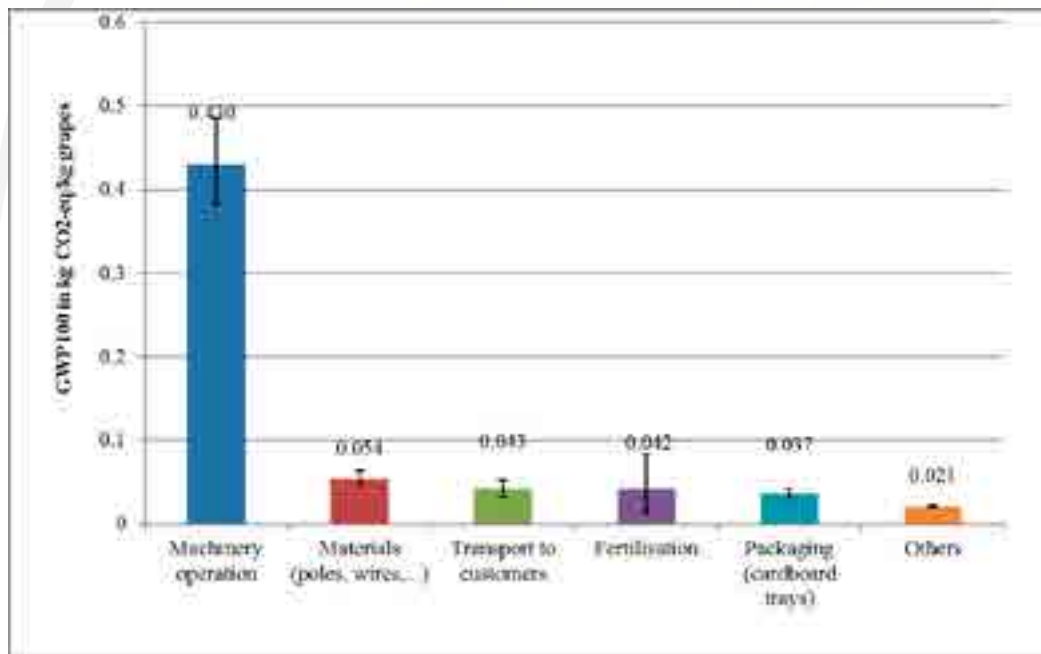


Figure 3: Contributions to global warming potential of vineyard A grapes (error bars = 5th and 95th percentiles, Monte-Carlo simulations, n=1000)

4. Discussion

The comparison between the three case study vineyards and with the hypothetical reference vineyard R demonstrates that impacts can vary substantially between vineyards. Not surprisingly, the grape yield largely determines relative environmental impacts; the yield may not only be limited by local soil and climate conditions, but also by quality considerations. Comparison with carbon footprints in literature on wine grape production shows that the table grapes' footprints of 0.30 - 1.05 kg CO₂-eq/kg are comparable to the wide range of carbon footprints in published wine grape studies. For example, Bosco et al (2011) give a values for four different wines from the Tuscany region in Italy that can be re-calculated to 0.11-0.32 kg CO₂-eq/kg (wine) grape, assuming 1.1 kg grapes used to produce 0.75 liters of wine. At the other end of the range, Neto et al. (2012) report a carbon footprint of 2 kg CO₂-eq for 0.75 liters of Portuguese *vinho verde* wine, which would correspond to 1.82 kg CO₂-eq/kg (wine) grape. As the dominating contributors, the two studies identified greenhouse gas emissions from the production and application of mineral fertiliser, as well as from machinery operations. A similar literature analysis for the impact category freshwater eutrophication potential (data not shown) indicates that the results calculated for the case study vineyards are comparable, but in parts lower, than literature results for wine grapes.

As part of a sensitivity analysis, the effects of three hypothetical changes to the vineyard models were calculated: First, the old tractors used for some operations in vineyards A and C (1965 and 1987, respectively) were replaced with the 2002 model from the ecoinvent 2.2 database. This resulted in a 28% reduction of GWP100, from 0.63 to 0.45 kg CO₂-eq/kg grape for vineyard A, and a 38% reduction for vineyard C, from 1.05 to 0.66 kg CO₂-eq/kg grape. Somewhat smaller reductions would affect mainly the human toxicity impacts in these vineyards. Second, the effect of altered packaging

material for the grapes was investigated in vineyard B, by exchanging the climate-change dominating PET trays with lighter LDPE bags. This results in large reductions, particularly in the eutrophication, climate change, and human toxicity impacts (Table 2). It should be noted, however, that the choice of packaging material is currently made by the vineyard's customer and not by the vineyard owner.

Table 2: Sensitivity result – Vineyard B, effect of packaging dematerialization

Impact category	Packaging materials		Effect on impact category (% change)	
	Baseline packaging ^a	Reduced packaging ^b		
GWP100 [kg CO ₂ -eq/kg]	0.50	0.34	- 0.15	(-30.5%)
FEP [kg P-eq/kg]	1.40*10 ⁻⁴	7.75*10 ⁻⁵	- 6.25*10 ⁻⁵	(-44.6%)
TETP [kg1,4-DCB-eq/kg]	2.40*10 ⁻⁴	2.10*10 ⁻⁴	- 0.30*10 ⁻⁴	(-12.5%)
HTP [kg1,4-DCB-eq/kg]	0.170	0.12	- 0.06	(-31.8%)

^a PET trays (0.5 kg) in one-way corrugated cardboard boxes (5 kg)

^b LDPE bags (0.5 kg) in reusable PP crates (5-7 kg)

Third, the effect of reducing the yield of the hypothetical reference vineyard R was estimated. A reduced yield of 7 t/ha/a was assumed (down from an original 14 t/ha/a; Ziegler 2011), which would be comparable to the case study vineyards' yields of 6.5 to 7.5 t/ha/a. As a coarse approximation, the fertiliser application was halved, but pesticide amounts were left unchanged. As a result, total impacts in all impact categories of vineyard R increased substantially, from an almost 50% increase in the climate change category, to almost doubling the terrestrial ecotoxicity potential impact (Table 3).

The LCA results presented here may be interpreted as estimates of selected potential environmental impacts; however, other environmental aspects of table grape production are not covered. For example, no data were available on the effect of table grape production on soil carbon dynamics. For wine production, Bosco et al. (2013) have shown that accounting for soil organic matter can change the vineyard portion of a wine's carbon footprint from a GHG source to a net sink, depending on the

Table 3: Sensitivity result – Vineyard R, effect of reducing yield from 14 t/ha/a to 7 t/ha/a, comparable to those of case study vineyards (6.5 to 7.5 t/ha/a).

Impact category	Annual grape yield		Effect on impact category (% change)	
	Baseline (14 t/ha/a)	Reduced (7 t/ha/a)		
GWP100 [kg CO ₂ -eq/kg]	0.30	0.44	0.145	(+ 48.8 %)
FEP [kg P-eq/kg]	1.20*10 ⁻⁴	1.89*10 ⁻⁴	0.69*10 ⁻⁴	(+ 57.8 %)
TETP [kg1,4-DCB-eq/kg]	4.90*10 ⁻⁴	9.60*10 ⁻⁴	4.70*10 ⁻⁴	(+ 95.9 %)
HTP [kg1,4-DCB-eq/kg]	0.13	0.24	0.11	(+ 82.3 %)

site and on management practices. Other environmental impacts that were not included here but may be of interest, such as solid waste production and land use, have been summarized for wine production by Christ and Burritt (2013). Further limitations include plant protection agents that could not be modelled due to a lack of LCA data (pheromones and phytostimulants), as well as the exclusion of the effects of temporal variability (for example, large increases in copper use in precipitation-rich years, and yield fluctuations in general). While parameter uncertainties were propagated to the end results using Monte Carlo simulations, most model uncertainties (for example, the actual copper emissions from copper fungicide application) could not be included in the calculation due to a lack of reliable data.

5. Conclusions

Selected potential environmental impacts could be quantified with an attributional cradle-to-gate LCA for three Austrian table grape production systems and for one hypothetical reference system, from vineyard establishment to the first point of sale. The comparison between the three case study vineyards and with the hypothetical reference vineyard R demonstrates that variability between vineyards can be substantial. Climate change impacts expressed as GWP100 range from 0.30 to 1.05 kg CO₂-eq/kg grape, mainly due to machinery operations, the production of packaging materials, and fertiliser application. Freshwater eutrophication impacts range from 0.09 to 0.18 g P-eq/kg grape, with major contributions from manufacturing machinery and vineyard infrastructure, from preventive fungicides and fertilisers, and from PET tray packaging production. Terrestrial ecotoxicity impacts range from 0.09 to 1.76 g 1,4-DCB-eq/kg, caused to a large extent by copper fungicide application emissions. Human toxicity impacts range from 0.13 to 0.28 kg 1,4-DCB-eq/kg, with contributing processes similar to freshwater eutrophication. Large uncertainties that preclude any differentiation between vineyards were also found for the two impact categories of eutrophication and human toxicity.

A high yield assumption by the literature source for the hypothetical reference vineyard R leads to lower impacts relative to the three case study vineyards across all but one of the four categories considered. However, a reduced yield assumption that is in line with the three case studies, increases impacts to levels that surpass two of the three vineyards in all impact categories but climate change. With climate change, relatively low machinery use and light grape packaging leave the carbon footprint for the reference system below those of the case study systems even at comparable grape yields.

A comparison with literature on wine grape production shows that the table grapes' impacts are comparable or somewhat lower than those of the analysed wine grape studies.

Options to reduce the impacts of table grape production in the case study vineyards could be identified based on the contribution analysis. They include the use of modern, efficient tractors with fewer cultivation steps, and a less material-intensive packaging system. In the vineyard with the highest total GWP100, just replacing the old tractor with a modern model would reduce the grapes' total carbon footprint by 38%. In vineyard B, replacing the PET tray packaging with LDPE bags would substantially reduce impacts, but the choice of packaging material is made by the vineyard's customer and thus may not be easily changed.

To the authors' knowledge, the present study for the first time presents LCA results on the environmental impacts of table grape production in small Austrian vineyards, and its findings may help to lower future impacts by pointing out effective improvement measures.

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236. Including product multicriteria quality in life cycle assessment, application to grape

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ABSTRACT

Objective: In the context of protected designation of origin (PDO) wine production, wine organoleptic quality, and hence grape quality is a key target of vineyard management. LCA-based Improvement of Technical Management Routes¹ (TMR) by choice of more environmentally friendly techniques needs to take into account this quality dimension in addition to the yield function usually considered in wine and grape LCAs. The aim of the paper is to present and discuss two proposals for including multi-criteria quality into the eco-efficiency assessment of PDO grape production in order to support the choice, design and assessment of vineyard TMRs.

Methodology: We propose the design of a combined quality index Q relating an observed multi-criteria quality to a targeted quality. A set of logical rules of inference (if...then...) was used considering several levels of correspondence of the product to the quality target. From this Index, a Quality Functional Unit (QFU=Q) and a Mass x Quality Functional Unit (MQFU= yield x Q) were defined. For application of Q to grape, a typology of grape quality targets with the different quality criteria and their levels for each grape type was established with expert-oenologists as a basis for Q. We tested the sensitivity of the two indexes to a change of quality target. We implemented the two FUs in LCA calculations on five real viticultural TMRs representing the Middle Loire Valley diversity for production of a same type of wine

Main Results: The quality index Q was calculated for the five TMRs. The two FUs derived from Q were sensitive to a change of quality target. The LCA results of the five TMRs expressed per MQFU differed from results per QFU and were sensitive to a change of quality target. Results with QFU were close to those obtained with classical FUs (1 kg grapes and 1 ha vines.1year) due to minor differences in the value of Q,

Implications, meanings, conclusions: QFU appeared too restrictive while MQFU allowed accounting for the main function of the system: the production of a given quantity of quality grapes with a target of a given type of wine.

Key words: Environment, practices, winegrowers, LCA, impacts, Protected Denomination of Origin, wine

1. Introduction

In the context of Protected Designation of Origin (PDO) wine production, wine organoleptic quality, and hence grape quality is a key target of vineyard management. Life Cycle Assessment (LCA)-based Improvement of Technical Management Routes¹ (TMR) by choice of more environmentally friendly techniques needs to take into account this quality dimension in addition to the yield function usually considered in wine and grape LCAs. However, to our knowledge, none of the published wine sector LCA studies (Gazulla *et al.*, 2010; Vázquez-Rowe *et al.*, 2012; Fusi *et al.*, 2014; Villanueva-Rey *et al.*, 2014) accounts for quality criteria in spite of the importance of both grape quality in the final wine quality (Bravdo, 2001; Guidetti *et al.*, 2010) and wine quality for wine consumers (Lockshin and Corsi, 2012; Jourjon and Symoneaux, 2014).

Quality, in its broad sense, is scarcely considered in food and crop LCA studies. Müller-Lindenlauf *et al.* (2010) included predicted milk quality as an additional impact besides classical LCA environmental impacts of milk production. Nevertheless, the most frequent option in food LCAs has been to consider quality as one of the main functions of the product. LCA calculates the potential environmental impacts of the whole life cycle of a product related to a Functional Unit (FU) which corresponds to the product's main function. (Heller *et al.*, 2013). Hence quality has been included in the FU: Charles *et al.* (2006) used an FU including a single quality criterion for wheat: "1 equivalent ton grain with 13% protein". The multi-criteria nutritional value of various foods composing diets has more recently been considered in LCAs of diets through single indices resulting from the aggregation of the nutritional values of each foodstuff related to daily consumer needs (Kägi *et al.*, 2012; Saarinen, 2012; Heller *et al.*, 2013). Some authors even included qualifying and disqualifying nutrients in the score (Van Kernebeek *et al.*, 2014). For LCAs of meals, Inaba and Ozawa (2008) proposed a comprehensive food-value index constructed on the same principle but involving the taste, nutrient balance and health function of the dishes of a meal including weighting factors determined by consumer survey. Nevertheless, as pointed out by van der Werf *et al.* (2014), including quality considerations in FUs remains a major challenge for the LCA food community, especially for certified productions that favour quality over volume as is typically the case for PDO wine production. Like food nutritional quality, wine-grape quality is multi-criteria (Geraudie *et al.*, 2010). Its assessment is widely used for technical decisions: harvest date, allocating grapes to different cuvees, winemaking management, and the payment of grape providers. The most common quality indicators for white grapes are the sugar and soluble acids content, and also the polyphenol content for red cultivars. However, this assessment of grape maturity is more and more complemented with on-field sensory analysis (Winter *et al.*, 2004; Le Moigne *et al.*, 2008; Olarte Mantilla *et al.*, 2012; Siret *et al.*, 2013) especially for aroma, color or texture. The health of the berries is also a key quality determinant. *Botrytis* bunch rot is especially problematic for white wine production (Hill *et al.*, 2014).

The aim of this paper is to present and discuss a proposal for the inclusion of quality in the eco-efficiency assessment of quality grape production in order to support the choice and design of vineyard TMRs to preserve the environment while maintaining the targeted quality. We considered the concept of eco-efficiency (Huppes and Ishikawa, 2005) as relevant to express the objective of improvement of environmental performance while maintaining a targeted quality level. We used it taken as the ratio between, as numerator, emissions and resource use and as denominator, the service they provide, expressed by the FU (Kicherer *et al.*, 2007). In this research, a synthetic index of grape quality and a (quality x yield) index were designed and used as FUs. This paper presents i) the two indexes formula construction and the grape quality measurements methods; ii) The results of their application to five contrasting TMRs, eco-efficiency results for mass- and quality-based FUs compared and discussed, iii) a wider discussion on methods and perspectives and iv) a conclusion.

2. Material and methods

The research work was conducted in Middle Loire Valley PDOs, on five real TMRs (TMR1 to 5) that represent the regional diversity of vineyard management for PDO Chenin Blanc dry white wine production (Renaud-Gentié *et al.*, 2014). The cases were studies for harvest 2011.

2.1 Life Cycle assessment

¹ Technical Management Routes (TMRs): logical chain of practices managed by a farmer in a field (Sébillotte, M., 1974)

Data for LCA were collected from winegrowers and completed with expertise, experimental results and databases. Impacts of grapes harvested were calculated on the basis of field operations implemented in 2011, occasional operations amortized according to the frequency of implementation and operations done during non-productive periods and amortized on 30 years. The LCA was conducted from cradle to field gate. Quantification of direct emissions linked to the use of all inputs and their distribution in environmental compartments were calculated with the models proposed by (Koch and Salou, 2014) for nitrate (NO₃⁻), nitrogen dioxide (N₂O), nitrogen oxides (NO_x), phosphorus, heavy metals, and fuel combustion. Pesticide emissions were calculated with PestLCI 2.0 (Renaud-Gentié et al., 2014) and Ammonia (NH₃) according to (Hutchings *et al.*, 2013) Tier2 approach.

Due to the huge quantity of data generated in this study, we present here only the results for three impact categories that proved to give very different patterns in the results: “Global warming potential at term 100 year” (GWP 100a), calculated with IPCC (Solomon et al., 2007) model, “Fresh water ecotoxicity potential” (FwEtoxP) calculated with USETox™ V1.03 (Rosenbaum *et al.*, 2008) characterization method and “Abiotic resources consumption” (Res) calculated with EDIP (2003) method. LCA results of the five TMRs were calculated per ha (not presented here) and in this paper, eco-efficiency per kg are compared with the results of the two new quality-based FUs:

2.2 Grape quality index

The quality of a product is defined by (ISO, 2005) as the “degree to which a set of inherent characteristics fulfills requirement” (we prefer “Target” to “requirement” for grape quality). Still, grape quality criteria do not always have a linear relationship with the target, but rather various types of relationship, depending on the nature of the criterion. For example, sugar content can be optimal (for a given targeted wine) between 200 and 220 g/l, refused under 200 g and above 250g/l, while be accepted with a lower satisfaction between 220 g/l and 250 g/l. We propose to solve this problem by a set of logical rules of inference for each criterion considering several levels e of correspondence to the target: $C_{ig} = 100\%$: perfect correspondence, $C_{ig} = 0\%$: refused. If secondary targets are acceptable, intermediate levels are added with lower but acceptable degree $C_{ig} = e_i\%$ of correspondence to the target, as many levels of correspondence as needed must be added. For a given targeted grape type, for an assessed grape g , described by n criteria, with $i=1$ to n , the degree of correspondence C_{ig} of the grape g to the quality target, for criterion c_i is calculated according to the following formula:

$$\text{Équation 1: } C_{ig} = \begin{cases} 100 & \text{if } c_{ig} \in A_i \\ e_i & \text{if } c_{ig} \in B_i \\ 0 & \text{if } c_{ig} \in D_i \end{cases}$$

with $A_i \cap B_i = \emptyset$ and $B_i \cap D_i = \emptyset$ and $A_i \cap D_i = \emptyset$
and with $A_i \cup B_i \cup D_i$ include all c_{ig} and where:

C_{ig} = degree of correspondence to the target of criterion i for grape g .

c_{ig} = value of criterion i for grape g

A_i = set of values of criterion i corresponding to the target quality

B_i = set of values of criterion i corresponding to a secondary target quality, considered as acceptable

e_i = value of degree of correspondence of a secondary target quality to the initial target for criterion i

D_i = set of values of criterion i considered unacceptable

The limits of sets A_i , B_i , and D_i and e_i are fixed considering that the secondary target is $(1-e_i)\%$ less satisfying than the primary target.

The quality index Q_g is the global degree of correspondence to the quality requirements for the grape g and is the result of the weighted average of the degrees of correspondence to the target of each criterion:

$$\text{Equation 2 : } Q_g = \frac{\sum_{i=1}^n w_i C_{ig}}{\sum_{i=1}^n w_i} \quad \text{with: } Q_g = \text{quality index of grape } g \text{ and } w_i = \text{weight given to criterion } i$$

Applying the previous definition of the quality index to wine grapes implies defining A_i , B_i , e_i and D_i , i.e. the primary and secondary target grape types and their inherent characteristics. These characteristics are the criteria describing the grape and the gap between the primary and secondary targets (grape types that will make an acceptable quality wine but not matching the initial target).

Expert knowledge elicitation has been used (Tobias and Tietje, 2007) with nine expert practitioners who frequently deal with Chenin Blanc grapes for different middle Loire Valley PDO dry wine type production. After individual face-to-face interviews with each of the experts, a consensus session between them enabled them to reach an agreement about the primary grape quality criteria, the main grape types and their characteristics. The sugar content, aroma maturity (green, fresh fruit, cooked fruit), health and color of the berries (green to golden) were identified as the key parameters that differentiate the types of Chenin Blanc grapes for middle Loire Valley PDO dry wines. The values of the criteria corresponding to the quantitative (for sugar and rot) or qualitative (for berry color and aroma) limits of A_i for each grape type are listed in Table 1.

Table 1: Chenin Blanc grape types suitable for dry still wine in the middle Loire Valley PDO context and their characteristics according to expert consensus

Berry color	Dominant aroma	Sugar content in potential % alcohol	% of rotted berries	Type	Type code
Green or yellow	Fresh fruits	11>>13	< 10% *	Fresh dry wine	FD
Golden	Ripe fruits	13>>14.5	< 10% *	Ageing dry wine, ripe aromas	ADR
Golden	Cooked fruits, jam, honey	14.5>>16	< 10% *	Ageing dry wine, over-ripe aromas	ADOR
Golden	cooked fruits, jam, honey	14>>16	Noble rot	Ageing dry wine, noble rot aromas	ADN

*if it is grey mould evolving into noble rot, then rot is accepted

We translated the qualitative values given by the experts for color and aroma into quantitative ones relative to the existing scales that our sensory analysis panel was trained to use.

The experts named the grape types according to the wine that they were the most suitable for. Correspondence between grape typology and single criteria assessment was carried out through the inference rules presented in Table 2. These rules were determined for the type of wine targeted by the growers in this study: type ADR = dry quality wine for ageing with ripe aromas as main target and types FD = fresh dry quality wine and ADOR = ageing dry wine with over-ripe aromas, as acceptable alternative targets (Table 2).

Table 2: Rules of inference determined for grape type ADR as the primary target and types FD and ADOR as secondary targets with both $e=50\%$

Criterion number	Measured parameter (scale or unit)	A_i	$e\%$	B_i	$e\%$	D_i	$e\%$
c_1	Berry color (/10)	$4 < c_1 < 9$	100	$2 < c_1 < 4$	50	$2 > c_1$ or $c_1 > 9$	0
c_2	Dominant aroma	$c_2 =$ ripe fruits	100	$c_2 =$ fresh fruits or $c_2 =$ cooked fruits, jam, honey	50	$c_2 =$ vegetal, or earthy/mouldy	0
c_3	sugar content (potential %alc.)	$3 \leq c_3 \leq 14,5$	100	$11 < c_3 < 13$ or $14,5 < c_3 < 16$	50	$11 > c_3$ or $c_3 > 16$	0
c_4	rot (%)	$c_4 < 10$	100			$c_4 > 10$	0

A_i = set of values of criterion i corresponding to the target quality, B_i = set of values of criterion i corresponding to a secondary target quality, considered acceptable, e_i = value of degree of correspondence to the initial target quality, D_i = set of values of criterion i considered unacceptable

2-3 Quality criteria measurements

Based on these criteria, quality assessment was carried out on the grapes sampled in the 5 selected plots at harvest time in October 2011. The sugar content was measured on a representative sample of 200 berries. The health of the berries was visually assessed on each bunch of 40 vines. An additional representative sample of 300 berries was used for berry sensory analysis from which the berry color assessment results were extracted. Must sensory analysis results were used to determine aroma. The musts were obtained by pressing grapes harvested on the same 40 vines at the same date. The berries and musts were assessed on a 0 to 10 continuous scale for each parameter by a trained expert panel of 11 judges for the berries and 13 judges for the musts. The attributes selected as corresponding to the experts grape typology criteria - berry color, must vegetal aroma, white fruit (for fresh fruit) and prune (for cooked fruit) aromas- were found discriminating in the analysis of variance with P-values lower than 0.01.

2.4. Application of quality Functional Units to LCA results

The environmental impact for each impact category relative to the quality index, Q_g was obtained by dividing the "per ha" LCA results by Q_g (%). The mass x quality (MQ_g) FU was derived from the calculation of a mass-quality index, MQ_g , by multiplying the annual grape yield by Q_g (%). The environmental impact results, in this case, were obtained by dividing the "per ha" LCA results by MQ_g giving results per MQ_g FU.

3. Results

3.1. Quality measurement results and quality and mass x quality index calculation

The results of the sensory analysis concerning aromas showed no differences between the TMRs in Fisher's Least Significant Difference test. These results were analyzed to identify the dominant aroma, in accordance with the expert description of grape types. The five TMRs yielded grapes dominated by a fresh fruit aroma. Table 3 reports the construction of the quality index Q_g based on the results of measured quality criteria and the dominant aroma. Two levels of Q_g appear for the five TMRs (62.5 and 75) due to a difference in berry color, those of TMRs4 and 5 being more golden.

Table 3: Quality results and Quality- and Mass x Quality indexes calculation (ADR grape type target, for the 5 TMRs in 2011)

Criterion number	Parameter (scale or unit)	TMR1		TMR2		TMR3		TMR4		TMR5	
		c_{iTMR1}	$e\%$	c_{iTMR2}	$e\%$	c_{iTMR3}	$e\%$	c_{iTMR4}	$e\%$	c_{iTMR5}	$e\%$
c_1	Color (/10)	3.95	50	3.40	50	3.59	50	5.25	100	4.16	100
c_2	Dominant aroma	Fresh fruit	50	Fresh fruit	50	Fresh fruit	50	Fresh fruit	50	Fresh fruit	50
c_3	Sugar content (pot. %alc.)	12.31	50	12.03	50	11.8	50	12.28	50	12.31	50

c₄	Rot (%)	14*	100	0.8	100	5	100	2.3	100	0.6	100
Q_g	Quality index		62.5		62.5		62.5		75		75
Yield 2011	(kg/ha)		6440		5250		7500		5880		5250
MQgADR	Mass x Quality index		4025		3281		4688		4410		3938

* turning to noble rot

The TMRs can be divided into two groups with the same Q_g : TMRs 1 to 3 and TMRs 4 and 5. The mass-quality index (MQg) results from $Q_g \times$ grape yield (Table 3), TMR 2 showed the smallest MQg while TMR3, due to a very high yield, had the highest MQg.

3.2. Comparison of TMR eco-efficiency according to the three different Functional Units

The results of the comparison between the five TMRs depend on the FU chosen (Figure 2). The hierarchy was the same between TMRs 1, 3 and 5 for the 3 FUs in the 3 impact categories. The most important change between the FUs concerned TMR2's GWP 100a and FwEtoxP impacts which were 13 to 30% higher relatively to other TMRs with the MQ_g FU than with the other FUs. TMR 3 remained the most eco-efficient for GWP 100a and Res whatever the FU, TMR1 remained the least eco-efficient for Res. TMR4 remained in an average position in GWP 100a and Res impact categories, and TMR5 remained the least eco-efficient for GWP 100a.

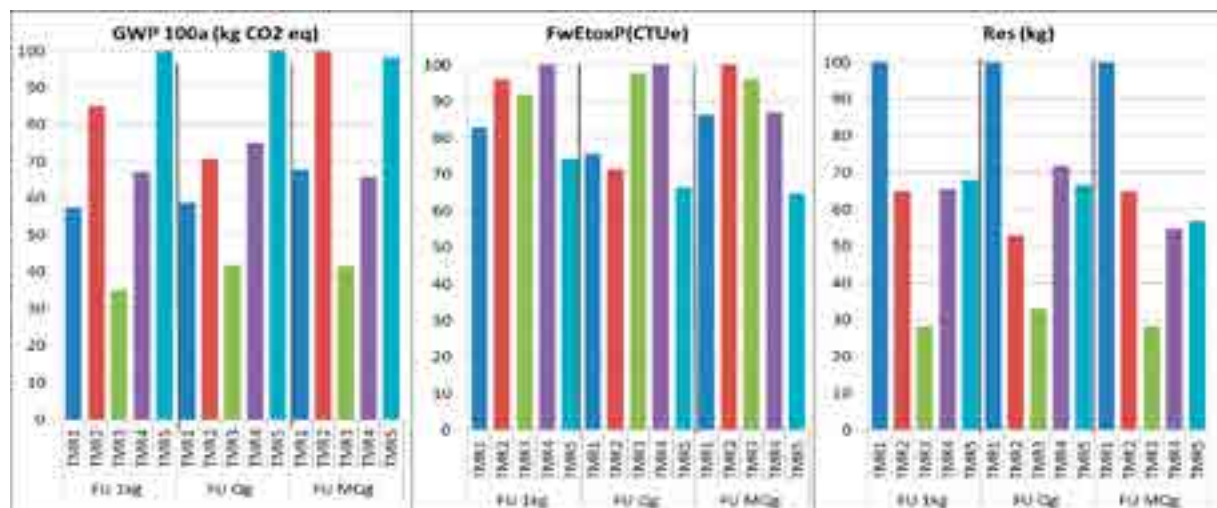


Figure 2: Eco-efficiency results of the five TMRs for GWP 100a, FwEtoxP and Res impact categories, according to three different Functional Units (FUs): 1 kg grapes, quality index: Q_g , 1 kg grapes \times Q_g : MQ_g . Results are in % of the impact of the most impacting TMR

4 Discussion

The quality index Q_g results for the five TMRs showed only minor differences mainly because of the standardization of harvest dates. Higher differences would have caused more contrast between the eco-efficiency results. The eco-efficiency results showed that the higher the Q_g , the better the eco-efficiency results in Q_g FU. This is true for the MQ_g FU but modulated by the yield. The yield had the same influence on eco-efficiency: for the same per ha impact, the higher the yield, the better the eco-efficiency for mass and mass quality FUs. Consequently the TMRs that combine high yield and high Q_g have the greatest gain in eco-efficiency when changing from 1 ha FU to MQ_g FU.

To our knowledge, the appreciation of grape quality related to a defined target has not been formalized to date in a specific indicator. This process is carried out spontaneously by the production stakeholders when they harvest or process the grapes, but the targets are often not precisely described, being more an objective fixed on an unconscious scale based on experience. This approach is generic to any grape, provided the criteria and thresholds are adapted to the cultivar and the regional, or even

local, and annual context. It can also be applied to any other product. However, this first proposal of an indicator construction might be improved in the future by the use of fuzzy logic (Zadeh, 1965), to avoid the threshold effects, and enable a gradual progression of “e” from the primary target to secondary ones and refused grapes (Coulon-Leroy *et al.*, 2012; Guillaume and Charnomordic, 2012).

Between $e = 100\%$, perfect correspondence to the primary target, and $e = 0\%$, off target, we fixed the value $e = 50\%$ for a grape corresponding to an acceptable secondary target. This threshold could be adapted, for generic situations, “e” can be determined with the experts who contribute to the determination of grape types and criteria. For specific studies, “e” should be adapted with the final user, because it can be dependent on the markets of the wine estate which determine the difference of commercial value between the primary and secondary target grapes or the corresponding wines.

The use of quality FUs in the assessment of TMR environmental performance improvement has the advantage that the advisor or the decision maker keeps in mind the quality objective of the production. These FUs are more appropriate than mass and surface FUs when considering the central function of grape production, especially in a premium wine production context like PDOs. The Q_g FU reflects only a quality level, without any reference to the yield. Using this FU reflects that the grape production exclusive -or very primary- objective is quality, whatever the yield. This can be the case in some specific situations (ultra-premium quality wines, or high quality oriented part of a vineyard being only a part of the income of the farm for example). However, in most situations, both quality and yield are needed to secure the income from the vineyard activity and to satisfy the markets in quantity. The second FU, (MQ_g), mixing mass and quality, permits to account for both objectives. According to (Heller *et al.*, 2013), this type of FU is well suited to comparing agricultural production methods. Table reports the main aspects we propose to account for in the choice of FU in quality viticulture.

Table 4: Advantages of four Functional Units for quality grape production TMRs

Functional unit	Advantage/usage
Surface area : 1ha of vineyard	minimizes impacts when cultivating a given surface area, accounts for multi-functionality of viticulture (landscape, ecosystem services), adapted to communication of LCA results to winegrowers
Mass : 1kg of grapes	minimizes impacts of a mass of grapes, considers the economic importance of yield, adapted to communication of LCA results to consumers
Mass with a quality level: 1kg grape x Q_g	minimizes the impacts of a mass of grapes, considers the central function of quality wine TMRs, avoids decreasing the quality when improving environmental performance,
Quality level : Q_g	avoids decreasing the quality when improving environmental performance

However, surface-based FUs are complementary to quality FUs to account for multi-functionality of viticulture, and also to communicate the results to the producers in a unit that meets their usual technical decision unit, 1 ha. These quality based FUs cannot totally replace per ha FU for another reason, the variability of grape composition due to climatic conditions (Jones and Davis, 2000). A climatic accident (like heavy rain before harvest) can cause a severe decrease of yield and grape quality which cannot be attributed to the TMR. In this last case, yield and surface FUs will be more reliable than MQ_g FU. Moreover, yield also varies for climatic reasons (Makra *et al.*, 2009), so MQ_g FU cumulates two sources of variations linked to climatic conditions which can be different climatic events for yield and grape quality. The climate of the year is an important variation factor in different aspects accounted for in grape LCA (Vázquez-Rowe *et al.*, 2012; Renaud-Gentié *et al.*, 2014), and the TMR itself is adapted to the climatic conditions by the growers. Accordingly, before planning important changes in the TMR on the basis of eco-efficiency, these results must be considered in the climatic context of the year in which they were obtained and LCA must be conducted on several climatically contrasting years unless a way is found to simulate the effects of

climate on these different parameters. Finally, one may also consider conducting the LCA separately from the quality assessment and combine both assessments afterwards (Beauchet *et al.*, 2014).

The effects of yield variability on LCA results may also be quantified through a sensitivity analysis of LCA results per ha, exploring the range of the known variability of yields instead of using the mass FU.

5. Conclusion

We have proposed a new grape quality assessment approach for inclusion in the eco-efficiency assessment of quality vineyard technical management routes. The quality indicator Qg expresses the degree of correspondence of the harvested grapes to the quality target assigned to the TMR. A typology of grapes was established with experts as a basis for this Qg indicator. The five contrasting vineyard TMRs, representing the middle Loire Valley diversity gave different quality-based eco-efficiency performances close to those obtained with classical FUs (1 kg of grapes and 1 ha of vines per year) due to minor differences in Qg.

Two functional units for life cycle assessment of TMRs were derived from this indicator. A quality FU: Qg, and a mass x quality FU: MQg including the yield. The QgFU alone appeared too restrictive while including the yield in this quality FU accounted for the main function of the system, i.e. the production of a given quantity of quality grapes for a given type of wine. Even though PDO wines do not respond to industrial quality standards, the wine growers and winemakers have a quality target in mind which is adjusted every year to the quality potential given by the vintage conditions. This adjustment of the quality target can be made whenever necessary. However including quality in TMR evolution or eco-conception demands further work that may involve the use of fuzzy logic in the indicator construction to avoid threshold effects and for grape quality prediction knowing the TMR.

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133. Using Acoustic Diversity in Life Cycle Assessment of Agriculture:

Case Studies of Oil Palm Production in Indonesia

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ABSTRACT

Recent important accomplishments in the assessment of biodiversity impact due to land use and land use change within the framework of life cycle assessment (LCA) include the publication of global characterization factors (CFs) and the establishment of procedures to use expert knowledge. However, the CFs are too rough to be used in management decisions and the use of expert knowledge, which can be considered as a method to resolve the problems with using the CFs, has some difficulties in dealing with the subjective information. In this paper, we appraise the possibility of constructing physical biodiversity indicators for LCA of agriculture using rapid acoustic assessment. In order to measure biodiversity in different land use (palm oil plantations, enclave forests, and natural forests), we conducted field recording and digital information analysis; the Acoustic Diversity Index (ADI) was utilized for quantifying each sound file. Case studies were carried out in South Sumatra and West Kalimantan. In average, the ADIs for plantations were lower than those for the forests, although the results were dependent on recording locations and dates. In addition, we realized that ADIs at dawn and dusk are important in assessing diversity and that paying attention to periodicity or rhythm of nature is crucial. Biodiversity assessment using acoustics is expected to establish the relationships between management practices and biodiversity, which will be useful in comparative LCA of agricultural production systems.

Keywords: biodiversity, Acoustic Diversity Index (ADI), plantation, forest, land use

1. Introduction

Global characterization factors (CFs) have recently published for assessing biodiversity impact due to land use and land use change within the framework of life cycle assessment (LCA). For example, de Baan et al. (2013) presented an approach to measure the potential regional extinction of nonendemic species caused by land occupation and transformation impacts within each WWF ecoregion and allocated the total damage to different types of land use per ecoregion. Furthermore, Chaudhary et al. (2016) calculated vulnerability scores for each ecoregion in order to take endemism into account, in addition to the quantification of regional species loss due to land occupation and transformation.

However, if we use the CFs to compare biodiversity impacts of several agricultural production systems in the same ecoregion, the impact scores per product unit are dependent only on crop yields. The reason is that the CFs were prepared for each ecoregion and for each land use type (e.g., agriculture, pasture, urban, and managed forest) as explained in Michelsen et al. (2014). It implies that the CFs are too rough to be used in assessing management practices and making management decisions. Furthermore, an environmental labelling policy using the CFs is not incentive compatible, because farm managers can improve indicator values (biodiversity scores per product unit) by increasing crop yields rather than practicing environmentally-friendly management.

Methodological development to use expert knowledge is another important achievement in the assessment of biodiversity. For example, Jeanneret et al. (2014) proposed a method to assess agricultural production systems, in which biodiversity scores are calculated as the products of the impacts (subjective rating) of management options on indicator-species groups, and habitat and management coefficients (subjective rating) for indicator-species groups. Fehrenbach et al. (2015) developed a method using the concept of hemeroby (naturalness), which has the hierarchical evaluation structure of criteria and metrics. Categorical scores are given to each metrics and overall average values are calculated. Lindner et al. (2014) proposed a combined use of geographical information and subjective evaluation. In their method, biodiversity impact is defined as the product of the ecoregion factor and the additive reciprocal of biodiversity potential, the average value of single attribute value functions that explain the relations of management intensity on biodiversity.

In essence, these methods based on subjective judgements can be classified as a constructive approach based on multi-criteria assessment (Beinat, 1997) and the methods mentioned above result in the development of constructed scales under the situation where natural measures do not exist (Keeney, 1992). Therefore, although these methods can resolve the problems with the use of the global CFs in agricultural production systems, they are based on human preferences and do not

provide physical facts. The use of preferences does not imply that the methods are useless. Rather, it means that the methods should be appropriate for prescriptive purposes in decision support contexts.

In this paper, we propose another way. We appraise the possibility of constructing physical biodiversity indicators, instead of developing indicators using subjective judgments. In this case, our intention is to develop an approach that makes comparative LCA of agricultural production systems possible, as in the case of the latter methods, rather than estimating environmental impacts caused by land-use categorical changes at the global scale. Our strategy is to construct physical indicators using labour-saving techniques and, therefore, we employed rapid acoustic assessment (Sueur, 2008), which is a digital information analysis without identifying flora and fauna based on simple field recording.

2. Methods

2.1. Assessment framework

We use the general framework for land use change impact assessment within LCA (Figure 1), which distinguishes between land transformation and land occupation by depicting a three-dimensional figure with time, land quality, and area. Our purpose in this study is to appraise the possibility to measure the quality using acoustics.

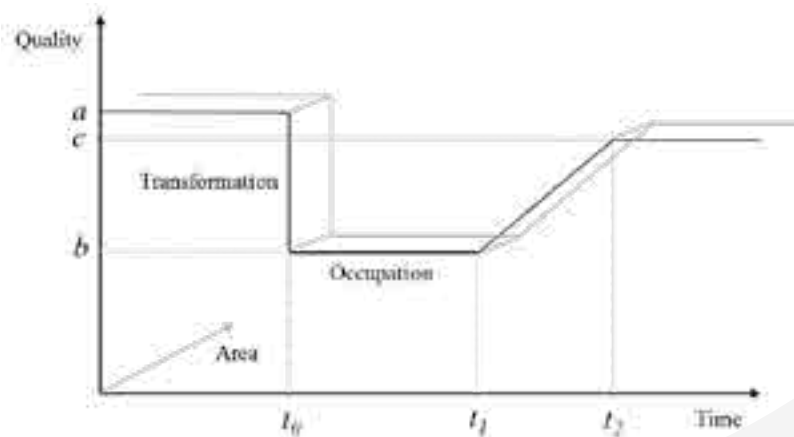


Figure 1: Simplified illustration of land use change impact assessment framework within LCA

Production systems analysed in this study using the above framework are oil palm production systems in Indonesia. Oil palm plantations between the time t_0 and t_1 (quality b) are compared with forests (quality a), which can be considered as a reference. Although pristine forests are precisely equivalent to the reference situation (quality a), we used natural forests near the plantations and enclave forests (forests within plantations), because the recording before transforming the land into plantations is not feasible and it was actually difficult to find pristine forests near the plantations analysed in this study.

2.2. Data collection

Sound data were collected in the Dawas plantation, South Sumatra and the Ngabang plantation, West Kalimantan. In the latter case, the enclave (secondary) forest in the Parindu plantation was also included. We made recordings on August 2015 using three digital recorders (SONY PCM-D100). Since natural sounds exceed the limit of the conventional CD quality (sampling frequency: 44.1 kHz, quantifying bit number: 16 bit), as well as human audible bandwidth (until about 20 kHz), the recorders were set at the mode of 192 kHz and 24 bit. Although it was difficult to record the sounds until 96 (192 divide by 2) kHz, because of the performance of the built-in microphones, we confirmed that the frequencies until at least about 50 kHz were observable. We recorded stereo sounds at the microphone angle of 120° , which are equivalent to recordings at two adjacent places. The recorders

were held horizontally at 1.5 m height. The recording locations are shown in Table 1. Outdoor loudspeakers in the plantations were not used during the recording.

Table 1: Locations for recording

Location	Dawas		Ngabang-Parindu	
	1st trial (Aug. 10-11)	2nd trial (Aug. 12-13)	1st trial (Aug. 19-20)	2nd trial (Aug. 20-21)
Plantation	2° 31' 59.31" S 103° 48' 12.39" E	2° 31' 0.96" S 103° 48' 46.99" E	0° 22' 12.47" N 109° 54' 31.71" E	0° 21' 19.72" N 109° 54' 31.32" E
Enclave forest	2° 31' 36.33" S 103° 48' 17.54" E	2° 31' 34.28" S 103° 48' 16.43" E	0° 15' 35.36" N 110° 18' 46.46" E	0° 15' 35.36" N 110° 18' 46.46" E
Natural forest	2° 30' 59.10" S 103° 48' 25.53" E	2° 30' 52.89" S 103° 48' 24.82" E	0° 22' 6.65" N 109° 47' 27.86" E	0° 22' 6.65" N 109° 47' 27.86" E

The actual land use history for the plantations are as follows. The Dawas plantation was transformed from secondary and rubber community forests in 2006, while the Ngabang and Parindu plantation was transformed from primary forests in 1980. The first generation of oil palm trees in the Ngabang plantation was planted in 1982 and 1983. After some unproductive phases, oil palm trees were replanted in 2005, 2009, 2010, 2011, and 2012. In the recorded location, the trees were replanted in 2005. The first generation in the Parindu plantation was planted in about 1980.

In order to synchronize the time information in the recorded files among the three recorders, the identical sounds were recorded in the three recorders. Using the sounds, the time stamps in the recorded files were adjusted in milliseconds. Each recorded data file (2 GB, the maximum size of sound files), which lasts about 30 minutes, was separated into one-minute files (30 pieces). The pieces that contain noises such as human voices during device setting were not used in the analysis.

2.3. Acoustic index to measure biodiversity

The Acoustic Diversity Index (ADI) (Villanueva-Rivera et al., 2011) was utilized for quantifying each sound file. In this index, the spectrogram of an audio file was divided into many frequency bands and the proportion of sound in each frequency band, which represent a specific “species”, was used to calculate the Shannon index. We used the package ‘soundecology’ in the statistical software *R* for calculate the index. Although the default maximum frequency in calculating the index was 10 kHz, we used the value of 50 kHz, because the sounds until 50 kHz were actually recorded in the files.

3. Results

We calculated average ADI values for day and night separately, because there were differences in the ADI values during the day and those at night (Table 2). The time for sunrise and sunset was calculated using the latitude, longitude, and elevation at the recording point. In Dawas, the results were different from what we expected. Although the ADI value during the day for natural forest was higher than that for plantation in the 1st trial, it is not applicable to the other cases. In contrast, anticipated results were available in Ngabang-Parindu. The descending order of the ADI values was natural forest, enclave forest, and plantation in many cases.

Table 2: Average ADI values at day and night for each location

Location	Dawas				Ngabang-Parindu			
	1st trial		2nd trial		1st trial		2nd trial	
Day/night	Day	Night	Day	Night	Day	Night	Day	Night
Plantation	2.464	3.183	2.785	3.144	2.289	2.456	2.831	2.220
Enclave forest	2.763	2.529	2.712	2.575	3.199	2.885	3.257	2.810
Natural forest	3.068	3.136	2.742	2.839	3.068	2.976	2.953	3.049

Since we recognized that the ADI can identify the higher activities of animals during dawn and dusk, we calculated average ADI values for dawn, defined as two hours after sunrise, and dusk, defined as two hours after sunset. In this case, the ADI values for natural forest were higher than those

for plantation in most cases. The differences in the ADI values between day/night and dawn/dusk were larger for natural forest than those for plantation. It implies that part of natural rhythm was disappeared in plantations.

Table 3: Average ADI values at dawn and dusk for each location

Location	Dawas				Ngabang-Parindu			
	1st trial		2nd trial		1st trial		2nd trial	
Dawn/dusk	Dawn	Dusk	Dawn	Dusk	Dawn	Dusk	Dawn	Dusk
Date	Aug. 11	Aug. 11	Aug. 13	Aug. 12	Aug. 20	Aug. 19	Aug. 21	Aug. 21
Plantation	2.961	3.435	2.904	3.401	2.397	1.946	2.752	3.156
Enclave forest	2.858	3.422	2.717	3.799	2.747	3.522	2.879	3.507
Natural forest	3.432	3.312	3.423	3.570	2.838	3.730	3.030	3.636

In addition, the results suggest that the differences between the two types of forests were smaller than those between the plantations and forests. It means that the both forests have a generality as forests and that the differences in the quality explained in Figure 1 between plantations and forests were detected, although there were exceptions.

4. Discussion

We demonstrated the possibility of acoustics in biodiversity appraisal within the framework of LCA, although further studies are necessary for the use of acoustic diversity in, for example, environmental labelling policies based on LCA. We will discuss how the approach using acoustic diversity can be further developed.

4.1. Definition of the reference state

One of the reasons why the ADI values for the forests were in some cases lower than those for plantations may be attributed to the degraded condition of natural forests surrounding oil palm plantations. Although we recorded the sounds in the two types of forests (natural and enclave forests), because we thought that the acoustic diversity in forests are dependent on locations and because it was difficult to find the pristine forests near the plantations, we have to be explicit about the difference between natural vegetation and potential natural vegetation in biodiversity assessment of land use (Souza et al., 2015). The both types of forests in this study should be recognized as the latter vegetation. Since most natural forests are used by local people for hunting animals and for logging precious woods, further investigation is necessary for understanding the relationship between land use intensity (forest management) and biodiversity.

4.2. Selection of acoustic indices

Another reason for the lower value cases can be related to the definition of acoustic indices. Because of the difficulty in finding a single index summarizing all biodiversity information, Sueur et al. (2014) recommend the complementary use of several α -diversity indices. In this study, we complementarily applied the Acoustic Complexity Index (ACI) (Pieretti et al., 2011). In average, the ACIs for plantations were larger than those for natural forest, although we did not judge that it implies biodiversity in plantations is richer than that in natural forests. The important differences were found in scale parameters (e.g. standard deviations), rather than location parameters, at dawn and dusk, although this trend is also applicable to the case of ADIs. That is, in the morning and evening, standard deviations for the natural forest were larger than those for the plantation. Further research on acoustic indices is necessary in clarifying the relationship between sound states and indicator values.

4.3. Implications to inventory analysis and impact assessment

Inventory data for land use and land use change contain the information about occupation and transformation (“from” and “to”) in the section for “inputs from nature”. The results suggest that

acoustic indices can be used as a proxy for the quality of biodiversity (the vertical axis in the assessment framework in Figure 1). However, applicability of acoustic diversity is not limited to this way of inventory construction.

One is the description as “outputs to nature”. In this case, understanding of multifunctionality plays an important role and further consideration on ecosystem services is necessary. Another way is the establishment of impact assessment based on the relationships between management practices (information such as fertilizer and pesticide application in the section for “inputs from technosphere”) and biodiversity.

5. Conclusions

The results of this study indicate that acoustic diversity can be applicable to biodiversity assessment within the framework of LCA of land use and land use change and that attention has to be paid to periodicity or rhythm of nature. Although this research direction will be important in establishing the relationships between management practices and biodiversity on the basis of site-specific conditions, there may be difficulty in applying to the locations where biodiversity is sparse. For example, crop production systems in temperate regions might be difficult to use acoustic indices, although acoustic analysis can be useful in, for example, identifying animal species.

An important implication of this study is that in assessing biodiversity, the concept of landscape and soundscape becomes crucial. Although LCA has already been used at the regional levels, agricultural provision of ecosystem services at the landscape/soundscape level can be embedded in LCA.

Although rapid biodiversity appraisal using acoustics is promising in the sense that it can gather site-specific biodiversity data without using much time and cost, there are limitations due to acoustics; the recorded data are limited to sounds produced by animals when moving, communicating, and sensing their environment. Further justification of methodology through the identification of animal species in the recorded sites and the integration with the other sensing technologies will be important research topics.

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ABSTRACT

Nutritional health effects from the ‘use stage’ of the life cycle of food products can be substantial, especially for fruits and vegetables. To assess potential one-serving increases in fruit and vegetable consumption in Europe, we employ the Combined Nutritional and Environmental LCA (CONE-LCA) framework that compares environmental and nutritional effects of foods in a common end-point metric, Disability Adjusted Life Years (DALY). In the assessment, environmental health impact categories include greenhouse gases, particulate matter (PM), and pesticide residues on fruits and vegetables, while for nutrition we consider all health outcomes associated with fruit and vegetable consumption based on epidemiological studies from the global burden of disease (GBD). Findings suggest that one fruit/vegetable serving increase may lead to substantial nutritional health benefits even when considering uncertainty; 35 μ DALY/serving_{fruit} benefit compared to a factor 10 lower impact. Replacing detrimental foods, such as trans-fat and red meat, with fruits or vegetables further enhances health benefit. This study illustrates the importance of considering nutritional effects in food-LCA.

Keywords: LCA, fruits, vegetables, nutrition, human health

1. Introduction

Dietary risks are leading the global burden of disease (GBD) with about 12 million annual attributable deaths globally, illustrating the strong relationship between dietary patterns and human health (IHME, 2015). Diet- and food-related life cycle assessments (LCA), up to date, mainly focus on human health impacts associated with environmental emissions. The ‘use stage’ of food products, although part of a product’s life cycle, does not typically consider nutritional effects that occur with consumption and can have substantial effects, positive and/or negative, on human health (Stylianou et al. 2016). Incorporating a nutritional assessment in diet- and food-related LCA would provide a comprehensive and comparable human health effect evaluation of food items and diets that could yield more sustainable dietary decisions.

The nutritional value and beneficial human health effects associated with fruits and vegetables consumption is widely recognized and evident by numerous recommendations urging consumers to increase their fruit and vegetable daily intake (USDHHS and USDA 2015; Nordic Council 2014). However, current conventional fruit and vegetable production methods require the application of pesticides which yields residues that have the potential of inducing human health impacts, a continuous concern requiring a constant monitoring and evaluation. As a result, increased consumption of fruits and vegetables – although considered as a healthier dietary option – could result in higher exposures to a wide variety of pesticides, alongside other environmental health related impacts associated with corresponding increase in production and distribution. The aim of this study is to assess the overall human health trade-offs between potential environmental and nutritional effects associated with one serving increase of fruits (141 g) and one serving increase of vegetables (123 g) over the average European consumption.

2. Methods

2.1. Framework for comparing environmental and nutritional effects of food

The Combined Nutritional and Environmental Life Cycle Assessment (CONE-LCA) framework evaluates and compares in conjunction environmental and nutritional effects of food items or diets expressed in a common end-point metric, Disability Adjusted Life Years (DALYs) (Stylianou et al. 2016). In this case study, the assessment starts from one serving of fruits and one serving of vegetables as a functional unit (FU) that are associated with environmental health impacts due to life cycle emissions of e.g. greenhouse gases (GHG) and particulate matter (PM) as well as chemical intake from pesticide residues on vegetal food. Nutritional impacts and benefits are assessed in parallel based on published epidemiology data that directly link fruit and vegetable consumption to nutritional health outcomes such as cardiovascular diseases and neoplasms, starting from the GBD.

Figure 1 illustrates the general CONE-LCA framework along with the framework used in the case study investigated in this paper (represented in the red dashed box).

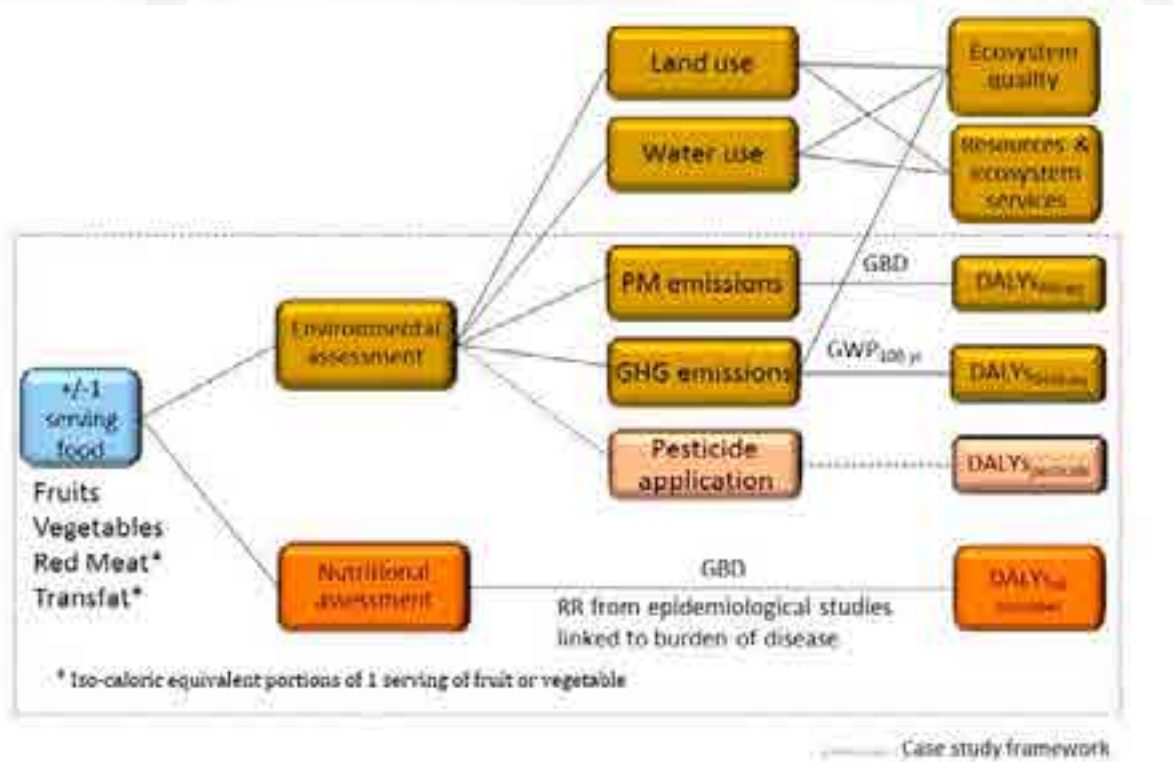


Figure 1. Graphical representation of the CONE-LCA framework. The dashed box denotes the scope of the presented case study

2.2. Case study: fruits and vegetables consumption in Europe

2.2.1. Dietary scenarios

The European Food Safety Authority (EFSA) Comprehensive European Food Consumption Database (EFSA, 2015) reports the average adult European diet. According to the latest data the current population-weighted European daily diet is consisted on average from 195 g of total fruits (fresh and processed) and 218 g of total vegetables (fresh and processed). These intakes correspond to a 1.2 and 1.8 servings of fruits and vegetables daily intake, respectively, which are below the dietary recommendation guidelines (USDHHS and USDA 2015; Nordic Council 2014).

To assess a potential dietary shift towards dietary guidelines, we investigate the case of one serving increase of fruits (141 g) and one serving increase of vegetables (123 g) over the average European consumption and evaluate the corresponding health effects. To consider for more realistic dietary scenario assessment, in addition to the increase in fruit or vegetable consumption, we also evaluate two substitution scenarios based on a default iso-caloric equivalent basis as a first proxy of a) trans-fat and b) red meat, two high burden dietary risk factors in the GBD (IHME, 2015).

One serving of fruits or vegetables has a nutritional energy content of respectively 102 or 74 calories, respectively. Hence, we investigated the following per person daily dietary scenarios:

- A. Add a serving of fruits (or vegetables), with no change to the rest of the diet.
- B. Add a serving of fruits (or vegetables) while subtracting an equal caloric quantity of trans-fat.
- C. Add a serving of fruits (or vegetables) while subtracting an equal caloric quantity of red meat.

2.2.2. Environmental assessment

The environmental assessment in our analysis follows a traditional LCA approach. Food group-specific emission factors for GHG and ammonia (NH₃) were retrieved from the work by Meier and Christen (2012), accounting for production, processing, packaging and transportation to retail. Other PM-related emission (primary PM_{2.5}, NO_x, SO₂) were extrapolated from GHG as described by Stylianou et al. (2016) since such information is not routinely reported in food LCA studies. Emissions are coupled with characterization factors (CF) to give human health impact in DALYs/FU. More specifically, CFs from Gronlund et al., (2015) and Bulle et al., (manuscript in preparation) were used for PM-related and a 100-year horizon global warming health impacts, respectively.

In regards to the pesticide residue exposure, human health impacts are determined based on the work by Fantke et al. (2012). Human health impacts have been quantified by crop class accounting for human exposure resulting from 133 pesticides applied in 24 European countries in 2003 and individual substances distinct environmental behavior and toxicity. Active ingredients found in pesticides were then associated to publically available consumption data. Adjusting for current European fruits and vegetables consumption (EFSA, 2015) and under a linear assumption, the human health impact estimate from pesticide residues on fruits and vegetables is 2.5×10^{-7} and 2.4×10^{-6} DALYs/year/person, respectively.

2.2.3. Nutritional assessment

For the nutritional assessment, there are numerous epidemiological studies investigating the association of fruit, vegetable, trans-fat, and red meat intake with various health outcomes. In our study, we focus on the various health outcomes considered in the GBD for each of these dietary risk factors. More specifically, cardiovascular diseases are the main health outcome associated with low fruit (86%), low vegetable (100%), and high trans-fat (100%) consumption while high red meat consumption is associated with diabetes (60%) and colorectal cancer (40%). We combine the total European burden reported by the GBD for each food group (IHME, 2015) with the corresponding current consumption (EFSA, 2015) to estimate the overall nutritional health effect, benefit or impact, per FU, accounting for the respective theoretical minimum risk intake (as defined by the GBD in the work by Forouzanfar et al., 2015).

3. Results

3.1. Environmental assessment: PM-related health impacts

Figure 1 illustrates the PM-related human health impact in μ DALY/serving corresponding to the iso-caloric food portions. Our analysis indicates that one serving of fruits is linked to a total of 0.065 g PM_{2.5}-eq, corresponding to a health impact of 0.08 μ DALY, mainly due to NH₃ (38%). The iso-caloric red meat equivalent is associated with substantially higher health impact (about 7 times), with NH₃ as the main PM-precursor contributor at 85%. For the vegetable serving we estimate PM-related health impact of 0.03 μ DALY/serving, again mainly attributable to NH₃ emissions (40%). The iso-caloric red meat equivalent had 6.5 times higher impact than a serving vegetable. The PM-related health impacts for the trans-fat substitutions are considered negligible.

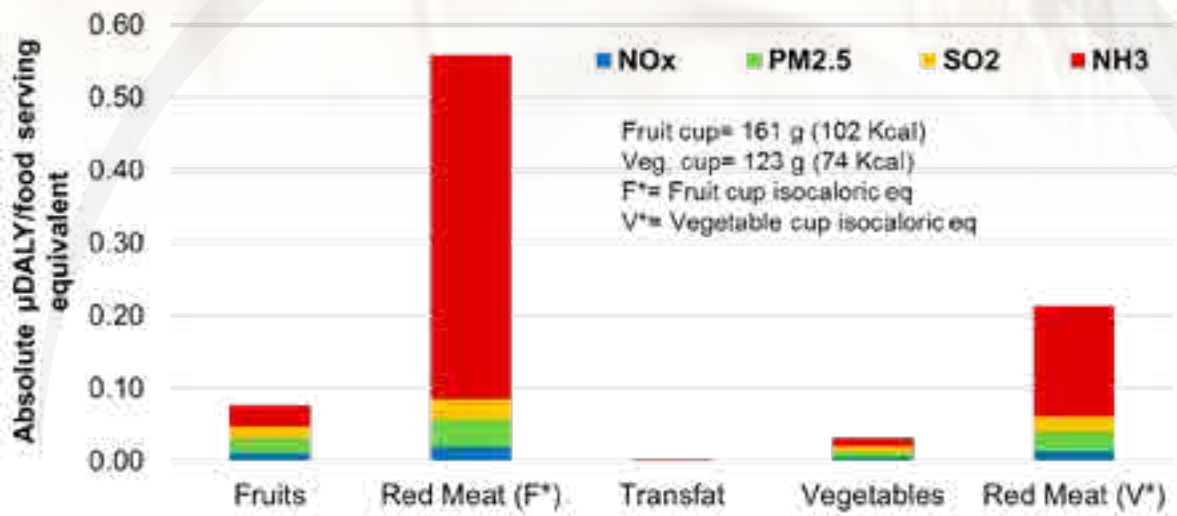


Figure 2. Particulate matter related human health impact measured in associated μ DALY/serving with an iso-caloric equivalent portion of distinct food intakes: (1) fruits, (2) vegetables, (3) trans-fat, (4) red

3.2. Nutritional assessment

A linear dose–response relationship relates food intake, expressed in g/person/day to all cause outcomes impact in DALYs/person/day. We use such dose–response functions to estimate the nutritional health burden attributable to food intake shift from the current consumption. For fruit consumption, we found that one serving increase in intake over current consumption would results in a benefit of 34.7 μ DALY (Figure 2). The analogous estimate for one additional serving of vegetable is a benefit of 17.2 μ DALY. Using the same approach for the considered substitutions, the fruit (or vegetable) iso-caloric reduction in trans-fat and red meat portion is associated with reductions in health impacts of 0.5 (0.4) and 1.5 (1.1) μ DALY, respectively.

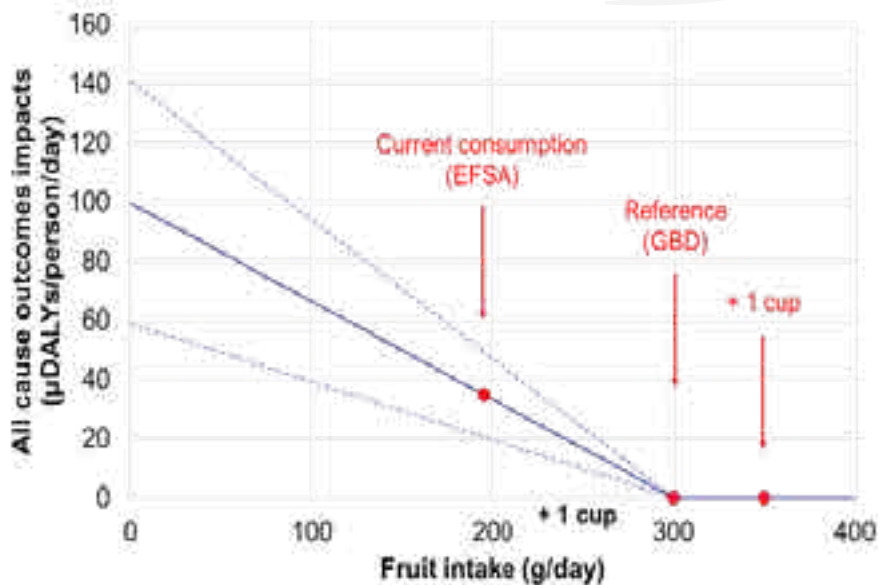


Figure 3. Dose–response function for fruit intake and all cause outcomes, with 95 % confidence intervals shown as dashed lines.

3.3. Overall comparison

Figure 3 represents the overall environmental and nutritional human health trade-offs associated with one serving of fruits without and with substitution scenarios. Adding one serving of fruits to the present European diet may lead to a considerable nutritional health benefits ($35 \mu\text{DALY}/\text{serv}_{\text{fruit}}$). The nutritional benefit is moderately enlarged when we consider the substitution scenarios since the substituted food items are associated with negative health effects and reduction in intake results in avoided human health impact. Overall environmental health impacts are substantially smaller, about an order of magnitude lower, compared to the nutritional benefits in each scenario. Benefits exceed impacts even when considering an uncertainty factor of 400 for the impacts of pesticide residues. Similar results are found for the case of adding one serving of vegetables to the average diet.

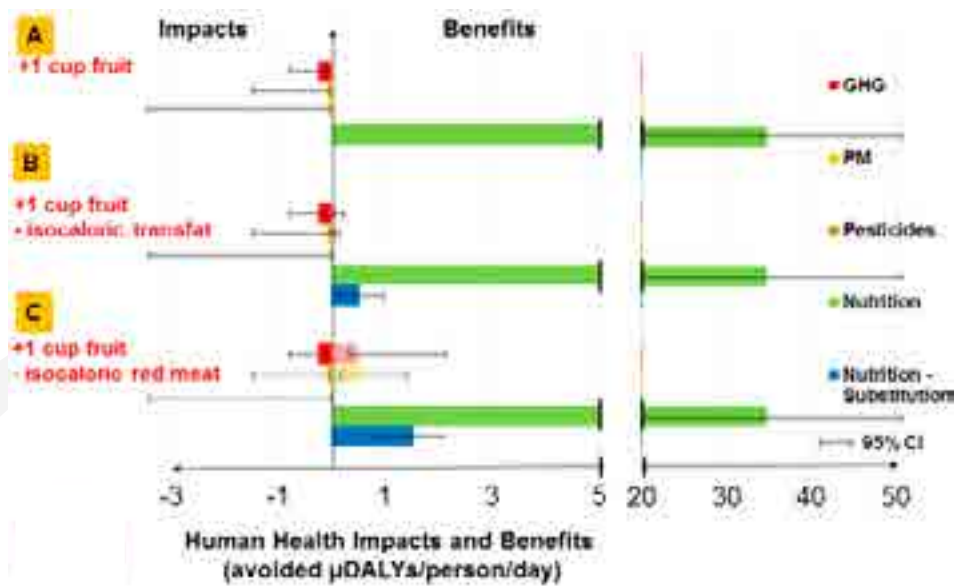


Figure 4. Comparison of daily environmental impacts and nutritional effects for a potential one serving fruit increase consumption, accounting for iso-caloric substitutions of trans-fat or red meat.

4. Discussion

In this paper, we use the CONE-LCA framework that enables a comparison between environmental and nutritional human health effects in a common end-point metric within a LCA context. In addition to the traditional environmental mid-point categories that are linked to human health impacts in LCA (GHG, PM), we also consider pesticide exposure in this case study since we are investigating consumption of fruits and vegetables. Although we limited our analysis to only three relevant environmental impact categories contributing to human health, it should be emphasized that the CONE-LCA framework can be extended to other human health-related environmental impact categories.

Specific to this case study, nutritional human health benefits associated with the addition of one serving of fruits or vegetables to the current European diet exceeded by far the corresponding environmental impacts in all three dietary scenarios. In scenarios B and C, where we considered potential substitution from trans-fat and red meat using an iso-caloric basis as a first proxy, the nutritional benefit was further reinforced due to avoided health impacts related to reductions of harmful food items. We acknowledge that such substitution choices come with limitations in terms of

scenario comparison and results interpretation. However, under the as assumption of an increase in healthy dietary choice consumption such as fruits and vegetables, an ideal substitution would occur from unhealthy food products such as trans-fat and red meat. We acknowledge that the trans-fat reduction as suggested in scenario B could be considerably hard to implement in practice. Although the content of trans-fat in food products has reduced and started to be labelled in food packaging (nutrition facts label), it still remains difficult to actually monitor and reduce daily intake due to the number of food items that contain trans-fat. Specific to our case study, the trans-fat substitution with fruits would require a reduction of 11.3 grams of trans-fat that could be achieved, for example, by removing 1.4 pieces of chocolate icing doughnut or 4 table spoons of margarine in stick form from the daily diet. To identify and assess realistic scenarios, substitutions should ideally build on detailed market-based and consumption-based surveys.

Finally, it should be mentioned that these are initial findings that depend on toxicological studies for the pesticide residue assessment and on epidemiological studies for the nutritional assessment. In addition, our findings are highly dependent on the quality and uncertainty of the data used. Hence, our findings should be interpreted within the context of this study and with caution. In the future our study aims to also consider epidemiological data that associate pesticide exposure to human health so that human health effects are assessed in a consistent manner with nutritional effects.

5. Conclusions

The present CONE-LCA framework enables us to compare in conjunction environmental impacts and nutritional effects on human health using a common end-point metric. The preliminary results of this case study indicate that nutritional health effects of food items, and specifically of fruits and vegetables, during the ‘use stage’ can be substantial and exceed by far any potential environmental impacts. In addition, our results emphasize the importance of affordability and accessibility to fruits and vegetables for the general public.

6. Acknowledgments

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ORAL PAPERS

DAIRY PRODUCTION

2. Dairy Production

112. Allocation choices strongly affect technology evaluation in dairy processing

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ABSTRACT

This paper illustrates how different allocation approaches affect the assessment of energy and water saving technology options in the dairy processing sector. The choice between allocation on facility level or process level was evaluated, as well as the choice between dry matter and economic allocation in a case study of a mozzarella facility based on primary data. It was found that the carbon footprint allocated to the main product, mozzarella, is very sensitive to these methodological choices, because the dry matter in mozzarella is valued relatively highly, and would receive impact from the energy intensive whey processing under the facility level allocation approach. Economic allocation on the process level gives results that are most unambiguous and straightforward to interpret in the specific decision context of technology options evaluation.

Keywords: Mozzarella, Whey, Dairy, Global Warming Potential, Multifunctionality

1. Introduction

In the dairy processing sector, saving energy and water use through developing and integrating different innovations is an important priority. Such innovations are being piloted in the EnReMilk project (an EC Framework Programme 7 project) in a German dairy facility that produces among others skim milk powder, and an Italian mozzarella facility. To support decisions on the selection of innovations from the EnReMilk project in these distinct facilities, Life Cycle Assessment (LCA) is considered as the most appropriate methodology. Environmental footprinting, i.e. calculating LCA impacts for distinct products, is gaining popularity because it provides easy-to-understand impact indicators that can be added up (Ridoutt and Pfister 2013). Companies, consumers and governments often take a simple choice oriented comparison mindset in which it is hard to consider external consequences of this choice. In the current decision context, external consequences outside the dairy processing facility, caused via market mechanisms, are not expected.

Although LCA studies are generally done according to the ISO-standards 14040 and 14044 (ISO Technical Committee ISO/TC 207 2006a,b), these standards leave several methodological choices up to the practitioner, which can strongly affect the results of the assessment (among others Yan et al. 2011, Kim et al. 2013). Among these choices, the approach for dealing with multifunctionality is an important point of attention in dairy processing. Multifunctionality arises in cheese production in the curd-whey separation, milk-cream separation, the milk-beef farming system and the feed-seed oil production system (Feitz, Lundie, Dennien, Morain, & Jones, 2007; Thoma, Jolliet, & Wang, 2013; Thomassen, Dalgaard, Heijungs, & de Boer, 2008; Tucker et al., 2010) and is an important methodological point of attention. How multifunctionality should be treated seems to become increasingly consistent, when considering the scientific and industrial guidance (Feitz et al. 2007, Aguirre-Villegas et al. 2012, European Dairy Association et al. 2015, IDF (International Dairy Federation) 2015).

Approaches that deal with the multifunctionality problem through allocation are relevant in the case of cheese, because avoiding allocation approaches (as recommended by ISO Technical Committee ISO/TC 207 (2006b)) cannot fully solve the problem in this case: Subdivision is not possible since the curd/whey separation reflects a chemical separation of milk; and Substitution is not possible because there is no realistic market-average alternative to producing ricotta, nor can a hypothetical alternative be considered that does not originate from the multifunctional process of curdling milk (Aguirre-Villegas et al. 2012).

In this paper, it will be evaluated in which way different allocation approaches affect the results of an LCA for a specific mozzarella producing facility. These findings will be discussed in the context of the application of the LCA, and critically reviewed in a broader context.

2. Methods

Goal and Scope: Since this paper aims to illustrate the effects of different allocation approaches, the global warming potential was used as a straight-forward and suitable indicator for the goal of this paper. Because the LCAs conducted in the EnReMilk project itself have a broader focus, these evaluate all impact categories from ReCiPe 2008 (Goedkoop et al. 2009). The functional unit has been defined as “1 kg of mozzarella for pizza applications at the gate of the mozzarella facility in the baseline year 2014”. The system boundaries of the overall LCA are from the cradle to the gate of the mozzarella facility, and all material inputs, consumables and capital goods have been included except office activities, facilities overhead and supporting services because these were estimated to contribute less than 1% of the total global warming potential.

Production description: Raw Milk is stored after delivery and pasteurized, and subsequently standardized by separating a small share of the milk fat (cream) from the raw milk. The standardized milk is curdled by addition of a bacterial starter culture and rennet in the substeps of pre-ripening, coagulation and curd cutting. The resulting (sweet) whey is drained and collected, and the curd left to ripen, from which additional (acid) whey is collected. Mozzarella is shaped from the ripened curd by a process of cutting, stretching and molding. The cylinder or ball shaped mozzarella is pre-cooled with water, cooled with ice water, and packaged for subsequent storage. Ricotta is produced through heating the sweet whey. Furthermore, cream is churned into butter. A complete picture with all main product and byproduct flows is shown in Figure 1.

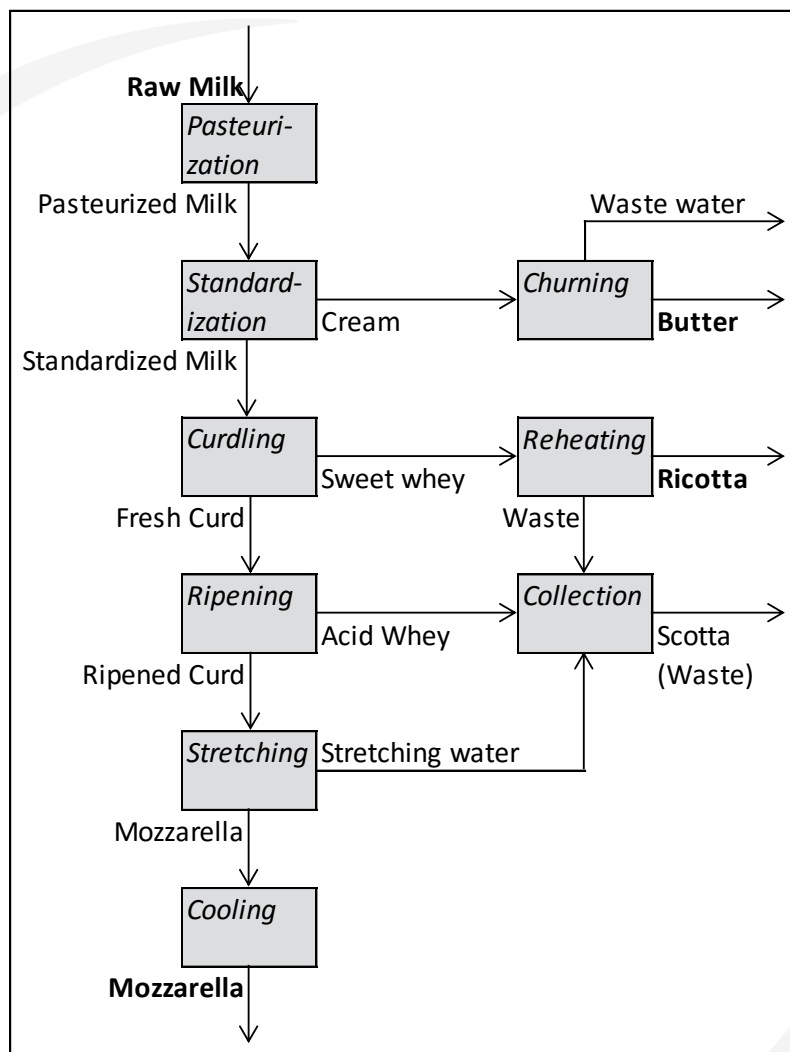


Figure 1: Overview of Mozzarella processing steps, excluding packaging

The remaining, protein-poor whey is combined with the acid whey and the waste water from mozzarella stretching. This combination of byproducts is called scotta, is sent off-site for waste treatment. Wastewater results from (pre-)cooling the mozzarella, butter production and from rinsing and cleaning equipment. Ice water cooling consumes significant amounts of electricity, whereas all other process steps consume much small amounts of electricity. Steam is consumed only during pasteurization, cleaning-in-place and mozzarella stretching, and water is consumed during pre-cooling and packaging.

Data Collection: Steam, electricity and water consumptions in the mozzarella facility were collected on the unit process level. This is possible because a monitoring system is being set up to evaluate the facility performance in different experimental technology pilots in the EnReMilk project. In addition to the baseline case of the mozzarella facility, a hypothetical scenario of moving from production of ricotta to whey powder was selected to illustrate the consequences of all four allocation approaches in an extreme case. The whey drying process was modelled using data from the skim milk powder facility in Germany, collected in the EnReMilk project. The selected facility data is representative for a typical day of production, excluding situations of intensive production or production problems. On a typical day, the mozzarella facility consumes 21 tons of milk and produces 3 tons of cow milk-based mozzarella cheese for pizza application, which is called fior di latte in Italy. As such, the facility has a limited size compared to cheese production facilities in the US (Aguirre-Villegas et al. 2012, Kim et al. 2013).

The impact of raw milk was derived from the Agri-footprint database, using economic allocation (Blonk Agri-footprint bv. 2014a, b). EcoInvent 3.2 processes (Weidema et al. 2013) were used to model the impact of electricity, steam, cleaning in place, waste water treatment and transport, augmented with specific grid mix from the International Energy Agency (IEA) (2014) and through personal interaction with the facility owner. Packaging of the final mozzarella product was included by following the draft PEFCR for Dairy (European Dairy Association et al. 2015) and including packaging raw materials from EcoInvent 3.2. SimaPro 8.2 was used for composing the model and extracting the results (Pré Consultants 2016).

Allocation: As discussed in the introduction, different allocation approaches can be identified. Firstly, the facility can be regarded as a whole with a total resource consumption (facility level, FL) or it can be subdivided into groups of unit processes that relate to all, a subset or one of the final products (process level, PL), as illustrated in Figure 2.

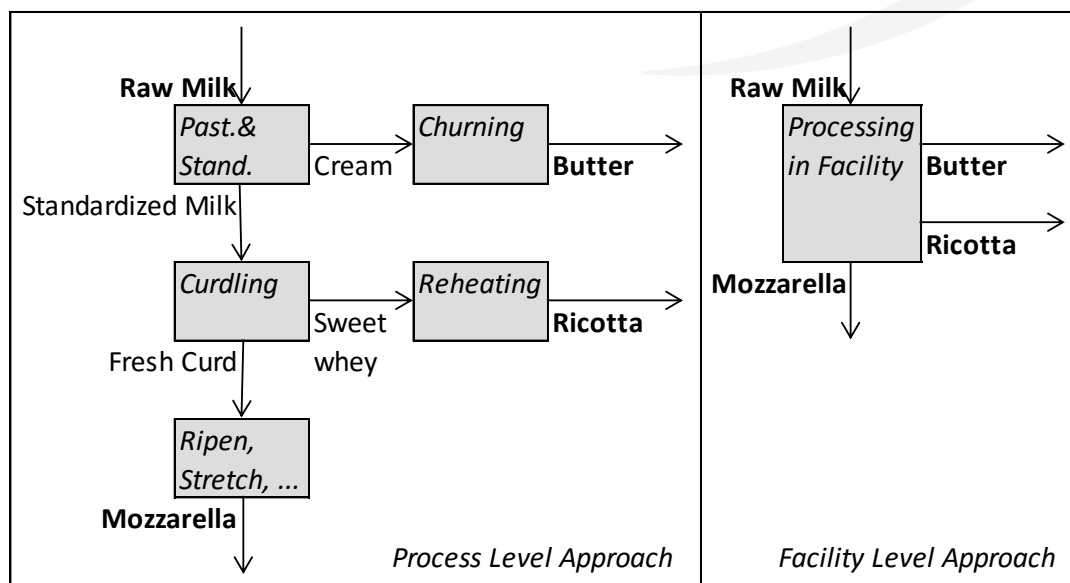


Figure 2: Two different allocation approaches for a mozzarella producing facility: on the left the process level approach, on the right the facility level approach. The distinction between dry matter allocation and economic allocation is not shown in this figure.

In the facility level approach, all impact is allocated between mozzarella, ricotta and butter. In contrast, two allocations are done in the process level approach: the impact upstream of the standardization is allocated between standardized milk and cream, and the impact upstream of curdling is allocated between sweet whey and fresh curd. Secondly, allocation between multiple flows from a process can be done according to dry matter content of the flows (dry matter allocation, DMA) or to the revenue generated with these flows (economic allocation, EA). Dry matter content data of all products were reported by the facility owner, as well as market prices of the final products. Market prices of intermediate products were derived from prices of raw milk and of the final products. These two choices lead to four allocation approaches.

3. Results

In Figure 3 the effects of the different allocation approaches can be seen for the entire cradle-to-gate assessment. It is clear that the raw milk production has the largest contribution with 61-77%, and that transport is the secondary contribution with 15-19% for all allocation approaches, when considering the baseline case (ricotta production). Figure 3 shows that less impact is allocated to mozzarella under dry matter allocation compared to economic allocation, for both facility and process level approaches, because mozzarella has a larger share in the total revenue than in the total dry matter utilized from the milk. The process level approach leads to a lower impact compared to the FL approach under DMA, because the allocation ratio between curd and sweet whey are different from the allocation ratio between mozzarella and ricotta. This is because curd and whey still include milk solids that ultimately go to waste (scotta), and could be corrected by only including the dry matter that is not wasted in the allocation factor calculation for curd and whey.

Under the dry matter facility level approach, the change in sweet whey processing (from ricotta to whey powder production) reduces the impact of mozzarella because the milk solids utilization has increased. On the other hand, under the dry matter process level approach, the mozzarella impact stays the same, because sweet whey processing is separated in the model. Under the economic allocation approaches on both levels, the change from ricotta to whey powder increases the impact of mozzarella, because less revenue is achieved by producing whey powder compared to ricotta.

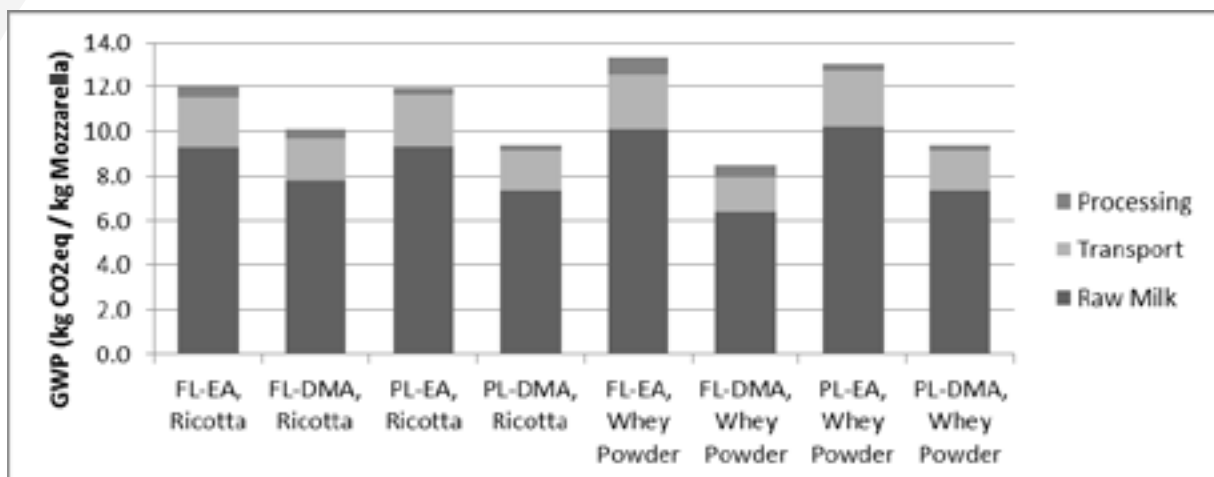


Figure 3: Cradle-to-gate carbon footprints (kg CO₂eq/kg of product) of mozzarella with contributions of raw milk, transport and processing, under different allocation approaches (FL=Facility level, PL=Process level, EA=Economic allocation, DMA=Dry matter allocation) for two scenarios: producing ricotta from sweet whey and producing whey powder from sweet whey

The contribution from the processing step is limited, compared to raw milk impacts and transport, but is affected by technological innovations within the dairy facility. For technology evaluation, it is specifically interesting how the impact of processing is distributed over the different products. As shown in Figure 4, mozzarella receives a larger share under economic allocation compared to dry matter allocation, because mozzarella has a larger share in the total revenue than in the total dry matter utilized from the milk. Mozzarella receives a much smaller share of the processing impact in

the process level approaches, because the large energy consumption in ricotta is more correctly attributed to the ricotta process, compared to the facility level approaches. The hypothetical change from ricotta to sweet whey production increases the total processing impact by 47%. Because this increase strongly affects the mozzarella contribution under the facility level approaches, the mozzarella impact is made strongly dependent on whether ricotta or whey powder is produced. The effect is most strong for economic allocation, because whey powder provides less revenue, while it increases dry matter utilization in dry matter allocation.

The effects of the trends described above translates into highly variable carbon footprints of individual products, as shown in Table 1. Mozzarella receives high impacts under facility level approaches, while whey products receive higher impacts under process level approaches, especially under dry matter allocation.

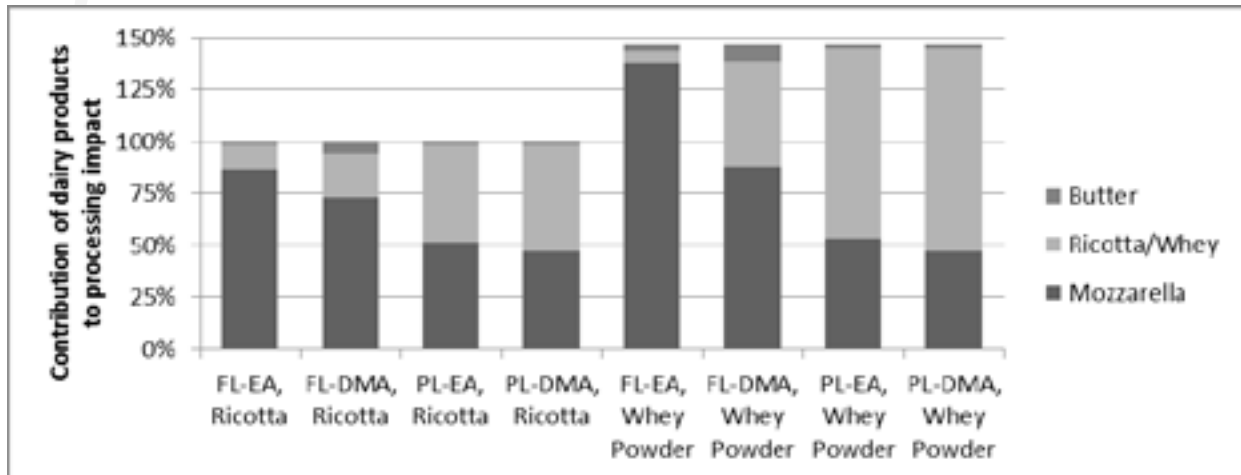


Figure 4: Percentage contributions of the products mozzarella, ricotta or whey, and butter, to the processing impact under the different allocation approaches, for the two scenarios. All data is relative to the impact of the ricotta scenario, so that the whey powder scenarios have a higher total impact.

Table 1: Carbon Footprints (kg CO₂eq/kg of product) of the processing step in the mozzarella facilities under the different allocation approaches, for the two scenarios

Allocation Approaches	Scenario: Producing Ricotta		Producing Whey Powder			
	FL	PL	FL	PL	PL	
EA	Mozzarella	0.48	0.28	Mozzarella	0.76	0.29
	Ricotta	0.15	0.60	Whey Powder	0.12	1.91
	Butter	0.25	0.20	Butter	0.39	0.20
DMA	Mozzarella	0.40	0.26	Mozzarella	0.48	0.26
	Ricotta	0.27	0.65	Whey Powder	1.07	2.04
	Butter	0.80	0.20	Butter	0.96	0.20

4. Discussion

The results show that different allocation approaches affect both the contributions upstream of the processing facility and the impact of the processing.

The process level approach correctly separates the considerations on how much energy to invest in whey processing from the mozzarella production, since a significant change in the whey processing does not affect the mozzarella production economically or physically. The facility level approach attributes some of the impact from whey processing to mozzarella. Combined with dry matter allocation this could give the perverse incentive of moving to whey powder production, in which more energy is consumed. The process level approach gives relevant information in the decision context of technology evaluation, so that the additional detail and effort could be justified (Ekvall and Finnveden 2001). Furthermore this subdivision is recommended by the ISO standard 14040.

However, the process level approach is only possible under high data availability. Although intensive contact with facility owners and technical experts is possible in the EnReMilk project, it turned out to be challenging to be completely certain of the mass and dry matter balances that were needed to achieve the highest possible reliability. It was noted before that it can be challenging to account for all resource uses on a process level (Ekvall and Finnveden 2001), and hybrid approaches have been proposed (IDF (International Dairy Federation) 2015). The impact of the same products from facilities with different product portfolios will be most comparable if these facilities use the process level approach. However, if facilities with lower data availability are not able to follow the process level approach, all facilities should use the facility level approach, because using different allocation approaches would make results even less comparable.

For technology evaluation, the process level approach is preferred, because they give better consistency for all products and comparability between different technology alternatives. In a scientific context, the unavailability of data is a bad argument to say that the data does not need to be collected, but when footprinting products from several businesses such practical considerations play a larger role. Thus, a trade-off can be recognized between, on the one hand, the benefits of internal comparability and consistency in the process level approach, and the practicality and external comparability in the facility level approach on the other hand.

The choice between dry matter and economic allocation approaches is more fundamental. While a causal relationship between the allocation property and the inputs of the multifunctional process is recommended (ISO Technical Committee ISO/TC 207 2006b), a causal relationship between the allocation property and the incentive to produce a product is also thinkable. Examples of incentives are generating revenue, nourishing people, etc., with properties like price, nutrient content and energy value. Economic allocation is criticized because prices of dairy products are variable and would introduce variability in economic allocation factors across time and regions (Feitz et al. 2007, Aguirre-Villegas et al. 2012). The variation in prices translates to variation in production incentives, which in fact should be addressed by using market-standardized price averages over several years. Dry matter allocation approaches follow the physical flows throughout the facility, and is more practical in dairy processing, because price information is not required and dry matter tracking is common in the industry (Aguirre-Villegas et al. 2012).

Considering these prior observations as well as the decision context of the dairy facility, which is essentially economic, economic allocation is preferred. The dry matter approach is not entirely consistent in this context, because it is influenced by the share of milk solids that is wasted. Since waste is produced when it is economically unattractive to turn a process flow into a valuable product, economic considerations are introduced in the dry matter approach. Furthermore, the implied causal link between the dry matter content and the environmental impact, is only valid for the raw milk impacts, but not for the processing impacts. Economic allocation is more practical in this context because prices vary less on the Italian mozzarella market than globally. Using averaged prices also matches the allocation with the time frames of decision context for technological innovations and other production changes.

For technology option evaluation, the product perspective is useful, because a producer is most rewarded by improving the impact of the main product. The total environmental impact of the product portfolio (1kg mozzarella plus the accompanying whey product and butter) will be an additional useful perspective, because it illustrates the total change from one technology to another, and excludes the high sensitivity to allocation. Figure 4 provides an illustration of this.

The process level approach may be valuable for replication in other production systems, in which byproduct flows separate from the main product flow early in the processing facility, or require large energy use in byproduct processing. Examples are whey processing (Aguirre-Villegas et al. 2012), drying of byproducts from sugar production or from wet milling wheat grain, and drying brewers grains from beer brewing.

5. Conclusions

This paper evaluated how different allocation approaches affect the results of an LCA for a specific mozzarella producing facility. The process level approach provides useful detail that clarifies the incentives for a producer to improve processes that are specific to each coproduct: Improving

processes in mozzarella production accurately benefits the mozzarella impact, and the same is true for whey processing. Economic allocation relates incentives for production to the different coproducts while dry matter allocation also includes economic considerations through the definition of waste. The different allocation approaches may result in different technology preferences. In the evaluation of technology options in one dairy facility, process level economic allocation was found most unambiguous and straightforward to interpret. Although a growing consensus on allocation may be recognized for footprinting in the developments of industry guidelines, the goal and context of different LCA studies may best be served with different allocation approaches. The ISO standard and scientific papers can be interpreted from different angles, which allows for these different approaches. This indicates that the debate on allocation is not likely to be finished.

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228. Evaluation based on data quality of allocation methods for calculation of carbon footprint of grass-based dairy production

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Abstract

A major methodological issue for life cycle assessment (LCA), commonly used to quantify greenhouse gas (GHG) emissions from livestock systems, is allocation from multifunctional processes. When a process produces more than one output, the environmental burden has to be assigned between the outputs, such as milk and meat from a dairy cow. National and international guidelines provide different recommendations on allocation. In the absence of an objective function for allocation, a decision must be made considering a range of factors. The objective of this study was to evaluate 7 methods of allocation to calculate the global warming potential (GWP) of the economically average (€/ha) dairy farm in Ireland considering both milk and meat outputs. The methods were: economic, energy, protein, energy, mass of liveweight, mass of carcass weight and physical causality. The data quality for each method was expressed using a pedigree matrix score based on reliability of the source, completeness, temporal applicability, geographical alignment and technological appropriateness. Scenario analysis was used to compare the normalised GWP per functional unit (FU) from the different allocation methods, for the best and worst third of farms (in economic terms, €/ha) in the national farm survey. For the average farm, the allocation factors for milk ranged from 75% (physical causality) to 89% (mass of carcass weight), which in turn resulted in a GWP / FU, from 1.04 to 1.22 kg CO₂-eq/kg FPCM. Pedigree

scores ranged from 6.0 to 17.1 with protein and economic allocation having the best pedigree. It was concluded that the choice of allocation method should be based on the quality of the data available, that allocation method has a large effect on the results and that a range of allocation methods should be deployed to understand the uncertainty associated with the decision.

Introduction

With the global human population predicted to increase to over 9 billion by 2050 there will be a rise in consumption of bovine milk and meat products (FAO, 2009). Increasing primary production from large ruminant systems to meet greater demand is expected to increase greenhouse gas (GHG) emissions. To tackle this problem, EU nations have legally agreed as part of the 2020 climate and energy bill to reduce GHG emissions from the non-emission trading sector, which includes agriculture. The EU aims to reduce these emissions by 10% (20% in an Irish context) by 2020 relative to 2005 levels.

Life cycle assessment (LCA), an internationally standardized methodology (ISO14040), is the preferred method to estimate GHG emissions from agricultural systems (IDF, 2010; Thomassen and De Boer, 2005). A single impact LCA focused on GHG emissions is commonly referred to as a carbon footprint (CF). A major methodological issue of LCA is allocation between multiple outputs of a process. When a system such as a dairy farm or a process produces more than one output, the environmental burden such as GHG emission, has to be allocated between these outputs, e.g. milk and meat.

Generally LCA guidelines (BSI, 2011; IDF, 2013) recommend where achievable, allocation should be avoided, but where this is not possible guidelines differ on how to allocate, e.g. PAS2050 recommends using economic relationships while IDF (2013) recommend using physical relationships. It is well documented that for LCA studies data quality has a significant impact on the uncertainty and robustness of the results (Henriksson *et al*

2011; Rice *et al* forthcoming). While ISO standards recommend that data quality be reported, this is not that common in LCA studies. Consequently, it is typically necessary for LCA practitioners to make judgement with respect to the accuracy of LCA outcomes. While a data quality scoring/judgement matrix has been developed (Weidema and Wesnaes, 1996), the concept has never been applied in the context of allocation and the choice of method. **The goal (objective) of this study was to evaluate and assess the suitability of seven different allocation methods applied to grass based dairy production in terms of data quality.**

The methods of allocation assessed were: economic, energy, emergy (novel application), protein, mass of liveweight (LW), mass of carcass weight (CW) and physical causality. The data quality (pedigree) was assessed using (1) reliability of the source and completeness; (2) temporal correlation; (3) geographical correlation; and (4) technological correlation, which is in keeping with the data quality requirement stipulations set out by the ISO (2006).

Materials and methods

The data used for completing the LCA of grass based milk production were derived from the 2012 Irish National Farm Survey (Table 1) (NFS, Hennessey *et al* 2013) as previously described by O'Brien *et al.*, (2015). The survey was carried out on 256 dairy farms in 2012 and was weighted according to farm area to represent the national population of specialized dairy farms (15,600). All the dairy farms in the NFS used grass-based spring calving with seasonal the milk supply matched to grass growth patterns, in order to maximise grazed grass intake (Kennedy *et al* 2005).

The LCA methodology was applied according to the ISO (2006) guidelines. The goal was to evaluate 7 methods of allocation using an economically average Irish dairy (€ / ha) farm between milk and meat. The system boundary was 'cradle to farm gate', including foreground processes of milk production and background processes for production and transportation of mineral fertilizer, cultivation, processing and transportation of concentrate

feed. Infrastructure (animal housing, slurry storage facilities, and roads), machinery (tractor, milk cooling system) were not included, as these have a small influence on the GHG's from milk production (O'Brien *et al* 2014). The functional unit was kg of fat and protein corrected milk (FPCM) calculated as to 4% fat and 3.3% protein using (Clark *et al* 2001) where FPCM (kg/yr) = Production (kg/yr) × (0.1226 × Fat % + 0.0776 × True Protein % + 0.2534).

The GHG emissions, methane (CH₄), nitrous oxide (N₂O), carbon dioxide (CO₂) and halocarbons (F-gases) were calculated using the cradle to farm-gate LCA model of O'Brien *et al* (2014) that was certified by the Carbon Trust. The model used previously published algorithms and data from the NFS to calculate on and off-farm GHG emissions using Intergovernmental Panel on Climate Change (IPCC) guidelines (IPCC, 2006) and Irish GHG national inventory methods (O'Brien *et al* 2014). Within the model, the various GHG emissions were converted to CO₂-equivalents (CO₂-eq) using the IPCC (2007; O'Brien *et al* 2014) revised guidelines for GWP and summed to establish the farm CO₂-eq emissions. The GWP conversion factors for the key GHG emissions in the model were 1 for CO₂, 25 for CH₄ and 298 for N₂O, assuming a 100 year time horizon. The CF of both milk and meat were estimated by allocating the GHG emissions between milk and meat.

Energy allocation is based on the 'embodied energy' in milk and meat from culled cows and surplus calves and quantified in solar energy equivalents (seJ). Allocation by physical causality was based on the IDF (2010) guidelines and reflected the underlying use of feed energy by the dairy animals to produce milk and meat. Economic allocation was based on sales receipts for milk and animals from culled cows and surplus calves at the farm gate. Mass allocation was based on the weight of milk and weight of culled dairy cows and surplus calves. The mass of animals was calculated in terms of LW (liveweight) and CW (carcass weight). Allocation by protein was expressed in kg of protein and based on the edible protein in milk and meat from culled cows and surplus calves. Energy allocation was expressed in

joules (J) of energy and based on edible energy in milk and meat from culled cows and surplus calves.

The quality of the data was assessed by the pedigree matrix of Weidema and Wesnaes (1996) for each allocation method. The overall pedigree score was calculated for each allocation method based on the sum of the component scores, weighted by proportional contribution to the calculation where this could be assessed (e.g. proportional mass of milk and meat). The methods were then ranked based on pedigree score. For each allocation method the highest possible score was 25 and the lowest was 5 and a lower score represents a better pedigree of data.

Results and discussion

From running the activity data through the LCA model of O'Brien et al.,(2014),it was estimated that 477791.17 kg Co₂e were generated by the 'mean' group of farms .The results of this study has shown significant differences in the allocation proportion to milk (and associated meat co-product) (Table 3). For the 'Average' farm, the allocation factors ranged from 75% (physical causality) to 89% (mass of carcass weight)(Table 3), which in turn resulted in over a 17% difference in the CF values, i.e. 1.04 – 1.22 kgCo₂-eq/kg FPCM, depending on which allocation method was used. This range in allocation of emissions to milk in turn resulted in over an 11-fold difference in the CF values for meat, i.e. 0.61 – 7.49 kgCo₂-eq/kg meat. Regarding both FPCM and meat, physical causality resulted in the smallest difference i.e. 2.5% less for FPCM and 15% more for meat, compared to when economic allocation was applied. Moreover, the application of allocation by way of mass of carcass weight (CW) resulted in the greatest difference, i.e. 15% more for FPCM and 90% less for meat, compared to when economic allocation was applied. The CF's were achieved with data of widely varying pedigree (Table 4), from the simpler allocation methods (mass LW, mass CW), to the more complex methods (energy, emergy, physical causality (Table 4).With regards to FPCM, both protein content and economic allocation methods had the best

pedigree of data, with a pedigree matrix score of 6 (Table 4), whilst the energy content allocation method had the worst pedigree of data with a pedigree matrix score of 17.1 (Table 4). With regards to meat, protein content had the best pedigree of data, with a pedigree matrix score of 6.5 (Table 4), whilst both the energy content and energy allocation methods had the same pedigree matrix score of 19.4 (Table 4), indicating that they had the worst pedigree of data behind them.

Conclusion

Allocation method has a large effect on the CF result, > 11 fold difference in the case of meat. Based on pedigree score, protein content followed by the simple mass allocation methods by LW or CW were best for milk. Energy and energy were of poorest pedigree and the others fitted in between. In most cases it was only the scores for one or two indicators that dominated the final pedigree score for each method. This was also observed by Weidema and Wesnaes (1996), so if a particular method is to be used for theoretical reasons, then focused effort will be required to ensure the best possible data are available in order to justify its use from a data pedigree perspective. A further reason to be careful with the more complex methods is that they are built on a foundation of the simple methods with a cascade of additional data. This study showed the importance of using country, technology and temporally specific data so the goal and scope specification for the study should be consistent with the time that can be committed to the allocation calculations. It was also noted that when assessing meat co-products the method chosen can be used to bias the study. From the data presented here it seems that physical causality will be biased in favour of milk, and in the case of physical causality, obtaining good pedigree data to justify such an approach is difficult. A range of methods should be deployed to understand the uncertainty associated with the decision.

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Table 1. Key technical measures collected by Hennessey et al. (2013) for the mean of a sample of 221 Irish dairy farms ranked in terms of gross margin/ha.

Item	Mean
Dairy farm area, ha	35
Milking cows, number	67
Culled cows, %	17
Stocking rate, cows/ha	1.89
Soil class (1 = yes)	59%
FPCM yield, kg/cow	5181
Fat, %	3.94
Protein, %	3.4
FPCM ¹ yield, kg/ha	9776
Milk yield, kg/farm	352915.62
Concentrates, kg DM/cow	898
Grazing days	239
N fertilizer, kg/ha	196
Purchased fuel, l/ha	110
Electricity, kWh/cow	182
Gross margin, €/ha	1,758

¹ Fat and protein corrected milk, standard milk corrected to 4% fat and 3.3% protein

Table 2. Data quality Pedigree Matrix by Weidema and Wesnaes (1996)

Indicators	Indicator Score				
	1	2	3	4	5
Independent of the study in which the data are applied:					
Reliability of the source	Verified data based on measurements	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 and industrial 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 a smaller number of sites but for adequate periods	Representative data from an adequate number of sites but for shorter periods	Representative data from a smaller number of sites and shorter periods, or incomplete data from an adequate number of sites and periods	Representativeness unknown or incomplete data from a smaller number of sites and /or from shorter periods
Dependent on the goal and scope of the study:					
Temporal correlation	Less than 3 years of difference to year of study	Less than 6 years of difference to year of study	Less than 10 years of difference to year of study	Less than 15 years of difference to year of study	Age unknown or more than 15 years of difference to year of study
Geographical correlation	Data from area under study	Average data from larger area in which the area under study is included	Data from area with similar production conditions	Data from area with slightly similar production conditions	Data from an unknown area or with very different production conditions
Technological correlation	Data from enterprises, processes and	Data from processes and materials under	Data from processes and materials under	Data on related processes and materials but from	Unknown technology or data on related processes or materials



	materials under study	study but from different enterprises	study but from different technology	same technology	but from different technology
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Table 3. The allocation proportion for milk under the seven allocation methods, for the mean category of Irish dairy farms in terms of gross margin/ha

¹Greenhouse Gas measured in kg CO₂e

	Method of allocation						
	Mass of Liveweight	Mass of carcass weight	Protein content	Energy content	Emergy	Economic	Physical causality
GHG¹ allocated to FPCM	88%	89%	83%	81%	84%	77%	75%

Table 4. Application of Pedigree matrix (Weidema and Wesnaes,1996) to the methods of allocation , with regards to milk and meat

	Method of Allocation						
	Mass of Liveweight	Mass of carcass weight	Protein content	Energy content	Emergy	Economic	Physical causality
Pedigree matrix score FPCM	6.1	6.1	6	17.1	16.9	6	9.5
Pedigree matrix score meat	6.6	6.6	6.5	19.4	19.4	8.4	9.5

Table 5. Individual pedigree component scores for contributors to allocation calculations

	Indicator							Method of allocation where used							
	Scored components	Reliability	Completeness	Temporal	Geographical	Technological	Total	Mass of Liveweight	Mass of carcass weight	Protein content	Energy content	Emergy	Economic	Physical causality	
Meat	Number of culled cows	1	1	1	1	1	5	x1	x	x	x	x	x	x	
	Culled cow KgLW/hd	1	1	1	2	1	6	x	x	x	x	x	x	x	
	Culled cow KO%	1	4	4	3	3	15	x	x	x	x	x	x	x	
	Culled cow KgCW/hd	1	1	1	2	1	6	x	x	x	x	x	x	x	
	Culled.cow carcass protein %	1	5	2	5	1	14	x	x	x	x	x	x	x	
	Culled.cow carcass fat%	1	5	2	4	5	17	x	x	x	x	x	x	x	
	Number of surplus calves	1	1	1	1	1	5	x	x	x	x	x	x	x	
	Surplus calf KgLW/hd	2	2	3	1	1	9	x	x	x	x	x	x	x	
	Surplus calf Ko%	1	4	4	3	4	16	x	x	x	x	x	x	x	
	Surplus calf kgCW/hd	1	1	1	2	1	6	x	x	x	x	x	x	x	
	Surp.calf carcass protein%	1	4	5	4	5	19	x	x	x	x	x	x	x	
	Surp.calf carcass fat%	1	5	2	4	5	17	x	x	x	x	x	x	x	
	Prot.Energy(J)	1	5	4	5	5	20	x	x	x	x	x	x	x	
	Fat energy(J)	1	5	4	4	5	19	x	x	x	x	x	x	x	
	Beef energy(sel/J)	1	5	4	4	5	19	x	x	x	x	x	x	x	
	Culled cow €/kgCW	1	5	1	2	1	10	x	x	x	x	x	x	x	
	Surplus male dairy calf (€/hd)	3	5	1	2	1	12	x	x	x	x	x	x	x	
	Surplus female beef calf (€/hd)	3	5	1	2	1	12	x	x	x	x	x	x	x	
	Milk	Parameter (Thoma 2012)	1	1	3	4	3	12	x	x	x	x	x	x	x
		Non corrected milk,kg	1	2	1	1	1	6	x	x	x	x	x	x	x
Total protein produced,kg		1	2	1	1	2	7	x	x	x	x	x	x	x	
Total fat produced,kg		1	2	1	1	4	9	x	x	x	x	x	x	x	
Total Lactose produced(kg)		1	3	3	3	2	12	x	x	x	x	x	x	x	
Prot.Energy(J)		1	5	4	5	5	20	x	x	x	x	x	x	x	

21. Environmental impact of South African milk: A comparison of Ayrshire and Holstein breeds

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ABSTRACT

In South Africa and other emerging economies, the demand for dairy products is growing rapidly. In view of the considerable environmental impacts caused by dairy production systems, environmental mitigation strategies are required. One factor affecting the environmental impact of dairy products is the breed type. In this case study, a Life Cycle Assessment (LCA) was performed for raw and processed milk from Holstein and Ayrshire cows. Primary data was provided by five South African dairy farms and two dairy plants.

The raw milk production is the decisive factor when analysing the environmental impact of dairy products. Regardless of the breed type, the carbon footprint of raw milk is dominated by direct methane and dinitrogen monoxide emissions, while the production of the feed, as well as the infrastructure and energy demand of the farms, play a major role in all other impact indicators. On farms where both Ayrshire and Holstein cows are kept, the environmental impact is lower for milk from Holstein cows. However, differences between farms outweigh differences between breeds. The implementation of best practices therefore has substantial environmental mitigation potential for livestock farming in South Africa.

Keywords: LCA; dairy; cattle; agriculture

1. Introduction

As in most emerging economies, livestock is one of the fastest growing sectors of the agricultural economy in South Africa (DAFF, 2013). Over the past ten years, milk production increased by 26%, 49% and 73% in South Africa, Brazil and India respectively (DAFF, 2015; FAO, 2015). The global dairy sector accounts for 4% of anthropogenic greenhouse gas emissions and is therefore a major contributor to climate change (FAO, 2010). Gerber et al. (2013) state that interventions in order to mitigate these emissions are largely based on technologies and practices that increase production efficiency. Consequently, the feed quality, the feeding regime and the breeding are seen as key factors in reducing the emissions of milk production (Gerber et al., 2013). The breed type can also have a significant influence on the environmental impact of dairy products. Capper & Cady (2012) show that the carbon footprint of cheese production in the US is lower for Jersey cows than for high-producing Holstein cows due to a reduced energy requirement of the Jersey herd. Given the increasing demand for dairy products in South Africa and other emerging economies, the question arises as to whether the selection of the breed type used to meet this rising demand can help to reduce the environmental impact of dairy production systems. In South Africa, Holstein, Jersey, Guernsey and Ayrshire are the major dairy breeds (DAFF, 2015). This study analyses and compares the environmental impact of raw and processed milk of Holstein and Ayrshire cows from five farms located in the South African province of KwaZulu-Natal.

2. Goal and Scope

In order to evaluate the environmental impacts of dairy farming in the South African Province of KwaZulu-Natal (KZN) and of the subsequent dairy processing, a cradle-to-gate Life Cycle Assessment (LCA) was performed. While the functional unit is 1 kg for processed dairy products at a dairy plant, the environmental impact of raw milk at the farm gate was computed for three scenarios: on a weight basis, on a fat and protein corrected milk (FPCM) weight basis, and on a price basis. This differentiation was introduced to account for different milk qualities depending on the breed: Ayrshire milk is advertised as having a superior taste by one of South Africa's largest retail stores. This is reflected in a slightly higher price for Ayrshire milk as compared to conventional milk.

The LCA covers the rearing of a female calf, the keeping of the adult dairy cow, the transport of the raw milk to a dairy factory and the milk processing. Data on milk production and processing was collected on five farms in KZN and two dairy plants in KZN and the Western Cape, respectively. The data collection took place in August 2014. The environmental impacts were assessed using six different impact indicators, namely climate change according to the Intergovernmental Panel on

Climate Change (IPCC, 2013), the cumulative non-renewable energy demand according to Hirschier et al. (2010), land use according to Frischknecht et al. (2013), freshwater and marine eutrophication according to Goedkoop et al. (2009) and freshwater ecotoxicity according to Rosenbaum et al. (2008). Background data for the life cycle inventories was taken from the international ecoinvent v3.1 database using the system model “allocation, recycled content” (ecoinvent Centre, 2014). The life cycle inventories and the impact assessment were issued with the SimaPro software v8.1 (PRé Consultants, 2016).

3. Life Cycle Inventory

On average, Holstein and Ayrshire cows participating in the South African National Milk Recording Scheme produce 7441 kg and 6072 kg of milk per lactation, respectively (Ramatsoma et al., 2015). Roughly 40% of the South African herds have a milk yield below 5500 kg per cow and for 7% of the herds the yearly milk production per cow is higher than 9125 kg (Milk SA, 2015). For the herds considered in this case study, the yearly milk yield per cow ranges from 4822 kg to 9200 kg. The milk yield and other key characteristics of these herds are listed in Table 1. Furthermore, all farmers participating in this case study provided detailed information on their feeding regimes, on their electricity demand and on the water use for irrigation and other purposes. Direct methane emissions were calculated using the Tier 2 approach described by the IPCC (2006). In addition, direct and indirect nitrous oxide emissions and ammonia emissions were computed. The allocation at farm-level is based on the physiological feed requirements to produce milk and meat. This approach is recommended by the International Dairy Federation IDF (2015). Furthermore, economic allocation was used to distribute the environmental impact of meat between calves and cull dairy cows.

Table 1: Characteristics of the five dairy farms. Values in italic have been estimated due to lacking primary data. H: Holstein; A: Ayrshire

Breed	Farm 1		Farm 2		Farm 3		Farm 4		Farm 5	
	H	A	H	H	A	H	H	A	H	
Milk yield (l/a)	8300	7300	9200	8000	4822	8250*	7500			
Live Weight (kg)	545	600	675	560**	420	460	650			
Age at first calving (months)	27	27.5	27.5	27	26	28	28			
Number of lactations	2.4	5.5	5.5	2.8	3.6	3.6	3.8			
Protein content of milk	3.20%	3.24%	3.15%	3.20%	3.34%	3.29%	3.25%			
Fat content of milk	3.40%	4.10%	3.48%	3.60%	3.80%	3.48%	3.75%			
Price for milk (ZAR/l)***	13.5	14.5	13.5	13.5	14.5	13.5	13.5			

* The yearly milk yield specified by Farm 4 for Holstein cows is 4282 kg. However, Farm 4 indicated that all Holstein cows were in their first lactation. For that reason, the average milk yield of Holstein cows on Farm 1, 2, 3 and 5 has been assumed to be true for Holstein cows on Farm 4

** Average life weight of Holstein cows on farms 1, 2, 3 and 5

*** Prices are in South African rand (ZAR) and refer to consumer prices for conventional milk and Ayrshire milk, respectively. Prices were taken from www.woolworths.co.za (retrieved 21.4.2016)

The seven datasets created based on the information in Table 1 were aggregated to a single production mix which was used to issue the life cycle inventories of processed dairy products. The two dairy plants considered in this case study provided information on their raw milk, energy, water and chemicals input and on the production volumes of different dairy products. Also transport, infrastructure and waste water were included in the inventory. Allocation of environmental impacts between different dairy products was performed based on the dry matter content of the dairy products as recommended by the IDF (2015).

4. Life Cycle Impact Assessment

Raw milk production in KwaZulu-Natal is associated with greenhouse gas emissions of between 1.2 and 2.0 kg CO₂-eq/kg. Direct methane and dinitrogen monoxide emissions account for 66%-73% of the climate impact of raw milk (Figure 1).

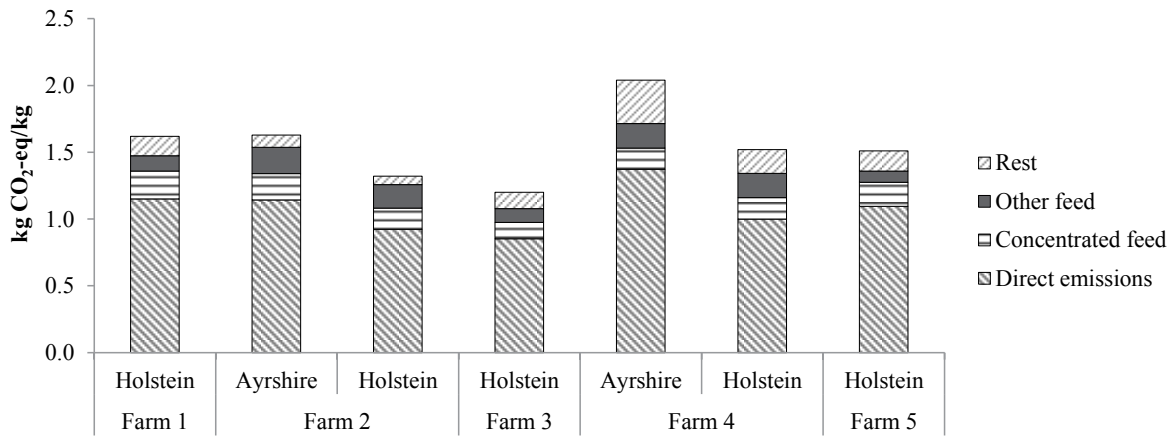


Figure 1: Greenhouse gas emissions of raw milk at five farms in KwaZulu-Natal

On farms 2 and 4, where both Ayrshire and Holstein cows are kept, the climate impact is lower for milk from Holstein cows, even when using FPCM or price as a functional unit (Figure 2). Holstein cows have a higher milk yield than Ayrshire cows and therefore a lower specific feed intake than Ayrshire cows. However, variations across farms are larger than variations across breeds.

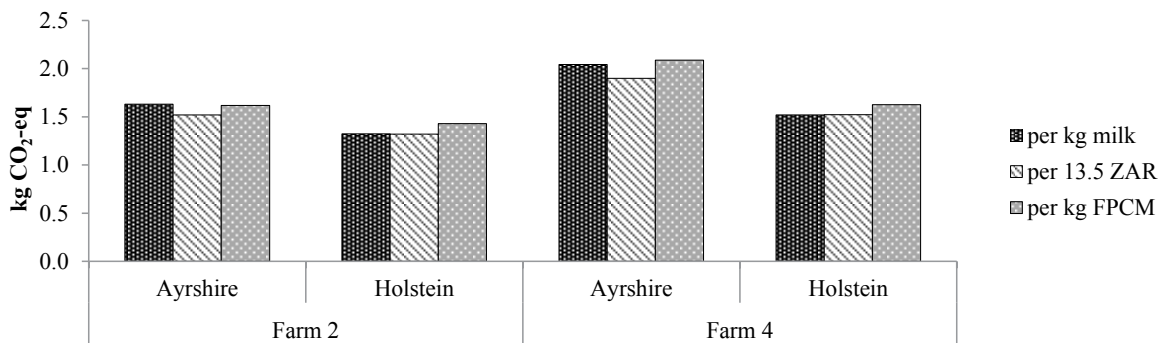


Figure 2: Greenhouse gas emissions of Ayrshire and Holstein milk at Farm 2 and Farm 4 depending on the functional unit used. 13.5 South African rand (ZAR) is the price assumed per liter of conventional milk (see Table 1).

While direct emissions play a major role in the greenhouse gas emissions of raw milk, other environmental impact indicators are dominated by the production of concentrated feed and the housing system (Figure 3). The non-renewable energy demand of raw milk depends largely on the electricity demand of the dairy farm and the use of diesel for the production of concentrate feed. The eutrophication potential can mainly be related to the use of fertilizers in the production of wheat and maize which are the main components of concentrate feed. The freshwater ecotoxicity is primarily caused by herbicide emissions from maize production. Land is predominantly used to grow wheat and maize for the production of concentrate feed.

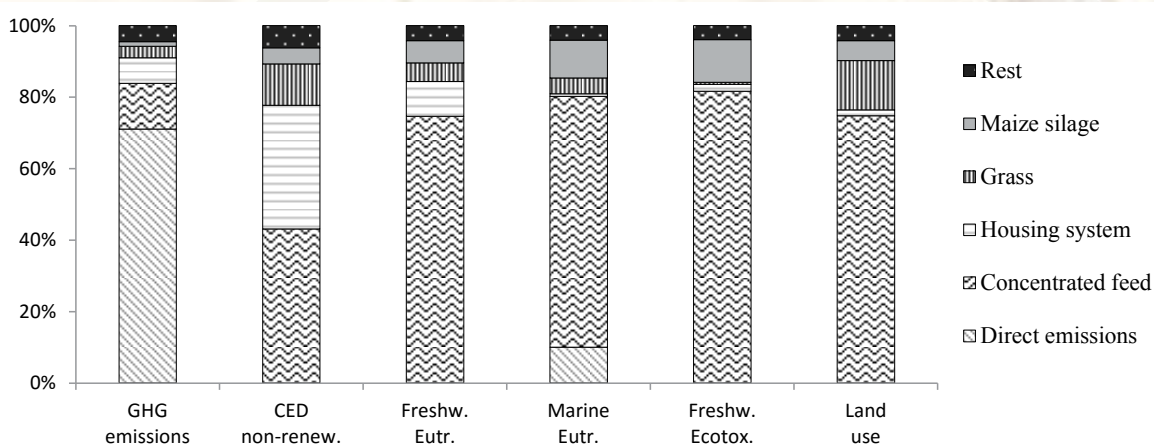


Figure 3: Contribution analysis for raw milk at Farm 1 for six impact indicators

The carbon footprint of dairy products can mainly be attributed to the production of raw milk, while milk transport as well as the infrastructure and energy demand of the dairy factory only play a minor role (Figure 4). For pasteurised milk, the greenhouse gas emissions range from 1.3 kg CO₂-eq/kg for milk with a fat content of 0.5% (dairy factory A) to 1.9 kg CO₂-eq/kg for milk from dairy factory B. Due to its high dry matter content, butter is the product with the highest climate impact (13 kg CO₂-eq/kg). In general, dairy products from dairy factory A have a lower environmental impact than products from dairy factory B. This can be attributed to a higher resource efficiency of dairy factory A as compared to dairy factory B.

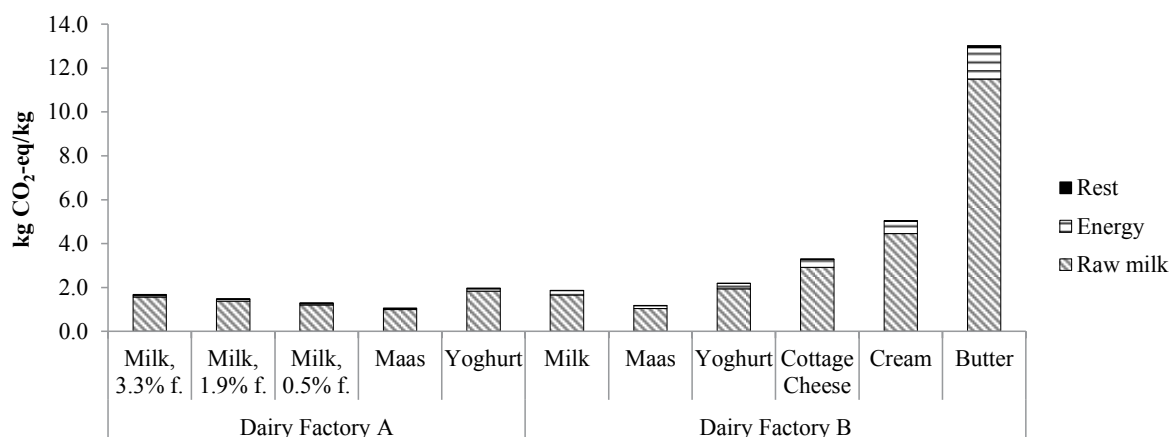


Figure 4: Contribution of the raw milk production, the energy demand of the milk processing and other processes to the carbon footprint of dairy products from two South African dairy factories. Maas: fermented milk (common South African dairy product); f.: fat.

Not only the carbon footprint but also the results for all other impact indicators are dominated by the production of raw milk. The energy demand of the dairy factory is only relevant when considering the cumulative energy demand (CED) of processed milk. In comparison with dairy factory B, dairy factory A uses 74% less energy for the processing of milk.

5. Interpretation and conclusion

Average greenhouse gas emissions from the production of raw milk in the South African province of KwaZulu-Natal amount to 1.6 kg CO₂-eq per kg (ranging from 1.2 to 2.0 kg CO₂-eq/kg). Similar results were published for other countries. The ecoinvent database reports a carbon footprint of 1.4 kg CO₂-eq/kg FPCM for Canadian raw milk (ecoinvent Centre, 2015) and the Agribalyse database shows greenhouse gas emissions amounting to 0.99 kg CO₂-eq/kg FPCM for French cow milk (ADEME, 2015).

The environmental impact of raw milk varies between farms and between breeds. Cross-farm differences can largely be attributed to differences in the resource efficiencies of farms. The environmental impact is lowest for milk from Holstein cows on farms 2 and 3 for almost all impact indicators. While milk production on farm 2 is characterized by a high energy intake of dairy cows coupled with a high milk yield, the energy intake of cows on farm 3 is low with at the same time relatively high milk yields (Figure 5).

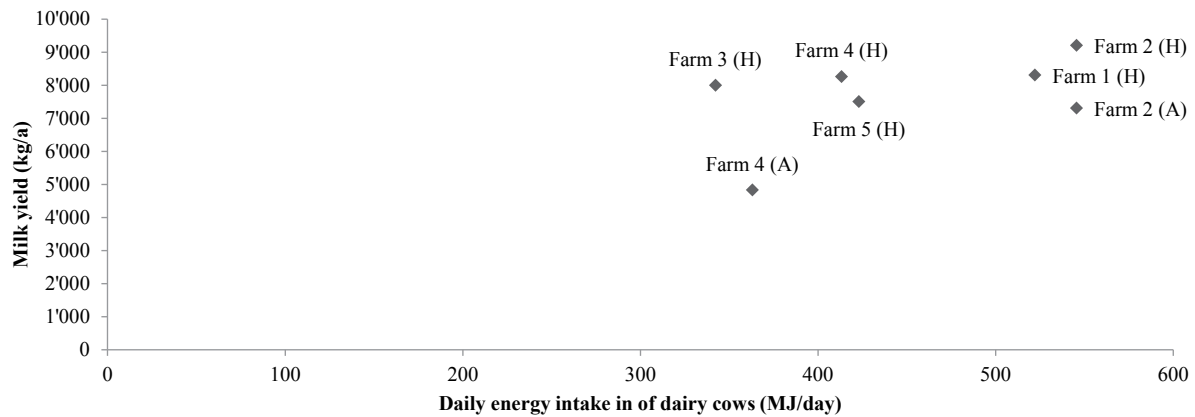


Figure 5: Relationship between daily energy intake of dairy cows and milk yield for the five farms. A: Ayrshire; H: Holstein

In contrast to Capper & Cady (2012), who compared Holstein and Jersey milk and showed that the environmental impact of dairy production is lower when using milk from a breed with a lower milk yield (Jersey), the present study found that the high-yielding Holstein breed performs better than the lower-yielding Ayrshire breed. However, differences between farms outweigh differences between breeds.

Direct methane emissions are a key factor in the carbon footprint of milk and, thus, measures to reduce emissions from enteric fermentation are decisive. According to Knapp et al. (2014) the most promising strategies to reduce enteric methane emissions combine genetic and management approaches. Effective measures include genetic selection for animals with lesser enteric methane emissions and higher production efficiency (genetic approach), as well as practices to reduce non-voluntary culling and diseases, improvements in nutrition and the reduction of stress factors such as heat (management approaches) (Knapp et al., 2014). Hristov et al. (2015) suggest using feed supplements to achieve a significant reduction in methane emissions from enteric fermentation. In an experiment with 48 Holstein cows the use of the methane inhibitor 3-nitrooxypropanol (3NOP) led to a reduction of 30% in rumen methane emissions, while milk production was not affected by the inhibitor (Hristov et al., 2015).

For impact indicators other than climate change measures to reduce the environmental burden related to the production of concentrated feed are decisive. Especially for irrigated crops, significant reductions can be achieved through the use of renewable energies in the agricultural production: Wettstein et al. (2016) show that the use of solar power for irrigation reduces the cumulative non-renewable energy demand and the freshwater eutrophication potential of irrigated South African maize by 43% and 12%, respectively.

In conclusion, a considerable variability of the environmental impact of milk from the South African province of KwaZulu-Natal can be observed. Compared to milk from Ayrshire cows, the environmental impact of milk from high-yielding Holstein cows tends to be smaller, but differences between farms are greater than differences between breeds. These findings indicate that the implementation of best practices has a substantial environmental mitigation potential for livestock farming in South Africa.

6. Acknowledgement

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307. LIFE CARBON DAIRY, the milk carbon footprint of 3,300 French dairy farms

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ABSTRACT

The French Livestock Institute (Institut de l'Élevage), in association with three partners, has launched the project LIFE CARBON DAIRY with the main objective to promote an approach allowing milk production to reduce by 20% the milk carbon footprint at farm level over 10 years. The three other leading partners are key players in the French dairy sector from the advisory services on dairy farms (Dairy advisors enterprises as ECEL and Chamber of agriculture) to dairy processors and CNIEL. To achieve the goal, project' partners developed a Life Cycle Assessment (LCA) tool named CAP'2ER® tool (Moreau et al., submitted to LCA food 2016) to assess the milk carbon footprint on dairy farms in France. Answering the LCA approach, the milk carbon footprint assessed in CAP'2ER® is covering the greenhouse gases (GHG) emissions to determine the Gross Carbon Footprint (GCF) and carbon sequestration to assess the Net Carbon Footprint (NCF). Applied on 3,316 farms representing various milk production systems in France, the project provides a good overview of the average national milk carbon footprint. In parallel, each individual evaluation gives participating farmer management factors to identify opportunities to improve farm efficiency and reach the carbon reduction target. On the 3,316 farms assessed, the average GCF is 1.04 kg CO₂e per liter Fat and Protein Corrected Milk (FPCM) and NCF, 0.93 kg CO₂e per liter FPCM. In the over way, carbon sequestration compensates 11% of GHG emissions. Variations in GCF (CV = 15%) are explained by differences in farm management. Practices with the largest impact on milk carbon footprint average are milk yield, age at first calving, quantity of concentrate, N-fertilizer used (organic and chemical) and fuel consumed. Farms with the lowest GCF (10% of farms) have an average carbon footprint of 0.85 kg CO₂e per liter FPCM and confirm that farm efficiency is a way to reduce carbon intensity.

Keywords: carbon footprint, French dairy, milk production, soil carbon sequestration.

1. Introduction

Agriculture is a contributor to global greenhouse gases (GHG) emissions, and particularly methane and nitrous oxide. In France, agriculture sector contributes to 18% of overall global GHG emissions (CITEPA, 2015) and 8% comes from ruminants taking into account animals and manure management. On the one hand, French government's targets to cut GHG emissions by 75% of 1990 levels by 2050. For agricultural sector a first target set under the Décret n°2015-1491 is to cut GHG emissions by 12% of 2015 levels by 2028. On the other hand, consumers ask for more information on the environmental impacts of products and their influence on GHG emissions. Meanwhile, retailer companies are enlarging their requirements for their suppliers to environmental impact and specifically carbon footprint.

As practices exist to reduce GHG emissions from livestock activities, we have to demonstrate their effectiveness on a widespread basis through the mobilization of business/professional and structural efforts of the entire livestock industry.

By involving a large number of farmers in six pilot regions which account for 65% of the French milk delivery and are representative of different climate conditions and feeding strategy, LIFE CARBON DAIRY represent a real opportunity to disseminate the carbon footprint on a large scale with the main objective to promote an approach allowing milk production at farm level to reduce by 20% the milk carbon footprint over 10 years.

To answer this goal, the milk carbon footprint calculation is a good way to disseminate to farmers about GHG emissions of their dairy system activity issue and how they can reach these tough environmental target. The objective of this study was to build an approach to apply Life Cycle assessment (LCA) methodology at a farm level, to determine the average

milk carbon footprint produced in France, to assess the sensitivity of the carbon footprint to farming practices and to promote a large carbon action plan in dairy farms.

2. Methods

Carbon footprint calculation

Dairy farms were assessed individually with the CAP'2ER® (environment footprint calculator and decision making for ruminants production systems) tool developed by Institut de l'Elevage (Moreau et al., submitted to LCA food 2016) for French production context. Answering to Life Cycle Assessment (LCA) standards, the system boundaries covered by CAP'2ER represents 'cradle-to-farm-gate' of the dairy unit (on-farm impacts plus embodied impacts from inputs used on the farm; Figure 1). The methodology developed to assess carbon footprint is based on international methodologies (IPCC Tiers 3, CML, LEAP guidelines). The tool also evaluates positive contribution as carbon sequestration and emissions to the environment are expressed in connection with the primary function represented by the product. The functional unit is the quantity of milk in kg Fat and Protein Corrected Milk (FPCM) leaving the farms. To standardize GHG emissions, the International Panel on Climate Change has established the global warming potential equivalence index to convert GHG to CO₂e units. In our model, the conversion factors are 25 kg of CO₂e/kg CH₄ and 298 kg CO₂e/kg N₂O (IPCC, 2007).

The GHG emissions from dairy unit are allocated between milk and meat (surplus calves and cull dairy cows) according a biophysical allocation rule based on feed energy required to produce milk and meat respectively (LEAP 2015).

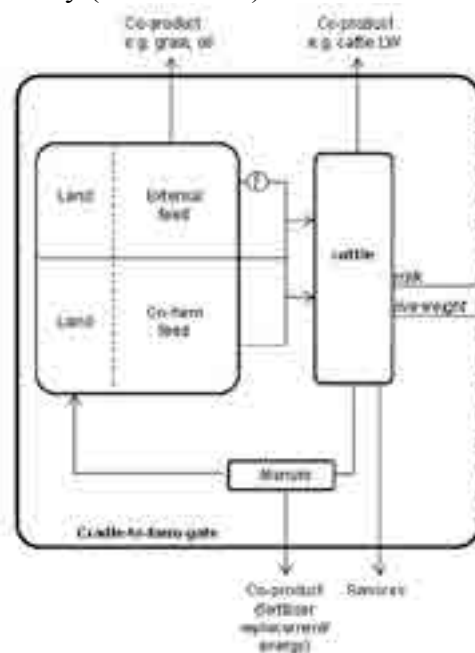


Figure 1: System boundaries (adapted from LEAP large ruminants)

Carbon sequestration

We have assumed that grassland and hedge increase the carbon content of soil every year. Respectively 570 kg carbon per year per ha and 125 kg carbon per 100 ml of hedges. On the other way, arable lands without grass in the crop rotation were considered to decrease the soil carbon content by 160 kg carbon per ha every year (Dolle et al., 2013). But including grass in the rotation cycle on arable lands can increase biomass return in soil's organic matter, and reduce disturbance to the soil through tillage. Thus, the average soil carbon balance per year for the crop rotation with grass was calculated with the assumption that crop decrease soil

carbon content by 950 kg carbon per year and ha and grass increase carbon soil content by 570 kg carbon per year and ha.

Data collection

Technical data were collected on an annual base (2013 and 2014) at the farm level and from a number of producers across six regions and three main forage systems (“Grass system” with less than 20% maize in the forage area, mixt system (“Grass-Maize system”) with grass and maize between 20 and 40% of maize in the forage area and “Maize system” with more than 40% maize in the forage area).

Calculate the milk carbon footprint need a large amount and complex farm level data. The data collection process was achieved on each farm with trained agents from ECEL companies (dairy advisors enterprises). The questionnaire consisted of 150 questions divided in six sections regarding: 1) herd demographics and milk production; 2) animal housing and manure management; 3) crop production for on farm produced feed; 4) feed rations and purchased feed; 5) energy use (fuel and electricity); 6) general information. To ensure the validity of data collected, there are checking points in the questionnaire to test consistency (e.g. comparison between animals dry matter intake need and produced and purchased feed) and the most important parameters were test within an expected range of values.

3. Results

General farm characteristics

In 2016, 3,316 farms have been assessed. General farm characteristics of this sample are different from the French average dairy farm because of an over representation of Western systems (83%), more intensive systems, using maize and producing more milk per cow than mountain grass-based systems. There is a large variation in characteristic and performance data between farms resulting from different farming conditions (climate and soil type), farmer strategies (breed and production system, size ...) and management practices (efficiency, health...). On this farm sample, the average dairy farm produces 467,000 liters of milk with 62 milking cows and 95 ha (Table 1). The stocking rate is 1.5 livestock units per hectare dedicated to the dairy herd. The age at first calving is 29 months and the replacement rate 29%. Dairy cows diet is mainly composed of maize silage with 60% of the total forage dry matter intake (DMI), grazed grass is 29%. Concentrate consumed by dairy cows represents 166 g per liter of milk produced. On farm area N fertilization is 145 kg N/ha.

Table 1: dairy farms characteristics, standard deviation (SD), coefficient of variation (CV), lower and upper 10%

Farm characteristic	Mean	SD	CV%	Lower 10%	Upper 10%
Total Farm size, ha	95	53	0.56	47	159
Area dedicated to the dairy herd, ha	64	32	0.50	35	101
Number of milking cows	62	24	0.39	38	93
Total Milk production, *1000 l	467	199	0.43	265	721
Labour productivity, *1000 l/labour unit	342	168	0.49	179	536
Stocking rate, LU/ha	1.49	0.40	0.27	1.01	1.96
Milk production standard, l FPCM/cow/year	7487	1143	0.15	5989	8808
Fat, g/l	40.1	2.0	0.05	38.0	42.3
Protein, g/l	32.4	1.3	0.04	31.1	33.8
Grazing days, days	184	37	0.20	137	226
Quantity of concentrate, g/l milk	166	56	0.34	107	236

N surplus at farm level, kgN/ha	78	40	31	31.324	125
Hedge, ml/ha	109	49	0.45	24	155
Electricity, kWh/LU	333	107	0.32	207	444
Fuel, l/ha	148	50	0.34	87	211

Milk carbon footprint

Performing a LCA study on a large number of commercial dairy farms provides an insight into the variation between milk carbon footprints that may be related to variation in farm performance and characteristics (e.g. milk yield per cow, forage system, ...).

On the 3,316 farms, the average milk Gross Carbon Footprint (GCF) is 1.04 kg CO₂e per liter FPCM (Table 2) with no significant difference between forage systems. Variations in GHG (+/- 15%) are the same whatever the forage system. On this farm sample, the carbon sequestration associated to grasslands (permanent and temporary) and hedges compensates the GCF by 11%, with variations between the forage systems. The grass system compensates till 30% of his GHG emissions, therefore, in Grass system, Net Carbon Footprint (NCF) is considerably lower than other systems.

Table 2. Milk carbon footprint of different forage systems

	Farm type – Forage system			
	Maize system	Maize/Grass system	Grass system	Total
Number of farms	1,418	1,536	362	3,316
Gross Carbon Footprint, Kg CO ₂ e/ l FPCM	1.05	1.04	1.05	1.04
<i>Coefficient of Variation</i>	14%	14%	19%	15%
Carbon sequestration, Kg CO ₂ e/ l FPCM	0.06	0.11	0.30	0.11
<i>Coefficient of Variation</i>	72%	64%	77%	106%
Net Carbon Footprint, Kg CO ₂ e/ l FPCM	0.99	0.93	0.76	0.93
<i>Coefficient of Variation</i>	15%	15%	31%	18%

A focus on the 10% of farms getting the lower milk GCF results is realized. In this group composed by 333 farms, the average GCF is 0.85 kg CO₂e per liter FPCM and the NCF is 0.75, with 12% carbon sequestration (Figure 2). The variation between total farms and this sample is 18%. These results ensure the possibility to reduce by 20% the national milk carbon footprint at farm level over 10 years.

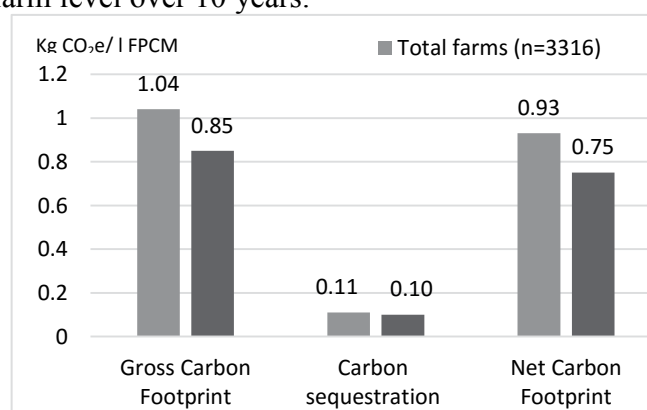


Figure 2: National and 10% lower milk carbon

The major contributor to GHG emissions is the enteric methane (49%) which average 147 kg CH₄/cow. The next largest contributor is the manure management (storage and grazing

cattle) which produce methane and nitrous oxide (18%), followed by nitrous oxide emissions from fertilizer application (11%). The remainder emissions are from inputs purchasing with fertilizer (3%), feed (12%), and direct energies (fuel and electricity, 5%).



Farm practices and carbon footprint

As the objective of the study is to identify the management practices that could be efficient to reduce the milk carbon intensity, relationship between the milk carbon footprint and farm practices and performance were tested using the correlation analysis.

Firstly, correlation is realized between milk GCF and farm practices and parameters. On the one hand, no correlation between the GCF and the herd size or the part of maize in the system was found (Table 3). On the other hand, strong linear correlations were found between the milk GCF and the milk production per cow, the quantity of concentrate, the age at first calving or the nitrogen surplus at farm level.

Table 3: Correlation between various farm performance and characteristics parameters and milk gross carbon footprint (GCF)

Parameter	Correlation with GCF	P-value
Farm size, ha	0.063	<0.001
Number of cows	-0.008	0.626
Total milk production, liter FCPM/cow/year	-0.401	<0.001
% Maize / Total area	-0.002	0.916
Age at first calving, months	0.255	<0.001
Replacement rate, %	0.082	<0.001
Concentrate rate, g/l milk	0.236	<0.001
N-fertilizer use, kg N/ha dairy herd	0.060	0.001
Fuel consumption, l/ha	-0.049	0.005
N surplus at farm level, kgN/ha	0.187	<0.001
Permanent grassland area, ha	0.085	<0.001
Carbon sequestration, kg CO ₂ e/l FPCM	0.199	<0.001

The analysis of the 10% lowest milk GCF farms show that their average GCF is 0.85 kg CO₂e/l FPCM. These farms are performant for not only one parameter accounted for the majority of variation between farm's milk GCF but for almost all. They produce more milk for the same level of concentrate and rear less heifers for the herd replacement. The carbon sequestration is quite the same than average with 0.10 kg CO₂e/l FPCM.

Secondly, the same analysis is realized between the milk NCF and farm practices and parameters. Results are different from the previous ones. Age at first calving, milk production per cow or quantity of concentrate are not correlated with milk NCF whereas part of maize in the system, use of fertilizer or fuel used are strongly correlated (Table 4). The strong correlation between permanent grassland area and NCF shows that carbon sequestration is an important parameter composing the NCF.

Table 4: Correlation between various farm performance and characteristics parameters and milk net carbon footprint (NCF)

Parameter	Correlation with NCF	P-value
Farm size, ha	- 0.062	<0.001
Number of cows	0.040	0.022
Total milk production, liter FCPM/cow/year	- 0.043	0.013
% Maize / Total area	0.415	<0.001
Age at first calving, months	- 0.025	0.150
Replacement rate, %	0.058	0.001
Concentrate rate, g/l milk	0.165	<0.001

N-fertilizer use, kg N/ha dairy herd	0.385	<0.001
Fuel consumption, l/ha	0.248	<0.001
N surplus at farm level, kgN/ha	0.365	<0.001
Permanent grassland area, ha	- 0.291	<0.001
Carbon sequestration, kg CO ₂ e/l FPCM	- 0.493	<0.001

4. Discussion

The LCA results obtained for this average dairy farm sample appeared to be consistent with other LCA studies of milk production obtained on the same perimeter, except for Irish dairy systems based on extensive grass dairy farms with a lower milk GCF (Table 5). The FAO report gives an average milk GCF of 1 kg CO₂e/kg FCPM (Gerber et al., 2010).

Table 5: Comparison of the milk GCF

Countries	Milk GCF (kg CO ₂ e/kg FPCM)	Publication
Ireland	0.74	O'Brien et al (2014)
New Zealand	0.93	Basset-Mens et al (2009)
	0.78	Dollé et al (2016)
France	1.06	Dollé et al (2016)
	0.95	Dollé et al (2015)
Sweden	1.16	Flysjö et al (2011)
World	1.0	Gerber et al (2010)

Our results give a 10% higher milk GCF than found with previous French study (Dolle et al., 2015), but the carbon sequestration is similar. A smaller sample and the profile of the farm with a greater efficiency could explain the lower milk GCF in the previous study.

Our results show that milk GCF is similar between the three forage systems. This means that low environmental impacts can be reached in every dairy system and farm. The grass-based system farms have better NCF results, but GCF, reflects of management efficiency is the same whatever the system. It is possible to say that, in order to reach low milk carbon footprints, two main strategies exist:

- on the one hand, the correlation between farm performance and milk GCF shows that dairy producer can mitigate GHG emissions with practices to improve efficiency. Farms increasing milk production per cow and reducing the number of replacement animals emit less methane enteric per unit of milk. And then, if associated with an efficient use of inputs (fertilizer, feedstuffs and fuel) that reduce direct and indirect CO₂ emissions, farms present a low net carbon footprint, even if their carbon sequestration is lower than average result. To perform better, this type of system can increase their grassland area in order to store more carbon,
- on the other hand, the grass-based system farms that seem to be less performant on the milk production per cow and replacement herd, present a higher carbon sequestration, because of a larger part of grassland storing carbon, and can compensate their methane enteric emissions. This management is a second way to mitigate milk carbon footprint. The most efficient farmers in grass-based system are efficient in the use of their input too. Their milk carbon footprint can be very close to 0.0 kg CO₂e/kg FCPM.

It should be mentioned that it is more appropriate to explore the milk GCF than the milk NCF to analyze the system efficiency. Indeed, in some cases, farms are not efficient and GHG

emissions are very high, but these results can be hidden thanks to an important carbon sequestration mitigating their milk NCF.

To finish, LCA tool as CAP'2ER® gives an interesting opportunity to assess a large number of farm and to involve farmers in decreasing GHG emissions from dairy sector. Given the wide range in milk GCF of 18%, significant opportunities exist for most farms to reduce their milk carbon footprint.

5. Conclusions

The project LIFE CARBON DAIRY represents the opportunity to determine carbon footprint of a first sample of 3,316 dairy farms. The project is up going for two more years with the objective to create a national dynamic and involve 600 news farms in the disposal. These first results are satisfactory and ensure our objective to reduce by 20% the milk carbon footprint at farm level over 10 years. Focused on the relationship between dairy performance and milk carbon footprint, our investigations concern also the other environmental burdens as air and water quality, energy consumption, biodiversity,

The CAP'2ER® tool calculates these others indicators as eutrophication, acidification, total energy use and biodiversity to quantify wider environmental impacts and positive contributions of milk production. Future analyses will be carry out on the carbon dairy database to get the global impact of milk production and to test if the carbon mitigation strategies identified don't cause any undesirable changes in other aspects of environmental performance.


The dissemination of these assessments on farm is permitting to implement a national carbon and environmental action plan to increase dairy sustainability and communicate with stakeholders on the progress done by the dairy sector to reduce environmental impact.

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265. The effect of field drainage on productivity and environmental impact of grass based dairy production systems

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ABSTRACT

The aim of this study was to quantify the greenhouse gas (GHG) emissions of a typical Irish dairy farm. The study considered two farms on poorly drained soil with and without drainage. System simulation and life cycle assessment methodology were combined to calculate the emissions. GHG emissions were quantified per unit of energy corrected milk. The GHG emissions for the farm with field drainage was 1.15 kg CO₂ equiv. per unit functional unit while for the farm without field drainage was 1.18 kg CO₂ equiv. per unit functional unit. It can be concluded that field drainage can improve the productivity of the farm thus reducing the GHG emissions.

Keywords: life cycle assessment, system simulation, dairy production, artificial drainage

1. Introduction

Globally, dairy production is an important source of greenhouse gas (GHG) emissions and accounts for 3% of global emissions (O'Brien *et al.* 2014). In Ireland, grass based dairy production is a key agricultural enterprise (O'Brien *et al.* 2014). The Irish dairy industry is expected to expand by 50% with the abolishment of EU milk quota (DAFF 2010). The Irish Environmental Protection Agency has estimated an increase of 12% in the GHG emission as consequence of expected increases in production (EPA 2012). Many farms in north-western Europe suffer from both poorly drained soil and high precipitation levels (Tuohy *et al.* 2016). Dairy production systems on well and poorly drained soils differ in system properties and management with poorly drained sites having lower productivity (Fitzgerald *et al.* 2008). The trafficability and workability for field operations depend on soil moisture status (Earl 1997). The profitability of milk production systems on well-drained soil is generally greater than on heavy, wet soils (Shalloo *et al.* 2004). Effective land drainage can improve the output of marginal lands (Tuohy *et al.* 2016), for instance subsurface tile drains have been widely installed in the US Midwest to facilitate improved crop production (Williams *et al.* 2015). Shallow drainage techniques are used for low permeability soils to drain excess rainwater in grassland systems (Tuohy *et al.* 2016).

Life cycle assessment (LCA) is widely used to evaluate the GHG emissions from milk production (O'Brien *et al.* 2014); (Chen *et al.* 2016). LCA of agricultural systems typically evaluate environmental impacts up to the farm gate instead of end of life (O'Brien *et al.* 2012). *Dairy_sim* was developed to model pasture-based spring milk production for different climate and soil types (Fitzgerald *et al.* 2008). *Dairy_sim* simulates an optimum farm in which the feed demand of the herd is met by herbage produced, limited concentrate input and minimum housing days (Fitzgerald *et al.* 2008). The three components of *Dairy_sim* are: a herbage growth model, an intake and grazing model and a nutritional energy demand model, combined with a framework for operational management. The output of *Dairy_sim* can be used as part of the inventory for LCA. Fitzgerald *et al.* (2008), simulated grass based dairy production on poorly drained soil in Ireland using *Dairy_sim* where the optimized system was considered to be the equivalent of having field drains installed.

Subsurface drainage of imperfectly drained soils removes surplus soil water and a significant amount of water contaminants (Monaghan *et al.* 2016), (Williams *et al.* 2015). It is also confirmed that sub-surface drains transport N, especially nitrate to streams and open drainage ditches (Williams *et al.* 2015). The Intergovernmental Panel on Climate Change (IPCC) have specified different emission factor for the managed/drained soil to calculate direct and indirect N₂O emissions.

The objective of the study was to evaluate GHG emissions of an optimized Irish pasture-based system on poorly drained soil type without and with field drains.

2. Methods

Dairy_sim was used to find optimum management practices for a theoretical 20 ha dairy unit supporting 2 dairy LU (live unit) ha⁻¹, yielding 5191 kg cow⁻¹ yr⁻¹ on poorly-drained soil before and after field drain installation. Thirty years (1981-2010) weather data for Fermoy, Co. Cork, Ireland were used for the simulations. Management decisions related to silage were taken from the National Dairy Blueprint described by O'Loughlin *et al.* (2001) and related to fertilizer were taken from Humphreys *et al.* (2008). *Dairy_sim* was initialized using Fitzgerald *et al.* (2008), with stocking rate 1.9 cow ha⁻¹, yielding 5871 kg cow⁻¹ yr⁻¹ dairy farm on poorly drained soil at Kilmaley, Co. Clare, Ireland.

The goal of the study was to quantify GHG emissions from a grass based dairy system in Ireland with and without artificial drainage to compare the emissions. The LCA model was developed using GaBi 6 (thinkstep 2014) software. The system produced milk from spring calving dairy cows. Only dairy cows were considered on the farm. The system boundary was from cradle to farm gate. On farm process included the milking unit, manure storage, cattle housing, manure application, grassland (grass and silage), fertilizer application, electricity and diesel use. Off farm processes include manufacturing and transportation of concentrate feed and fertilizer. The functional unit was defined as 1kg of ECM with the reference flow of the herd output in 1 year, where ECM = milk delivered * (0.25 + 0.122 * %fat + 0.077 * %protein) (Yan *et al.* 2013). The fat and protein percentage were assumed to be 3.94 and 3.4, respectively (Hennessy *et al.* 2013). The allocation between milk (88%) and meat (12%) was based on energy and protein requirement (O'Brien *et al.* 2012).

Partial inventory for the LCA was derived from *Dairy_sim* outputs. Energy use for grassland management, fertilizer and slurry spreading was taken from Yan *et al.* (2013). The data for fertilizer processing, cattle housing, manure storage and spreading were taken from Ecoinvent (2014). Concentrate feed mix was taken from O'Brien *et al.* (2012).

Methane (CH₄) emission factors for enteric fermentation (106.2 kg CH₄ head⁻¹ yr⁻¹) and manure management (15.9 kg CH₄ head⁻¹ yr⁻¹) were taken from O'Mara (2006). CH₄ emission from slurry spreading (autumn slurry: 6.8 g CH₄ m⁻³; spring slurry: 12 g CH₄ m⁻³) was taken from Chadwick *et al.* (2000). Direct and indirect nitrous oxide (N₂O) emission factors for manure storage (Direct: slurry: 0.002 N₂O-N (kg N)⁻¹; manure: 0.005 N₂O-N (kg N)⁻¹; Indirect: 0.01 kg N₂O-N), grassland (direct and indirect: 0.01 kg N₂O-N) were taken from IPCC (2006). N₂O emissions from grassland include emissions from manure excretion, fertilizer and manure spreading. Emission factor for direct N₂O emission from field drainage were taken as 1.6 kg N₂O-N ha⁻¹ (Hiraishi *et al.* 2014). Indirect N₂O leaching emission from field drainage were 0.0025 kg N₂O-N kg⁻¹ mineral N whereas without field drains were 0.0075 kg N₂O-N kg⁻¹ mineral N (IPCC 2006). The emission factor for indirect N₂O emissions from field drains was taken as 0.0025 instead of 0.015 kg N₂O-N kg⁻¹ mineral N, as the latter is considered to be very high (IPCC 2006). Ammonia (NH₃) emission factors for manure storage (94 g m⁻² over 30 days) were taken from Duffy *et al.* (2011). NH₃ emission factor for manure spreading (solid: 0.81; autumn slurry: 0.37; spring slurry: 0.60 of TAN applied) were taken from Webb and Misselbrook (2004).

3. Results

Comparison of initial parameterization compared to Shalloo *et al.* (2004), Fitzgerald *et al.* (2008) (Table 1) indicated minor differences, mainly cause by the different weather data because of climate period used. The differences in the simulated optimum management practices of the dairy unit on poorly-drained soil before and after installation of field drains (Table 2) indicated that with field drains the dairy unit can sustain 1.95 cows ha⁻¹ whereas with no drains it can only sustain 1.55 cows ha⁻¹. Housing time was 44 days more for the poorly drained unit without field drains. The amount of

N fertilizer input were largely unchanged. Figure 1 shows the GHG emissions for the dairy farms with poorly drained soil and field drains. For poorly drained soil with no drains the emissions was 1.18 kg CO₂ equiv. per kg ECM whereas with field drains installed it was 1.15 kg CO₂ equiv. per kg ECM.

Table 1: Comparison of *Dairy_sim* parameterization results with previous studies

Property	Unit	Poorly drained soil Shalloo <i>et al.</i> (2004)	Poorly drained soil Fitzgerald <i>et al.</i> (2008)	Dairy_sim simulated farm
Stocking rate	cows ha ⁻¹	1.89	1.90	1.90
Fertilizer	kg ha ⁻¹	238	236	232
Concentrate	kg cow ⁻¹ yr ⁻¹	759	455	464
Grass intake	kg cow ⁻¹ yr ⁻¹	2121	1949	2332
Silage intake	kg cow ⁻¹ yr ⁻¹	2375	2355	2225
Grazing time	days	149	177	187

Table 2: Management practices derived from *Dairy_sim* for a dairy farm on poorly drained soil with and without drains installed assuming national average milk production

Property	Unit	Poorly drained soil	Field drains installed
Cows	numbers	31	39
Stocking rate	cows ha ⁻¹	1.55	1.95
Milk output	kg cow ⁻¹ yr ⁻¹	5191	5191
Fertilizer	kg ha ⁻¹	195	197
Concentrate	kg cow ⁻¹ yr ⁻¹	471	545
Grass intake	kg cow ⁻¹ yr ⁻¹	1953	1996
Silage intake	kg cow ⁻¹ yr ⁻¹	2502	2272
Housing time	days	205	161

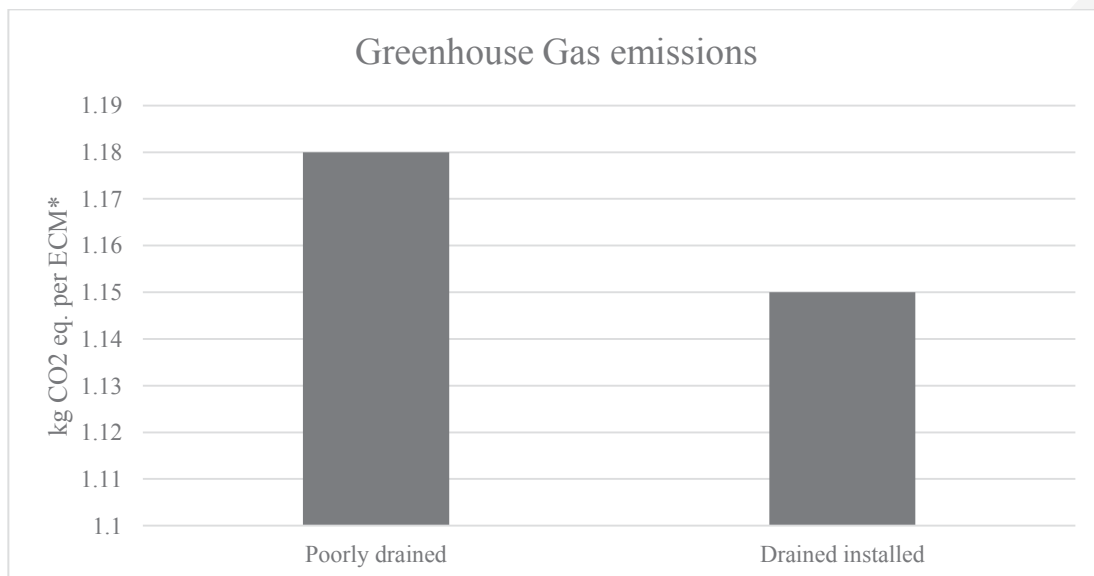


Figure 1: Greenhouse gas emissions of dairy farm on poorly-drained soil and with field drains installed

*ECM energy corrected milk

4. Discussion

The *Dairy_sim* simulation model predicted differences in management practices for the dairy unit following installation of field drains. The dairy unit with field drains could sustain a 20% increase in stocking rate and the housing days were reduced by 21%. The GHG emission values were in range

with Casey and Holden (2005b) and O'Brien *et al.* (2014). Due to difference in scope, assumptions and inventories the results cannot be compared directly with previous studies. The GHG emissions were less by 2.5% per kg ECM for an optimally managed farm following field drains installation. The decrease in emissions for drained farms were mainly due to the increase in productivity. Enteric methane was the major contributor to GHG emission for both situations, as found by Casey and Holden (2005a) and Yan *et al.* (2013). The other main sources of GHG emissions were N₂O from grassland and solid/liquid manure storage. The N₂O emissions from drained farms were higher as than with no drains by 4%.

The percentage of fat and protein assumed in the study can have effect on total emission. If the values of %Fat and %Protein (%F:4.31; %P:3.49; (O'Brien *et al.* 2012) increase the total ECM will increase thus reducing the total emissions. Regional emission factor for enteric and manure management methane were assumed instead of national average (108.81 kg CH₄ head⁻¹ yr⁻¹ and 20.53 kg CH₄ head⁻¹ yr⁻¹) from O'Mara (2006), would result higher methane emissions. Emission factor assumed to calculate indirect N₂O emissions result in lower N₂O emissions from artificial drainage, as the default IPCC value 0.015 is considered to be overestimation of losses (Sawamoto *et al.* 2005); (Reay *et al.* 2003). Similarly, the direct N₂O emissions would increase drastically, if the IPCC default value (8 kg N₂O-N ha⁻¹) is considered, these are now supplement by Hiraishi *et al.* (2014). However, animal live weight and stocking rates will not affect the soil physical properties and herbage production but higher stocking density grazing on poorly drained grassland will have greater poaching damage (Tuohy *et al.* 2014).

5. Conclusions

The result of simulating a dairy unit using *Dairy_sim* before and after drain installation indicated that management practices would change. LCA predicted greater GHG emissions for poorly drained soil compared with the situation once field drains were installed. It was concluded that field drainage can theoretically improve the productivity of optimally managed dairy farms on poorly-drained soil and this was reflected in lower GHG emissions per unit output. Further work is required to evaluate this result for different agroclimatic regions, target milk outputs and other environmental impacts.

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213. Spatial variation of secondary PM_{2.5} exposure and health impact from milk production

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ABSTRACT

Secondary PM_{2.5} human health impacts in life cycle assessment (LCA) are based on linear and simplified assumptions that may lead to a potential double counting. We investigate secondary PM_{2.5} intake fractions (iF) spatial variability due to milk production PM-related emissions in Wisconsin (WI), New Jersey (NJ), and New York (NY) using the Intervention Model for Air Pollution (InMAP). iFs are coupled with epidemiology-based dose-response to provide region-specific characterization factors (CF) for secondary PM_{2.5} and are then tested using a large dairy farm with 2436 animals, performing a sensitivity analysis with location and how impacts compare with nutritional benefits. Our findings suggest that there is substantial spatial variation of iF_{PM_{2.5}} in the U.S. linked to population density and atmospheric chemistry. Although WI emissions can result in PM_{2.5} with a travel distance of about 1500 km, the resulting magnitude of exposure is lower by about a factor of 10 compared to NJ emissions that have the highest exposure estimate with shorter PM_{2.5} travel distance. In regards on our case study, milk production in highly populated NH₃-limited regions, such as NJ, has substantially higher PM_{2.5}-related health impacts to populations downwind than production in agricultural regions where NH₃ is in abundance, such as WI. This research contributes to spatially-explicit CFs for the agricultural sector.

Keywords: agriculture, particulate matter, ammonia, intake fraction, milk

1. Introduction

Agriculture- and food-related processes are usually associated with high ammonia (NH₃) emissions that contribute to the formation of secondary inorganic PM_{2.5}, particles with an aerodynamic diameter of <2.5 μm that are formed in the atmosphere from through photochemical reactions and oxidation processes. Ambient particulate matter (PM_{2.5}) is an important environmental risk factor according to the global burden of disease (GBD) with a burden of about 3 million annual deaths globally (IHME, 2015).

Up to date, secondary inorganic PM_{2.5} human health impacts in life cycle assessment (LCA) have been treated based on linear (Hofstetter, 1998) and simplified (Van Zelm et al., 2008) assumptions related to exposure characterization that do not fully capture the complex relationship between precursors and secondary PM formation and may lead to a potential double counting of corresponding health impact estimates (Fantke et al., 2015). It has been found that precursor availability, and more specifically NH₃ ambient background concentrations, can substantially affect the magnitude of secondary PM_{2.5} exposure (Paulot and Jacob, 2014). The first aim of this study is to provide spatial intake fractions (iF) for secondary inorganic PM_{2.5} for the U.S. and to identify potential factors of influence. The second aim is to estimate PM-related health impacts that may result in from an increase in milk production so as to meet the Dietary Guidelines for Americans. The final aim of this study is to quantify and compare the overall environmental and nutritional effects linked to the addition of one serving milk to the average U.S consumption.

2. Methods

2.1. Intake fraction

Intake fraction (iF) is a metric that links environmental emissions to human exposure (Bennett et al., 2002). To estimate iF we use the following equation by Greco et al., (2007):

$$iF_i = \frac{\sum_i (P_i \Delta C_{ij}) BR}{Q_j} \quad (\text{Equation 1})$$

where P_i is the population in the region of impact, ΔC_{ij} is the change in ambient PM_{2.5} concentration in region of impact i measured in μg m⁻³ due to precursor emissions (PM_{2.5}, NH₃, NO_x, SO₂)

indicating the , Q_j is the precursor emissions in the region of emissions j , and BR is the population breathing rate set at $13 \text{ m}^3 \text{ day}^{-1}$.

2.2. Emissions - Concentration Model

The (In)tervention (M)odel for (A)ir (P)ollution (InMAP) is a multi-scale emissions-to-health impact model that can operate as an alternative to comprehensive air quality models for marginal emission changes (Tessum et al., 2015). This model estimates primary and secondary $\text{PM}_{2.5}$ concentrations and corresponding health exposure and impacts that result in from annual changes in precursor emissions. It operates on annual-average input parameters using transport, deposition, and reaction rates estimates from the chemical transport model (WRF-Chem) within an Eulerian modeling framework. The model allows exposure-dependent resolution with higher resolution (1 km grids) for urban areas and a lower resolution (48 km grids) for remote areas. In addition the model allows for a low computational cost. Finally, although the model currently covers the greater region of North America (U.S., Southern Canada, Northern Mexico, etc.), it has the potential of being extended from a regional to a global scale.

2.3. Study locations

Spatial $\text{PM}_{2.5}$ concentration estimates are determined for three distinct locations in the U.S. in Wisconsin (WI), New Jersey (NJ), and New York (NY). The three locations have been selected as to reflect various population density and precursor limiting conditions (Table 1). The WI location represents a region with abundance in NH_3 but limited in NO_x and SO_2 and low population around the source. The NJ location represents a region with abundance in NO_x and SO_2 but limited in NH_3 and high population around the source. The NY locations has no dominantly limiting conditions and hence represents the point of reference.

Table 1. Population data and atmospheric conditions around source locations

	State	County	Pop _{50 km}	Pop _{500 km}	Atm. conditions			
			(Million)	(Million)	NH_3	SO_2	NO_x	$\text{PM}_{2.5}$
1	WI	Clark	0.02	28.1	Medium	Low	Low	Medium
2	NY	Onondaga	0.5	84.8	Medium	Medium	Medium	Medium
3	NJ	Union	3.2	68.4	Low	High	High	High

2.4. Characterization factors

Having the approach of Humbert et al, (2011) as a foundation, characterization factors (CF) for primary and secondary $\text{PM}_{2.5}$ will be calculated to reflect the human health impact in DALYs per unit of precursor emitted ($\text{PM}_{2.5}$, NH_3 , NO_x , SO_2). CF can be estimated using the equation (Humbert, et al., 2011):

$$CF_i = iF_i \times ERF \times SF \quad (\text{Equation 2})$$

where i represents one of the precursors $\text{PM}_{2.5}$, NH_3 , NO_x , SO_2 , CF_i indicates the human health impact per mass precursor i emitted (DALYs/kgemitted), iF_i is intake fractions of $\text{PM}_{2.5}$ from precursor i emissions that is inhaled by the exposed population, ERF is in deaths/ unit mass concentration and indicate the $\text{PM}_{2.5}$ exposure-response factor, and SF is in DALYs/death indicating the severity factor. For ERF and SF are utilize the work by Gronlund et al., (2015) with an effect factor ($EF=ERF \times SF$) estimate of $78 \text{ DALY/kg}_{\text{PM}_{2.5} \text{ inhaled}}$.

2.5. Case study: milk consumption in the U.S.

These CFs are then applied to a milk case study that investigates the potential environmental and nutritional effects associated with a one serving increase in milk consumption so as to meet dietary guidelines (USDHHS and USDA 2015). For a comprehensive assessment of this potential dietary change, we employ the Combined Nutritional and Environmental Life Cycle Assessment (CONELCA) framework that consistently evaluates and compares environmental and nutritional effects of foods or diets (Stylianou et al., 2016). For this case study, the environmental assessment, emissions associated with one milk serving are determined based on a U.S. specific milk LCA (Henderson et al., 2013) and are then linked to human health impact in regards to global warming and PM_{2.5} following a tradition LCA approach. For the nutritional assessment, impacts and benefits associated with one serving milk increase over the current average consumption, such as colorectal cancer, stroke, and prostate cancer are estimated based on epidemiological studies, starting with the GBD, as described in Stylianou et al., 2016. As a sensitivity study, we assume that milk production occurs in our three study locations.

3. Results

3.1. Intake fraction summary

Figure 1 illustrates the spatial variations of human exposure as iFs resulting from NH₃ emissions in NY (A), WI (B), and NJ (C). In all locations, highest estimates are found as expected around the source of emission. iFs in WI show the greatest dispersion of exposure that expands from the northern Midwest to the eastern coastal regions of the U.S with an exposure travel distance of more than 1500 km. However, iF absolute values due to the WI emissions are substantially lower by about 2 orders magnitude when compared to iF due to NJ emissions. In NJ, a considerably smaller region is affected (150 km) by emissions since particles are transported off-land, but iFs are substantially higher than the other two regions, with estimates ranging between the order of 10⁻⁶ and 10⁻⁷ kg_{in}/kg_{emitted} close to the source where there is a high population density.

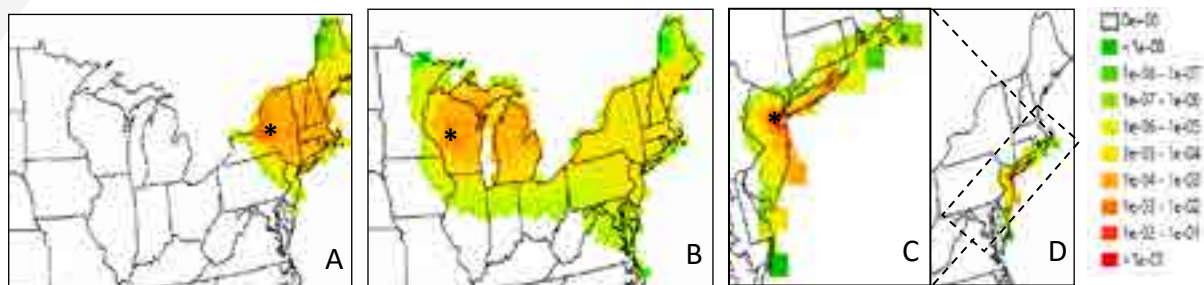


Figure 1. PM_{2.5} exposure estimates resulting from NH₃ emissions in NY (A), WI (B), and NJ (C & D) presented as iF_{PM_{2.5},NH₃} in parts per million (ppm) per grid cell. C is a close-up of D.

* indicates the location of the source of emission

Cumulative iFs are summarized in Figure 2 for each location of emission and precursor. There is a considerable spatial variation of exposure per precursor between locations, due to main factors of influence, i.e. population around the source and background precursor's concentrations (limiting conditions). NJ shows the highest iFs followed by WI and NY, reflecting population around sources. Of the total PM_{2.5} population exposure, nearly all (90%) happens on average within 20 km for NJ, 350 km for NY, and 1500 km for WI. Differences in atmospheric chemistry are also reflected by the fact that the iF_{PM_{2.5},NH₃} is 18 times higher than iF_{PM_{2.5},SO₂} in NJ (where NO_x and SO₂ are in abundance), but only 2 times higher in WI (where NH₃ is in abundance), suggesting that PM_{2.5} population exposures resulting from adding 1kg NH₃ are higher in NH₃-limited regions compared to NH₃-abundant regions.

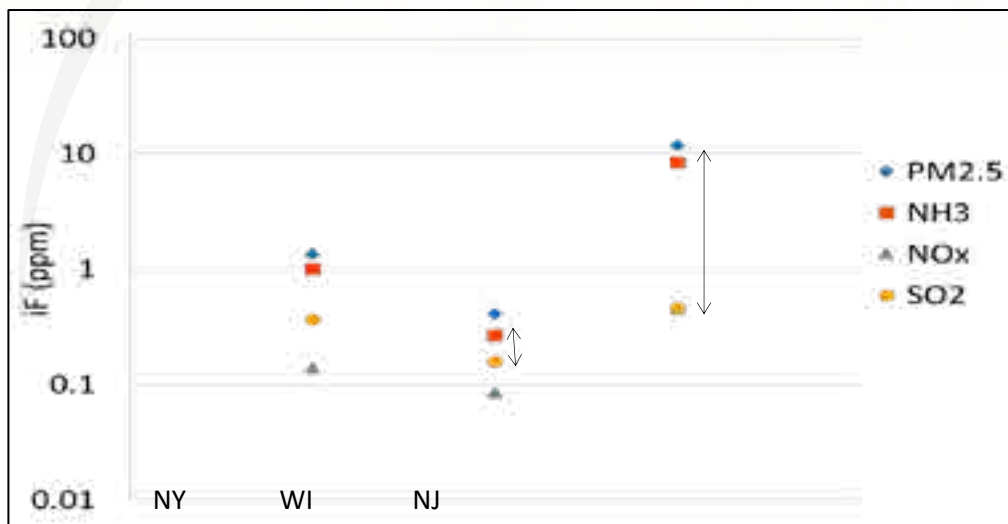


Figure 2. Primary and secondary PM_{2.5} Intake Fractions by precursor emissions in NY, WI, and NJ.

Combining the iF values reported here with an exposure-response (78 DALY/kg_{PM_{2.5} inhaled}) results in CFs (units: 10⁻⁵ DALY/kg precursor emitted) of 0.7–7.8 (NH₃), 0.6-3.5 (NO_x), 0.1-3.1 (PM_{2.5}), and 1.2-3.5 (SO₂).

3.2. Case study: Overall comparison

Figure 3 represents the overall comparison of environmental and nutritional effects associated with adding one serving of milk to the average U.S. milk consumption for the three locations of production. In regards to PM_{2.5}-related health impacts, about 63-73% of the impact is related to NH₃ emissions, with the highest contribution in NJ where milk production induces the highest total PM_{2.5}-related health impacts (9.1 μDALY/kg_{milk}). Overall, this dietary change leads to an overall health benefit due to nutritional benefits if production was to occur in NY or WI. However, if the corresponding production was to take place in NJ, PM-related health effects become substantial and comparable to nutritional benefits.

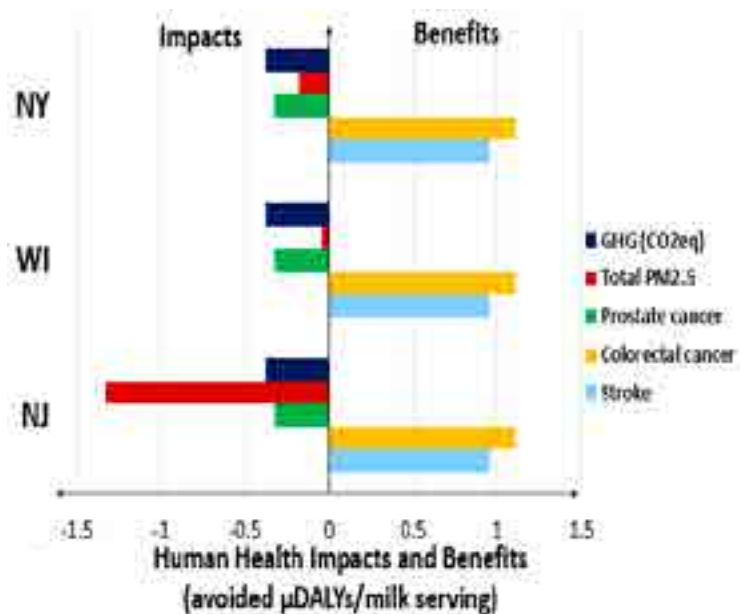


Figure 3. Comprehensive human health assessment for one milk serving increase per production locations.

4. Discussion

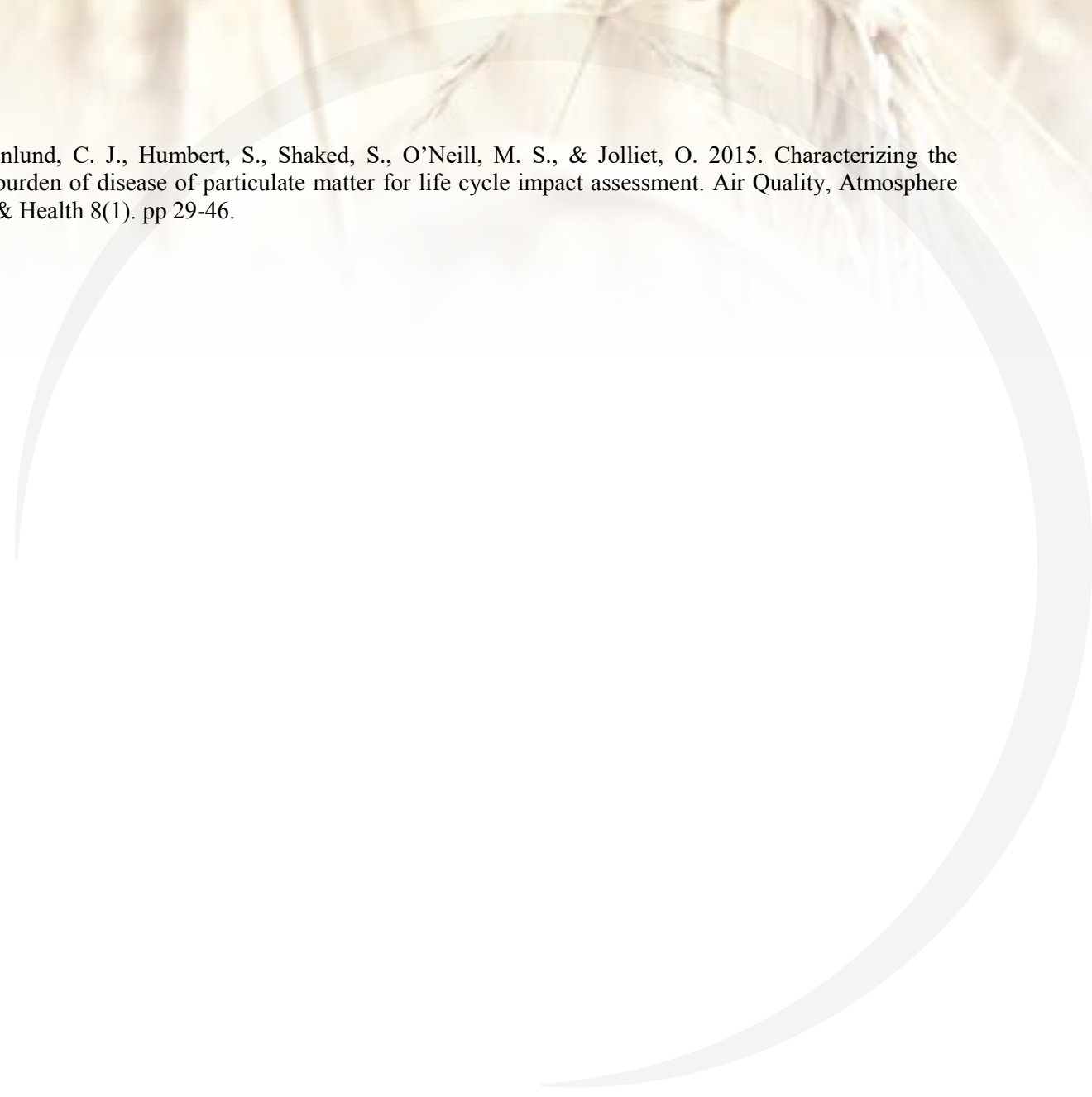
In this paper, we use the InMAP model to estimate spatially-explicit PM_{2.5} iF for PM-related precursors (PM_{2.5}, NH₃, NO_x, SO₂) in three distinct locations. This model allows us for resolution according to exposure and low computational intensity. Although we limited our analysis to three locations, WI, NY, NJ, our preliminary results support spatial variation of exposure that is linked primarily to population density. However, there are indication that atmospheric conditions, and in particular, NH₃ limiting conditions, influencing exposure estimates. Finally, our spatial estimates for PM_{2.5}, SO₂ and NO_x are in agreement with previous estimates in the literature (Humbert et al., 2011) while for NH₃ results suggest a higher maximum and wider range of estimates.

5. Conclusions

Our preliminary results support a spatial variation of secondary inorganic PM_{2.5} exposure in the U.S. linked to population density and atmospheric chemistry. Milk production in highly populated NH₃-limited regions has substantially higher PM_{2.5}-related health impacts to populations downwind than production in agricultural regions where NH₃ is in abundance. These findings is especially important for food-related decision makers since potential emission relocations might have considerable health effect to populations downwind. This research contributes to spatially-explicit CFs for the agricultural sector.

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73. The impact of subclinical ketosis in dairy cows on greenhouse gas emissions of milk production

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ABSTRACT

This study aimed to estimate the impact of subclinical ketosis (SCK) and related diseases in dairy cows on greenhouse gas (GHG) emissions of milk production. A dynamic stochastic Monte Carlo simulation model was developed and combined with life cycle assessment (LCA) to quantify the impact of SCK and related diseases on GHG emissions per ton fat-and-protein-corrected milk (kg CO₂e/t FPCM). The model simulates on cow level and the impact on GHG emissions was assessed from cradle-to-farm gate for the Dutch situation. Emissions of GHGs were increased on average by 18.4 kg CO₂e/t FPCM per case of SCK. Our study showed that LCA is a useful method to estimate the impact of diseases on GHG emissions and showed that reducing SCK and related diseases will reduce GHG emissions of milk production.

Keywords: disease, life cycle assessment, environment

1. Introduction

Subclinical ketosis (SCK) in dairy cows is a metabolic disorder that occurs around the calving period. In this period, the energy requirement of the cow can exceed her energy intake, resulting in a negative energy balance (NEB) (Grummer 1995). An NEB results in an increase of non-esterified fatty acids and beta-hydroxybutyrate levels in the blood. A cow has SCK when the beta-hydroxybutyrate level is higher than 1.2-1.4 mmol/l blood, but has no clinical signs (Raboisson et al. 2014). SCK results in reduced milk production and reproduction. Moreover, SCK increases the risk on related diseases, e.g. displaced abomasum, metritis, mastitis, lameness and clinical ketosis (Berge & Vertenten 2014, Raboisson et al. 2014). These related diseases also have an impact on milk production and reproduction and, in addition, may result in discarded milk and an increased risk on culling and dying. The prevalence of SCK in European dairy cows varies between 11 and 49% (Berge & Vertenten 2014, Suthar et al. 2013).

Diseases result in inefficient production, and, therefore, have an impact on the economic and environmental performance of dairy farming. Economic costs per case of SCK have been estimated at €257 (Raboisson et al. 2015) and \$289 (McArt et al., 2015). The impact of SCK on the environmental impact of dairy production, however, has not been analysed.

An important environmental problem is climate change. The livestock sector is responsible for about 14.5% of human induced greenhouse gas (GHG) emissions (Gerber et al. 2013), of which 30% is emitted by the dairy sector. With an expected increase in milk consumption of 58% in 2050 (FAO 2011, OECD/FAO 2013), reducing GHG emissions from the dairy sector becomes more important.

This study aims to combine the dynamics and consequences of diseases with life cycle assessment (LCA) to estimate the impact of SCK on GHG emissions of milk production. The method proposed can support decision making to improve health of dairy cows while reducing the environmental impact of food production.

2. Methods

A dynamic stochastic Monte Carlo simulation model was developed and combined with LCA to quantify the impact of SCK and related diseases on GHG emissions. The model simulates on cow level and the impact on GHG emissions was assessed from cradle-to-farm gate, based on parameters representing a Dutch dairy farm with day grazing. The model consists of four parts: parameters of the cow, dynamics of diseases, losses because of diseases, and estimation of GHG emissions. The model was developed in R (R_Core_Team 2013) and ran with 100,000 iterations.

Parameters of the cow

Each cow in the model received a parity (1-5+), based on an average herd composition. Based on the parity, a milk production, a body weight, and calving interval were attributed to each cow (CRV 2014, CVB 2012). The calving interval included a 60 days dry period. A lactation curve was utilised to

estimate the daily milk production (Wood 1967). Subsequently, energy requirement for maintenance, growth, pregnancy and fat-and-protein-corrected milk (FPCM) were estimated and summed per cow per lactation (CVB 2012).

Dynamics of diseases

Diseases that occur in the first 30 days after calving were included. First, cows had a probability to get SCK, which was dependent on her parity. Second, cows with SCK had an additional probability to get one related disease: mastitis, metritis, displaced abomasum, lameness or clinical ketosis. The additional probability was the difference between the probability of a cow with and a cow without SCK on getting the disease (Berge & Vertenten 2014). Thus, a cow could have no disease, SCK only, or SCK and a related disease. Cows without SCK (and related diseases) were excluded from further analyses. All probabilities were based on Berge and Vertenten (2014). Total disease incidence among the 5 parities in the first 30 days was, 25.0% for SCK, 12.5% for mastitis, 10.2% for metritis, 4.0 % for displaced abomasum, 4.5% for lameness, and 1.6% for clinical ketosis (Berge & Vertenten 2014, Bruijnjs et al. 2010, Raboisson et al. 2014). Third, cows with SCK and a related disease had a probability on culling and dying (Bar et al. 2008), together called removal. The events of SCK, a related disease or getting removed were determined with discrete distribution functions.

Losses because of diseases

Cows with SCK (and related diseases) had milk losses during one lactation because of reduced milk production, discarded milk, and removal of the cow. Milk production of cows with SCK (and related diseases) during one lactation increased because of an extended calving interval.

Cows with SCK (and related diseases) had a reduced milk production. The reduction of milk production (%/d) and duration of reduced milk production were disease specific. Cows with SCK only had a reduced milk production during the first 30 days, whereas cows with SCK and a related disease also had a reduced milk production after day 30 (Gröhn et al. 2003, McArt et al. 2012, Raizman et al. 2002, Seegers et al. 2003). Cows with SCK and a related disease, except for lameness, were always treated with antibiotics for a disease specific period and milk was discarded during treatment and the withdrawal period. The calving interval was extended for cows with SCK (and related diseases). The extension of calving interval (in days) was disease specific. A cow was removed at day 30 and replaced with a heifer with average production parameters.

Subsequently, net energy requirement and the FPCM per lactation of cows with SCK (and related diseases) were estimated.

Estimation of GHG emissions

Emissions of carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) were estimated for processes along the dairy production that were affected by the consequences of SCK, including feed production, enteric fermentation, and manure management. Emissions of GHGs were estimated for a cow with SCK (and related diseases) and without SCK for one lactation and were expressed as the sum of kg CO₂ equivalents (100 years' time horizon) (Myhre 2013) per ton of FPCM (kg CO₂e/t FPCM). The difference between a cow with SCK (and related diseases) and without SCK was the impact of SCK (and related diseases) on GHG emissions.

A diet for the summer and winter period was composed of concentrates and roughages (grass, grass silage, maize silage) (CBS 2014, Nevedi 2012, 2013, 2014, 2015). Roughages were produced on the farm and concentrates were purchased. Feed intake (kg DM/ cow) was estimated based on the weighted average energy content of the diet (MJ/ kg DM) and the energy requirements of the cow (MJ/ cow). First, the emissions related to feed production (kg CO₂e/ kg DM) were estimated (transport, crop cultivation, processing, feed mill) (Vellinga et al. 2013). Emissions from land use and land use change (LULUC) related to feed production were included based on Vellinga et al. (2013). Second, emissions related to enteric fermentation were estimated based on feed specific emission factors (kg CH₄/ kg DM) (Vellinga et al. 2013). Third, emissions from manure were estimated. Direct and indirect N₂O (i.e., N₂O derived from volatilization of ammonia (NH₃) and nitrogen oxides (NO_x) and from leaching of nitrate (NO₃)) and CH₄ emissions from manure in stables, storage, and on grass were included and based on national inventory reports (De Mol 2003, De Vries 2011, Oenema 2001,

Schils 2006) and IPCC (2006). Finally, all emissions were summed and divided by the total amount of FPCM in tons.

System expansion was applied to account for the production of meat from removed cows. The method has been shown to be a solid method for the evaluation of environmental impacts related to changes in milk production (Zehetmeier et al. 2012). Meat from dairy cows, except for cows that died, was assumed to replace meat from chicken, pork or beef. Emissions of GHGs related to the production of meat from chicken, pork or beef were estimated per kg edible product (Van Middelaar et al. 2014a) and weighted to an average emission factor based on the average consumption of pork, chicken and beef in OECD countries (OECD 2015).

Cows that were removed before the end of parity 5 were assumed to be removed to early, resulting in additional GHG emissions for raising extra heifers. First, we calculated the emissions of GHGs from raising a heifer based on system expansion, to determine how much of these emissions were related to milk production. Emission were estimated as the difference between emissions of raising a heifer and emissions of the meat to be replaced by meat from that heifer. Second, emissions related to raising were depreciated over the total amount of milk that was produced by the cow at the moment of removal. For cows that were removed at parity 2+, average milk production levels were assumed for all previous parities.

3. Results

The emissions of GHGs increased on average by 18.4 (median 8.5) kg CO₂e/t FPCM per case of SCK, but variation was large. The increase in emissions was lowest for cows that had SCK only (≤ 11.4 kg CO₂e/t FPCM) followed by cows with SCK and a related disease (16.8-55.5 kg CO₂e/t FPCM). Cows that were removed showed the highest increase in emissions per case of SCK (≥ 56.7 kg CO₂e/t FPCM) (Figure 1).

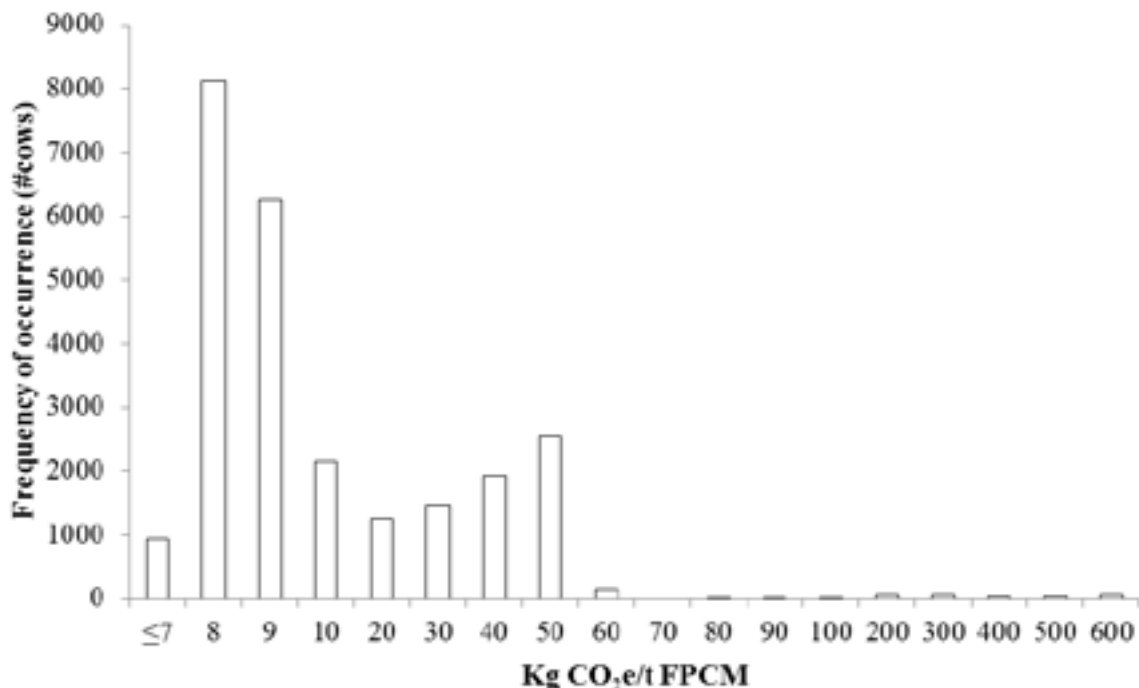


Figure 1. Extra kg carbon dioxide equivalents (CO₂e) per ton fat-and-protein-corrected milk (t FPCM) per case of subclinical ketosis (SCK), and frequency of occurrence. Average increase in emissions was ≤ 11.4 kg CO₂e/t FPCM for cows with SCK only, 16.8-55.5 kg CO₂e/t FPCM for cows with SCK and a related disease, and ≥ 56.7 kg CO₂e/t FPCM for cows that get culled or died.

4. Discussion

The average impact of SCK (and related diseases) on GHG emissions was 18.4 kg CO₂e/t FPCM per case, but showed large variation. Most cows with SCK (and related diseases), however, had SCK only, which had a lower increase in GHG emissions than the average cow with SCK (and related diseases). Cows with SCK and a related disease and cows that were removed had a higher increase in GHG emissions and had an important impact on the average increase of GHG emissions per case of SCK. Cows with SCK only had a reduced milk production and an extended calving interval, whereas cows with SCK and a related disease had higher reduced milk production, a longer extended calving interval and also had discarded milk. Cows that were removed had a lower milk production and additional emissions for breeding replacement heifers, which were not needed if the dairy cows were not removed. Literature shows a huge variation in disease probabilities, relation of SCK with other diseases, removal probabilities, losses related to diseases, and GHG emissions of flows. Therefore, the impact of cows with SCK (and related diseases) on GHG emissions might differ between farms.

Most LCA studies use economic allocation (De Vries & de Boer 2010) to allocate emissions to the different outputs in case of a multiple output system. Estimating the economic allocation factor of milk based on one lactation requires additional assumptions about e.g. longevity of the cow. In this study, therefore, we applied system expansion. We assumed that milk production was the main purpose of dairy farming and that the meat of dairy cows substitute the production of beef, pork or chicken. Assumptions regarding the type of meat that is replaced can have an important impact on the result (Van Middelaar et al. 2014a). Therefore, our GHG calculations were based on the average consumption pattern of meat products in OECD countries. Future studies can include multiple types of meat products in order to show the potential range in results.

We estimated the impact of SCK (and related diseases) on GHG emissions for one lactation, because there was no data available for the incidence of diseases over multiple lactations. Estimating the impact of diseases during one lactation only, however, is a common method to estimate the economic costs of diseases (Bruijnijis et al. 2010, Huijps & Hogeveen 2007, Inchaisri et al. 2010).

Based on an average emission factor of 1,000 kg CO₂e/t FPCM (De Vries & de Boer 2010), a case of SCK increased GHG emissions on average by 1.8%. The impact of SCK on GHG emissions at herd level depends on the disease incidence of the herd. Complete eradication of SCK in a herd might not be achievable, but a minimum incidence of 10% of SCK at herd level might (Raboisson et al. 2015). Reducing the incidence of SCK from 25% to 10% at herd level, therefore, might have, on average, a minor impact on GHG emissions. On dairy farms with a higher incidence of SCK and related diseases and a higher removal rate of cows, however, reducing SCK and related diseases might have a higher impact on GHG emissions. In addition, our study only estimated the impact of SCK and the additional impact of related diseases in the first 30 days after calving. Including more diseases during the whole lactation might result in a higher impact of diseases on GHG emissions.

Examples of other mitigation options to reduce GHG emissions in the dairy sector are feeding and breeding strategies. Different feeding strategies showed on average a higher reduction in GHG emissions (9-32 kg CO₂e/t FPCM) (Van Middelaar et al. 2014b) than reducing SCK. These strategies, however, reduced the income of the farmer, whereas reducing SCK will increase the income of the farmer (McArt et al. 2015, Raboisson et al. 2015). In addition, farmers might prefer a reduction of SCK above a different feeding strategy, because this may be easier from a management perspective. Increasing the milk yield with 698 kg/year per cow and the longevity with 270 days per cow showed a reduction of 27 and 23 kg CO₂e/t FPCM (Van Middelaar et al. 2014a). Achieving this production by breeding, however, might take several years, whereas reducing SCK might not.

This study combined the dynamics and consequences of diseases with LCA to estimate the impact of SCK on GHG emissions of milk production. The method proposed can also be used to evaluate the impact of other diseases and can be extended to other environmental impact categories.

5. Conclusions

The average increase of GHG emissions per case of SCK was 18.4 kg CO₂e/t FPCM. The increase in emissions varied from ≤ 11.4 for cows that had SCK only, to ≥ 56.7 for cows that were removed. Our study showed that LCA is a useful method to estimate the impact of diseases on GHG emissions and that reducing SCK and related diseases will reduce GHG emissions of milk production.

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215. Use of process-based models to quantify life cycle nutrient flows and greenhouse gas impacts of milk production: influence of beneficial management practices and climate change scenarios

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Assessing and improving the sustainability of dairy production systems is essential to secure future food production. This requires a holistic approach that reveals trade-offs between emissions of the different greenhouse gases (GHG) and nutrient-based pollutants and ensures that interactions between farm components are taken into account. Process-based models are essential to support whole-farm mass balance accounting, however, variation between process-based model results may be large and there is a need to compare and better understand the strengths and limitations of various models. Here, we use a whole-farm mass-balance approach to compare five process-based models in terms of major nutrient (N, P) flows and greenhouse gas (GHG) emissions associated with milk production at the animal, farm and field-scale (figure 1). Results of these models are then used as input for a farm Life Cycle assessment of a US farm of located in NY state with 2436 animals. Results show that predicted whole-farm, global warming impacts were very similar for the two whole farm models with a predicted global warming impact of approximately $1.1 \cdot 10^7$ kg CO₂eq./year for both models and a dominant contribution of enteric CH₄ emissions (figure 2). Model predictions were also highly comparable, i.e. within a factor of 1.5, for most nutrient flows related to the animal, barn and manure management system, including enteric CH₄ emissions, and NH₃ emissions from the barn. In contrast, predicted field emissions of N₂O and NH₃ to air, and N and P losses to the hydrosphere, were very variable across models. This indicates that there is a need to further our understanding of soil and crop nutrient flows and that measurement data on nutrient emissions are particularly needed for the field. A systematic analysis of Beneficial Management Practices (BMPs) and climate change scenarios will be presented, including comparison between feed rations scenarios (incl. high forage with Neutral Detergent Fiber Digestibility), between manure processing and storage systems (incl. digester vs lagoon), and crop & soil management practices (incl. all manure applied in Spring or in season, no till, all manure injected) and 15 climate change scenarios including RCP 2.5, 4.5, 6.0 and 8.5 (figure 3). The whole-farm mass-balance approach is advocated as an essential tool to assess and improve the sustainability of dairy production systems.

a. Empirical N balance

b. Empirical P balance

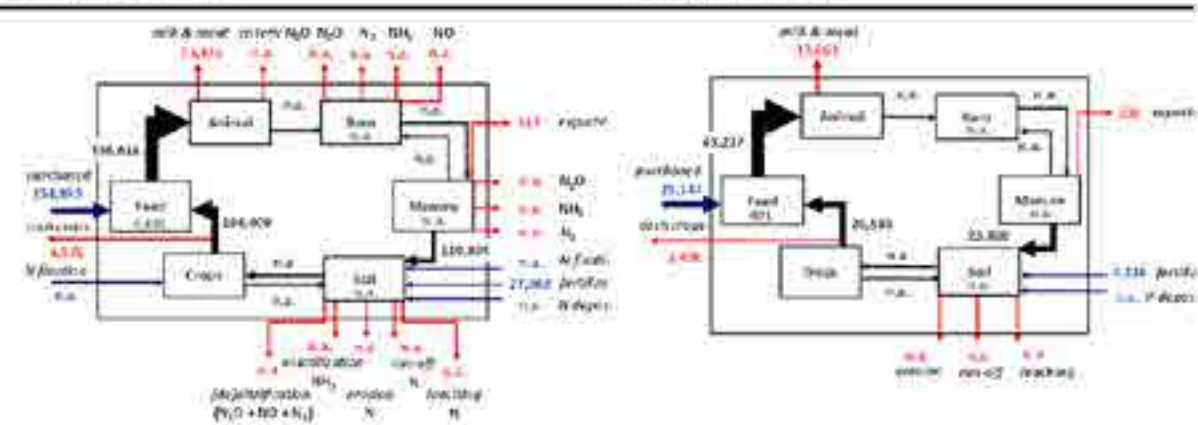


Figure 1. Whole-farm nutrient mass-balance: predictions from IFSM for N- and P-balance for the farm case study.

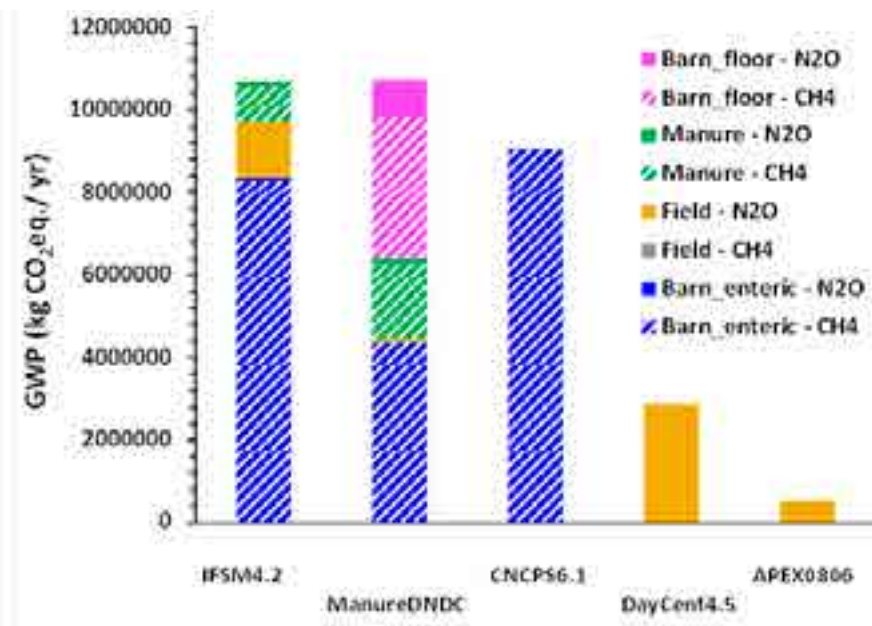


Figure 2. Predicted global warming potentials per farm component (in kg CO₂ eq./year)

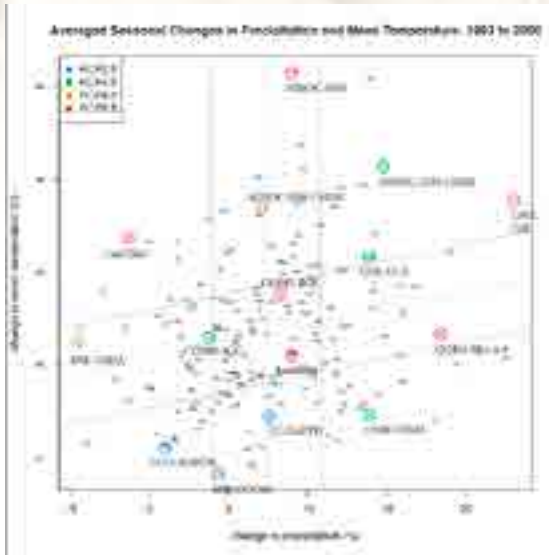


Figure 3. Change in mean temperature and in precipitation for various climate change scenarios

ORAL PAPERS

CROPS AND FRUITS

3. Crops and Fruits.

24. South African maize production: Mitigating environmental impacts through solar powered irrigation

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ABSTRACT

Agriculture is among the largest contributors to global greenhouse gas emissions. Clean technologies, such as renewable energies, have the potential to significantly reduce these environmental repercussions of agriculture. Countries like South Africa have a coal intensive electricity mix, as well as high solar irradiation and a dry climate which is why agricultural crops are produced under fossil energy intensive irrigation. At the same time, the high solar irradiation could be used for the generation of photovoltaic electricity as a renewable power supply for irrigation. A joint research project between the University of Cape Town and the Zurich University of Applied Sciences quantified the environmental impacts of South African maize production (*Zea mays*) and the improvement potential of maize irrigation with photovoltaic electricity by means of life cycle assessment (LCA).

The LCA includes the whole value chain of maize production from cultivation to storage in a silo for six months, respectively with a functional unit of one kilogram of maize at silo storage produced either on dry land or under irrigation. Electricity consumption for irrigation was identified as an environmental hotspot in the impacts related to greenhouse gas emissions from maize production. Therefore, clean electricity would be the starting point to reduce the carbon footprint of South Africa's maize. We calculated that replacement of South African electricity mix with photovoltaic electricity in the maize irrigation can reduce environmental impacts by up to 48 %. The calculated greenhouse gas emissions per kilogram of maize on dry land without irrigation, under irrigation and under irrigation using photovoltaic electricity, are 0.51 kg CO₂-eq. and 0.81 kg CO₂-eq. and 0.56 kg CO₂-eq., respectively, with a potential reduction of 31 % if the electricity is supplied from photovoltaics compared to the conventional fossil electricity mix. The analysis of further indicators reveals a reduction for freshwater ecotoxicity and human toxicity of carcinogenic substances. The irrigation of a maize field of one hectare consumes 1'900 kWh of electricity per year, which, in turn, requires a solar power plant with an area of 9 m². We computed that a total area of 199 ha of solar panels would suffice to produce the total electricity requirement of the current maize production area under irrigation. This corresponds to more than approximately 100'000 t CO₂-eq. saved per year.

Compared to data representing maize production in the United States and in Switzerland, South African maize production has a higher global warming potential per kilogram of maize due to lower yields in South Africa.

The replacement of the South African electricity mix in the irrigation with electricity from photovoltaics has proven to be an effective clean technology to reduce environmental impacts associated with maize production in South Africa. Compared to the irrigated field area, land use for PV panels is almost negligible and is therefore no limiting factor in the implementation of irrigation using photovoltaic electricity.

Keywords: Photovoltaic, greenhouse gas emissions, crop production, emerging economy

1. Introduction

Maize is the major feed grain and the most important staple food for the majority of South Africa's population. South African maize production is largely dependent on sufficient and timeous summer rainfall and can reach 12 million tonnes per year, planted in an area of over 2.6 million hectares (Grain SA, 2014). Maize is produced throughout South Africa. The provinces Free State, Mpumalanga and North West are the largest producers, accounting for approximately 83 % of the total production.

A distinction is made between maize production for feed and for human diet. White grain maize is primarily produced for human consumption and is on average 60 % of the total maize production in South Africa. Yellow maize is mostly used for animal feed and comprises about 40 % of the total South African maize production (DAFF, 2014).

In 2013 there was a shift towards higher maize production for animal feeding at the expense of maize production for human diet. The production volume of maize for feeding was 5'933'100 t spread over an area of 1'164'000 ha. The production of white grain maize for human diet is estimated to 5'580'300 t planted on 1'617'200 ha (Grain SA, 2014).

Agriculture is among the largest contributors to global anthropogenic non-CO₂ greenhouse gas emissions, accounting for 56 % of emissions in 2005 (IPCC, 2014). While animal production contributes most by methane emissions, arable production is associated with dinitrogen monoxide emissions (Johnson *et al.*, 2007).

In South Africa crop irrigation is typically operated with fossil fuel based energy. Using green and clean technologies, such as renewable energies, some of the environmental impacts of agriculture can be reduced.

In a joint research project between the University of Cape Town and the Zurich University of Applied Sciences the environmental impact, as well as the potential for improvement through the use of clean technologies, was assessed for South Africans maize production.

2. Goal and Scope

In order to define optimization strategies through the use of clean technologies, a Life Cycle Assessment (LCA) of the status quo of the grain maize production in South Africa for human diet was performed, following the ISO 14040 and 14044 standards (ISO, 2006a; 2006b). The system boundaries of the LCA include the whole value chain of grain maize production, from seed bed preparation, to maize cultivation and harvesting, to storage of the harvested maize in a concrete silo for approximately six months. The LCA considers the production and application of fertilizers and pesticides, as well as the particular use of irrigation water. Land use, direct field emissions, the production and use of tractors and agricultural machines and consequential diesel consumption of tractors in use as well as transport are also taken into account. The most relevant production data, including production area, application of fertilizers and pesticides, diesel consumption and yield, were taken from planning models provided by Grain SA, the national representing and consulting institution for grain producers in South Africa (Grain SA, 2014). The planning models, which give a detailed compilation of any costs in the maize production, cover the circumstances of grain maize production in three different regions of South Africa: Eastern Highveld, North West, and Central and Northern Free State. As a functional unit one kilogram of grain maize at silo storage produced either on dry land or under irrigation for human diet was defined.

South Africa has a high level of solar irradiation and a dry climate, which leads to an agricultural crop production under irrigation. Electricity supply for irrigation is a coal intensive electricity mix, leading to high environmental impacts, which are associated with irrigation. The goal of this study was to quantify the reduction potential by using solar irradiation for the generation of photovoltaic electricity as a renewable power supply for irrigation.

The environmental impacts were assessed using five different impact indicators, namely climate change according to IPCC (2013), the cumulative non-renewable energy demand according to Hirschier *et al.* (2010), acidification and freshwater eutrophication according to the European Commission (2011), and human toxicity (cancer) according to Rosenbaum *et al.* (2008).

3. Life Cycle Inventory

In 2013, 82'000 ha with grain maize for human diet were irrigated in South Africa. Under irrigation yields are higher but also the use of fuel and need for fertilizers increase. An overview of the key inventory data for South African maize production for human diet on dry land and under irrigation is given in Table 1. Transport distances are estimated using online distance calculators for sea and land routes.

Field emissions to air as nitrous oxides (N₂O) and ammonia (NH₃) and leaching of nitrate to ground water (short and long term) are calculated according to Meier *et al.* (2012; 2014). Phosphate emissions to ground water through leaching and to surface waters through run-off and water erosion, as well as emissions of heavy metals to soil are modelled according to Nemecek *et al.* (2007). In addition, all pesticides applied for maize production were assumed to end up as emissions to the soil (Nemecek *et al.*, 2007). Background data for the life cycle inventories were taken from the international ecoinvent v3.2 database using the system model "allocation, recycled content" (ecoinvent Centre, 2015).

Seed, pesticides and fertilizers are transported by lorry from retailers to the farm (650 km) and harvested maize is transported from the field to the farm and further to a silo co-operation by tractor (5 km and 40 km, respectively). Tractor and agricultural machines are imported from the United States, Canada, Europe and Japan, whereas 1'700 km and 650 km are assumed representative for inland transportation by lorry in the export land and in South Africa, respectively. Overseas transport

is an average distance from the mentioned export countries above to South Africa, accounting for 14'400 km by a transoceanic ship. The life cycle inventories and the impact assessment were modelled with the SimaPro software v8.2 (PRé Consultants, 2016).

Table 1: Summary of life cycle inventory of grain maize for human diet produced on dry land and under irrigation in South Africa (ZA), representing maize production in 2013

	Unit	Grain maize, dry land	Grain maize, irrigated
Production area	ha	1'519'557	85'924
Yield	kg/ha	3'770	8'134
Seed	kg/ha	10.0	23.1
Fertilizers			
lime	t/ha	0.9	1.0
NPK	kg/ha	84.7	282
manure	t/ha	2.5	2.5
Pesticides			
herbicides	kg/ha	0.5	0.0001
insecticides, fungicides	L/ha	7.7	2.2
Diesel consumption	L/ha	71.9	79.7
Irrigation			
water	m ³ /ha	0	7'000
electricity	kWh/ha	0	1'900
Transports	km/t	281	179

South Africa, as an emerging economy, is used to irrigating its crops with fossil fuel based energy. Although only 10 % of the maize crop is produced under irrigation (DAFF, 2014), the potential to reduce environmental impacts by using green and clean technologies for irrigation is worth considering in more detail.

The main irrigation system in South Africa is a Centre Pivot system. In ecoinvent, inventory data for Centre Pivot irrigation systems are not available, therefore a new dataset was generated according the South African conditions concerning electricity supply and water use, based on personal communication with Jan Coetzee, extension officer at The South African Breweries.

A new inventory was also established for photovoltaic electricity. The dataset is based on a 570 kWp open ground installation with multi crystalline silicon panels. Annual yield is adapted to the main growing regions for maize (North West and Free State), and accounts for 1'770 kWh/kWp (European Commission, 2012). According to the IEA PVPS Methodology Guidelines, a life time of 30 years and an annual degradation of 0.7 % have been assumed (Fthenakis *et al.*, 2011).

4. Life Cycle Impact Assessment

Production of fertilizers, direct field emissions, diesel consumption and (if present) electricity consumption for irrigation were identified as environmental hotspots in the South African grain maize production (Figure 1). The global warming potential (GWP) of irrigated grain maize in South Africa amounts to 0.82 kg CO₂-eq. per kilogram of grain maize and is 39 % higher than the global warming potential of grain maize produced on dry land (0.50 kg CO₂-eq. per kilogram grain maize). The higher yields of irrigated maize cannot compensate for the additional electricity and diesel consumption for irrigation (Figure 1). If irrigation is supplied by the South African electricity mix, the contribution of irrigation to the overall GWP accounts for 36 %. The replacement of the South African electricity mix in the irrigation with electricity from photovoltaics results in a reduction of 0.27 kg CO₂-eq. per kilogram of grain maize, which is equivalent to 33 % (Figure 1). The GWP of irrigated grain maize using photovoltaic electricity is similar to grain maize production on dry land.

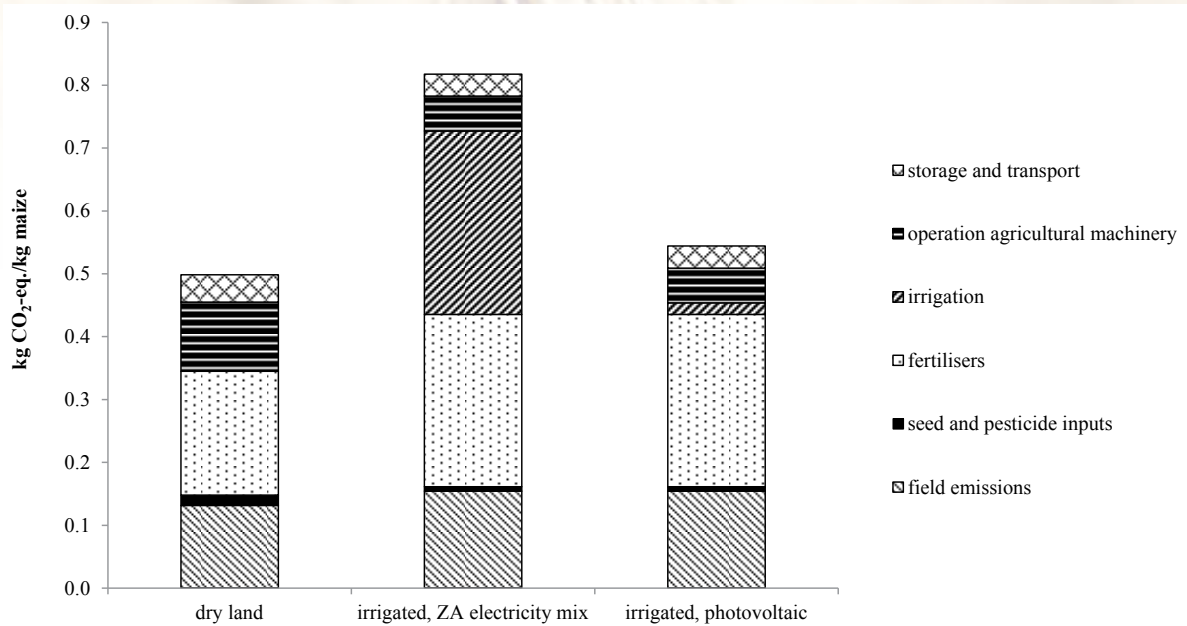


Figure 1: Global warming potential in kg CO₂-eq. of the production of 1 kg of grain maize on dry land (left-hand column) and under irrigation. Irrigation is either supplied by the South African electricity mix (central column) or by electricity from photovoltaics (right-hand column).

The analysis of further indicators reveals a significant reduction for non-renewable energy demand (fossil and nuclear) of 47 %. Acidification, human toxicity of carcinogenic substances and freshwater eutrophication are reduced by 21 %, 19 % and 13 %, respectively (Figure 2). However, not all environmental and human domains are equally affected. Freshwater ecotoxicity, marine eutrophication, land use and human toxicity of non-carcinogenic substances remain almost unaffected by the change of electricity supply.

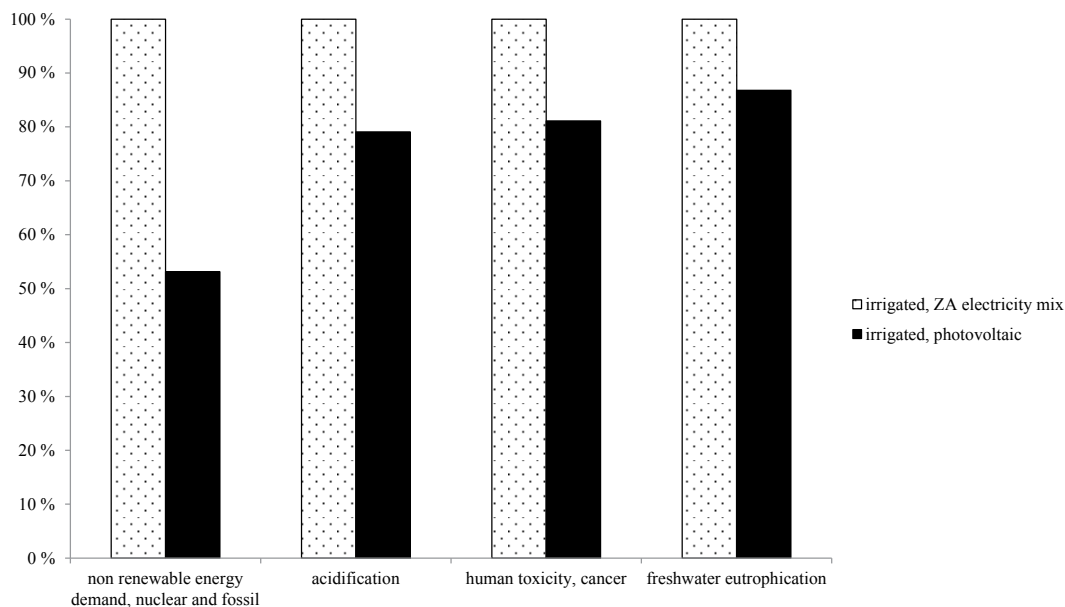


Figure 2: Comparison of selected environmental impacts for irrigated grain maize for human diet in South Africa using the South African electricity mix or electricity from photovoltaics for irrigation. All results are normalized by the results for maize irrigated using the national grid mix.

The high environmental impacts of the South African electricity mix are due to its composition: 88 % of the electricity is supplied by hard coal power plants and 5 % by nuclear power plants (ecoinvent Centre, 2015). By eliminating contributions of electricity with high environmental impacts, overall environmental impacts can be considerably reduced.

The irrigation of a maize field of one hectare consumes 1'900 kWh of electricity per year, which, in turn, requires a solar power plant with an area of 9 m². This means that in order to supply the power used for the irrigation of a field by means of photovoltaic panels, an area of only 0.09 % of the irrigated maize field is required.

5. Interpretation

We estimated that a total area of 76 ha of solar panels would be needed to produce the electricity to supply the current grain maize production area for human diet under irrigation (85'924 ha). This corresponds to about 190'900 t CO₂-eq. saved per year. Including the maize production under irrigation for feed, which covers a production area of 139'964 ha in South Africa, a total of more than 502'000 t CO₂-eq. could be saved per year (additional 311'200 t CO₂-eq. from feed). The required solar panel area to supply the total current maize production area under irrigation in South Africa, including maize for human diet and for feed, would increase up to 199 ha or 0.09 % of the irrigated maize area. The calculations about land use revealed that the installation area of PV panels is almost negligible compared to the irrigated production area. Consequently, land use is no limiting factor in the implementation of photovoltaics to irrigate the whole maize production throughout South Africa.

Compared to inventory data in ecoinvent, the modelled South African grain maize inventory has a higher global warming potential (0.82 kg CO₂-eq./kg maize) than maize produced in the United States (0.54 kg CO₂-eq./kg maize), in Switzerland (0.51 kg CO₂-eq./kg maize) or interpolated in global maize production (0.60 kg CO₂-eq./kg maize). System boundaries of the data inventories in ecoinvent are comparable to the maize inventory in the present study, including inputs of seeds, fertilizers, pesticides and irrigation water, as well as machine operations, field emissions and transport, and are therefore not crucial for the discrepancies regarding global warming potential. In contrast to our study, drying of grains at the farm is included, but not storage in a concrete silo. In South Africa, yields of 8'134 kg per hectare are lower than the yields of 9'315 kg per hectare gained in the United States, in Switzerland or in global maize production, leading to the higher greenhouse gas emissions per kilogram of maize, as mentioned above.

A further clean technology process, which is not yet widely used, is wireless sensor irrigation networks (WSIN). WSIN involve soil moisture sensors, specialized software interfaces and decision-supporting tools, which allows a more efficient and precise 'water on demand' irrigation. Water-saving technological processes are very important, especially where water is scarce and yield is highly dependent on proper irrigation, as is the case in South Africa. Majsztzik *et al.* (2013) show a decline in average water consumption of approximately 50 % compared to traditional irrigation in ornamental plant production in the USA. A reduction in fertilizer application, nutrient runoff and related greenhouse gas emissions can be attributed to the implementation of wireless sensor irrigation networks in horticulture. Further study is required to estimate the reduction potential through the implementation of WSIN in agronomic crops such as maize in open field production. By applying a combination of WSIN and renewable energy, the potential for mitigating environmental impacts could possibly be maximized.

6. Conclusion

The replacement of the South African electricity mix in irrigation with photovoltaic electricity has proven to be an effective clean technology to reduce environmental impacts associated with irrigated maize production. As the calculations showed, land use is no limiting factor for installing PV panels in order to generate solar energy for the large scale irrigation of maize fields in South Africa.

Depending on the impact indicator, up to 47 % of the environmental impacts can be saved with irrigation supplied by photovoltaic electricity compared to energy supply by fossils. The

environmental benefit would be even higher if renewable energy were expanded to further irrigated crops and additional clean technology processes like WSIN were implemented in South Africa.

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259. An evaluation of energy use within organic farming systems

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This review investigated the energy efficiency of a range of organic farming systems through a structured review of 35 LCA-based studies. Comparisons were made in relation to the amount of fossil energy required per unit of product and per unit of land. Energy output/input ratios were also compared for each product category.

Organic systems were found to use less fossil-fuel energy on a unit of land area basis for most crop and livestock products, although the energy use associated with imported compost could lead to higher inputs per hectare for some horticultural crops (Figure 1). Results were more variable per unit of product where lower yields and higher energy requirements for weed control could make some organic cropping systems perform worse (e.g. potatoes and tomatoes under glass, Figure 2). In addition, higher feed conversion ratios and mortality rates make some organic poultry systems less efficient per unit of output. For most grazing systems, organic farming resulted in lower energy use per unit area or weight of product. This results from using clover and other forage legumes in leys, which leads to more efficient forage production compared to the conventional Haber-Bosch based practice. Organic dairy production also tended to require less energy use per litre of milk produced, due to greater energy efficiency in the production of forage and reduced reliance on concentrates.

Lower levels of inputs were found to lead to higher output/input ratios in most organic systems, although with some exceptions (e.g. organic top fruit production and stockless arable systems, Figure 3). In many cases, organic farmers' diesel requirements were comparable to conventional.

Overall, the review found that organic farming systems have potential to contribute towards more energy efficient agriculture, but with lower yields. The review also highlighted that organic systems do not offer a radical alternative, as they are still reliant on fossil fuel sources and the differences in energy use per unit of product were often marginal. Organic methods could still be applied to increase the efficiency of the agriculture sector as a whole, although

energy use is only one aspect of sustainability and trade-offs may occur (e.g. between fossil energy and water use) for some products.

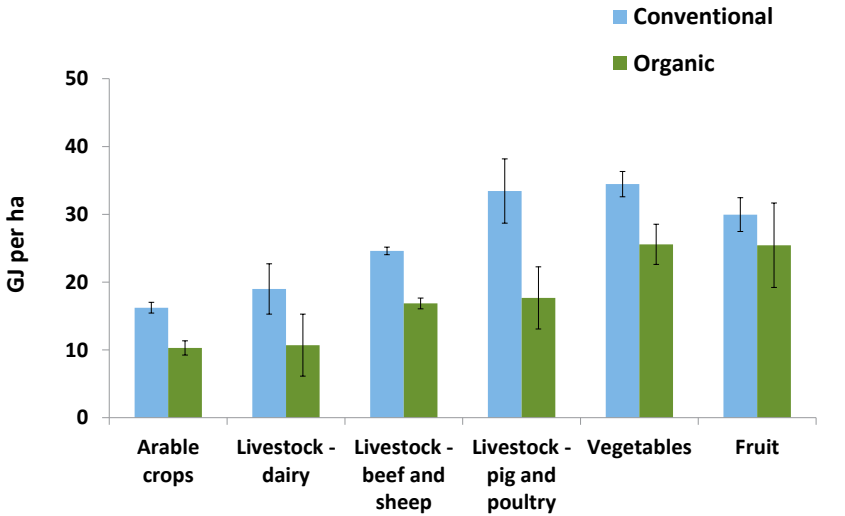


Figure 3: Average of all studies in gigajoules (GJ) per hectare with standard error

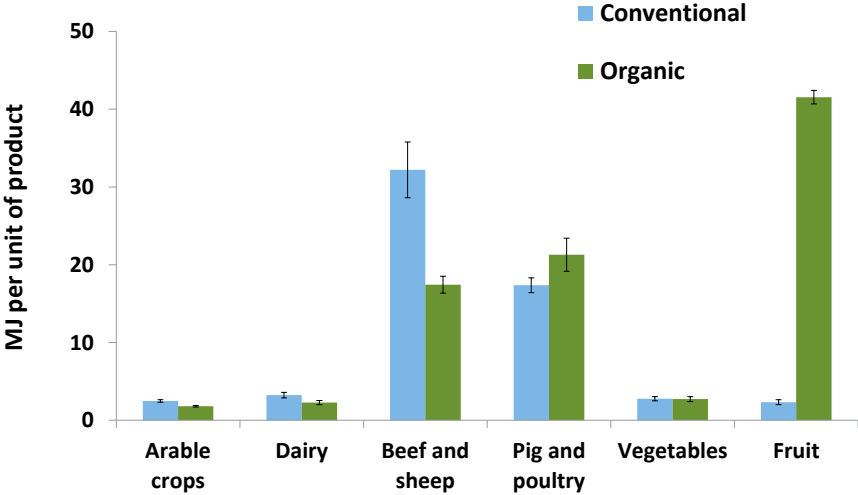


Figure 4: Average of all studies in megajoules (MJ) per unit of product with standard error

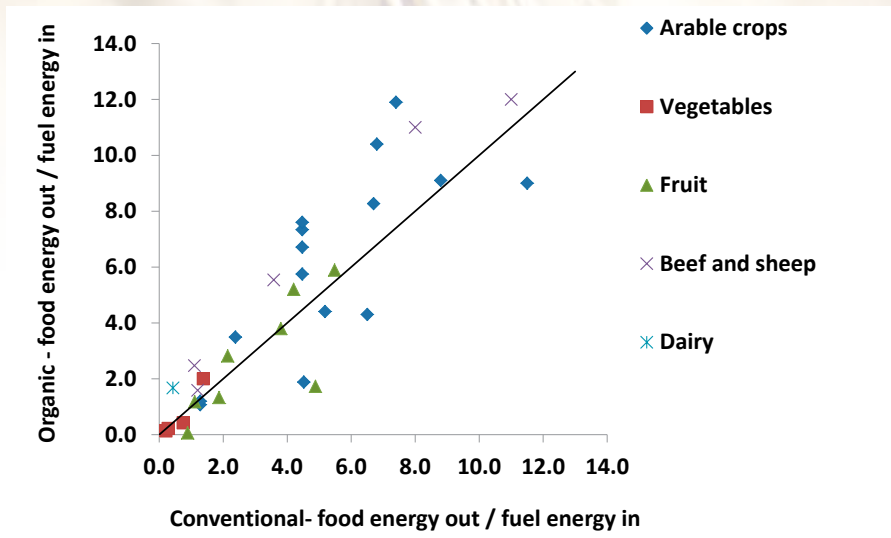


Figure 5: Average of energy ratios (energy output divided by input)

292. Life Cycle Assessment of Thai Organic Hom Mali Rice Farming to Support Policy Decision on Area Expansion

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ABSTRACT

Thailand has the strategic national policy to promote organic Hom Mali (Jasmine) rice farming in its quest to become the regional hub for organic agricultural products. The target for organic rice is set to be increased from a modest 0.18% to an ambitious 20% of the total rice production area in 2021. Different scenarios based on varying cultivation area ratios of organic rice farming in the North and North East regions of Thailand were proposed and their potential environmental impacts were evaluated by using life cycle assessment (LCA). The impact categories of interest included Climate change, Terrestrial acidification, Freshwater eutrophication, Terrestrial ecotoxicity, and Freshwater ecotoxicity and the impact assessment method applied was ReCiPe. In general, the rice yield in the North was higher than that in the North East. In terms of inputs, the fertilizer application rate and the amount of water use in the North were lower. The GHG emissions from the rice fields in the North East were higher ranging between 1.97 – 3.17 kgCO₂e/kg; those in the North ranged between 1.15 – 1.70 kgCO₂e/kg. It was revealed that the potential impacts on all the impact categories considered per kg of paddy rice were generally lower in the North. The expansion of organic rice farming in the North for 100% performed the best with the lowest impacts in all categories. The quantitative environmental performance of different areas could be useful for supporting the policy decisions on the area expansion.

Keywords: Life Cycle Assessment, Greenhouse gas emission, Hom Mali rice, Organic Rice Farming, Thailand

1. Introduction

Jasmine rice is a key product significantly contributing to socio-economic benefits for Thailand. At the same time, rice cultivation activities are highlighted as a major source of greenhouse gas (GHG) emissions from the agricultural sector, which ranks after energy production and industry operation, respectively. Water consumption for rice cultivation, especially for “Hom Mali” (Jasmine) rice, is also raised as a concern as it requires a wetland system maintained at least one month during a production cycle mostly through irrigation or rainfall. The potential impact on biodiversity losses are additional issues of concern for environmental sustainability of rice production systems.

Organic farming is seen as an alternative system for more sustainable rice production due to lower risks from chemical use, increasing biodiversity, lower production costs, and higher price. At present, the proportion of organic rice is only 0.18% (19,994 ha) of the total area of rice production at the national level. However, it is targeted to be increased to 20% in 2021 as stated in the national strategic plan of organic agricultural production with the aim to become the regional hub for organic agricultural products (MOAC, 2016). At this stage, it is not clear which areas should be promoted for increased organic rice farming for achieving maximum environmental benefits.

2. Methods

Life cycle assessment (LCA) was conducted to evaluate the environmental performances of different potential paddy farming areas to provide supporting information for policy decisions. The life cycle inventory data of organic rice farming were collected by interviewing 184 farmers covering about 4% (887 ha) of the total production in the North and 208 farmers covering about 1% (290 ha) of the total production in the North East of Thailand. The direct greenhouse gas (GHG) emissions from rice fields were obtained from the literature based on the field measurement and supplemented by the default values as defined in the national Product Category Rules (PCR) of rice products when necessary (TGO, 2013).

Different scenarios based on varying cultivation area ratios of organic rice farming were proposed in the North and North East regions, and their potential environmental impacts were evaluated. The impact categories of interest included Climate change, Terrestrial acidification, Freshwater eutrophication, Terrestrial ecotoxicity, and Freshwater ecotoxicity. The impact assessment method used was ReCiPe (Goedkoop et al., 2008).

3. Results

Table 1 shows the average gate-to-gate inventory data of organic rice farming over the crop period of 120 days. It was found that the productivity in the North was higher than that in the North East. A higher fertilizer application rate was being used in the North East due to the lower fertility of soil. Most of the rice farms in the North were irrigated, while those in the North East were mainly rain-fed. The water use was higher in the Northeast, due to the climate conditions that are rather dry.

Table 1: The average gate-to-gate inventory data of organic rice farming (the crop period was 120 days)

Region	Inputs		Outputs	
	Item (unit)	Quantity	Item (unit)	Quantity
North	land (ha)	0.16	paddy rice (kg)	525
	seed (kg)	43	GHG emission	27.65
	water (m ³)	3,928	(kgCH ₄)	
	diesel (L)	42.8	GHG emission	0.0001
	manure (kg)	147.85	(kgN ₂ O)	
	organic fertilizer (pellet) (kg)	56.23	water emissions	
	compost (kg)	46.62	- nitrogen (kg)	0.0048
	organic fertilizer liquid (L)	9.56	- phosphorus (kg)	0.0001
		0.16		
North East	land (ha)	43	paddy rice (kg)	482
	seed (kg)	4,550	GHG emission	42.26
	water (m ³)	42.8	(kgCH ₄)	
	diesel (L)	96.08	GHG emission	0.0001
	manure (kg)	13.06	(kgN ₂ O)	
	organic fertilizer (pellet) (kg)	142.23	water emissions	
	compost (kg)	5.81	- nitrogen (kg)	0.0044
	organic fertilizer liquid (L)		- phosphorus (kg)	0.0001

4. Discussion

Table 2 indicates that the life cycle environmental impacts per kg of paddy rice cultivated in the North were generally lower in all impact categories than that of the North East. The life cycle impact assessment (LCIA) results indicated that Sukothai, Chiang Mai, Chiang Rai, Lampang and Phrae provinces in the North offered the lowest impact on climate change while the other impacts were not significantly different. In the same manner, Amnat Chareon, Yasothon, Mukdahan, Beng Kan, and Udon Thaini provinces in the North East performed the best with the lowest impact on climate change. Table 3 shows the results from the LCIA associated with different scenarios. It was revealed that the expansion of organic rice farming 100% in the North performed better, with the lowest impacts in all categories considered.

Table 2: Life cycle environmental impacts per kg of paddy rice cultivated in the North and North East

Impact category	Unit	North East	North
Climate change	kg CO ₂ eq	1.52E+00	1.09E+00
Terrestrial acidification	kg SO ₂ eq	2.71E-02	2.36E-02
Freshwater eutrophication	kg P eq	8.78E-06	7.50E-06

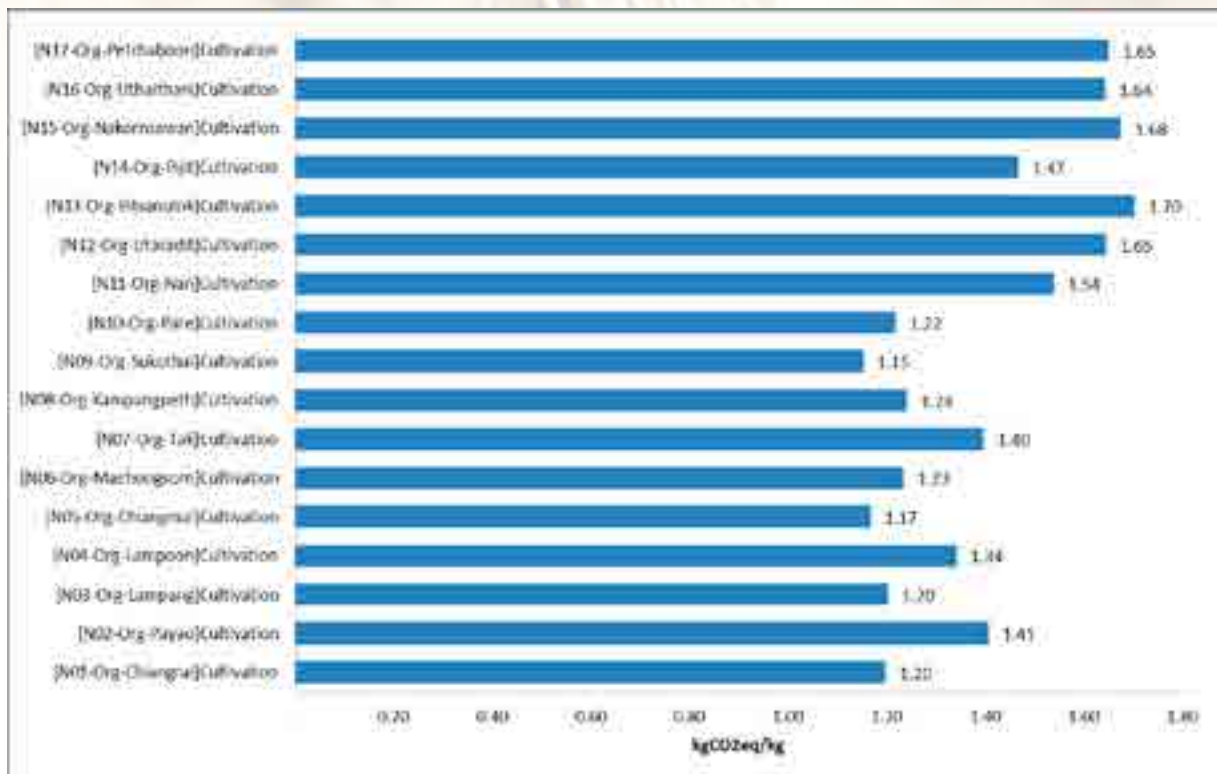
Terrestrial ecotoxicity	kg 1,4-DB eq	4.89E-06	4.16E-06
Freshwater ecotoxicity	kg 1,4-DB eq	1.12E-04	9.56E-05

Table 3: Results from the life cycle impact assessment

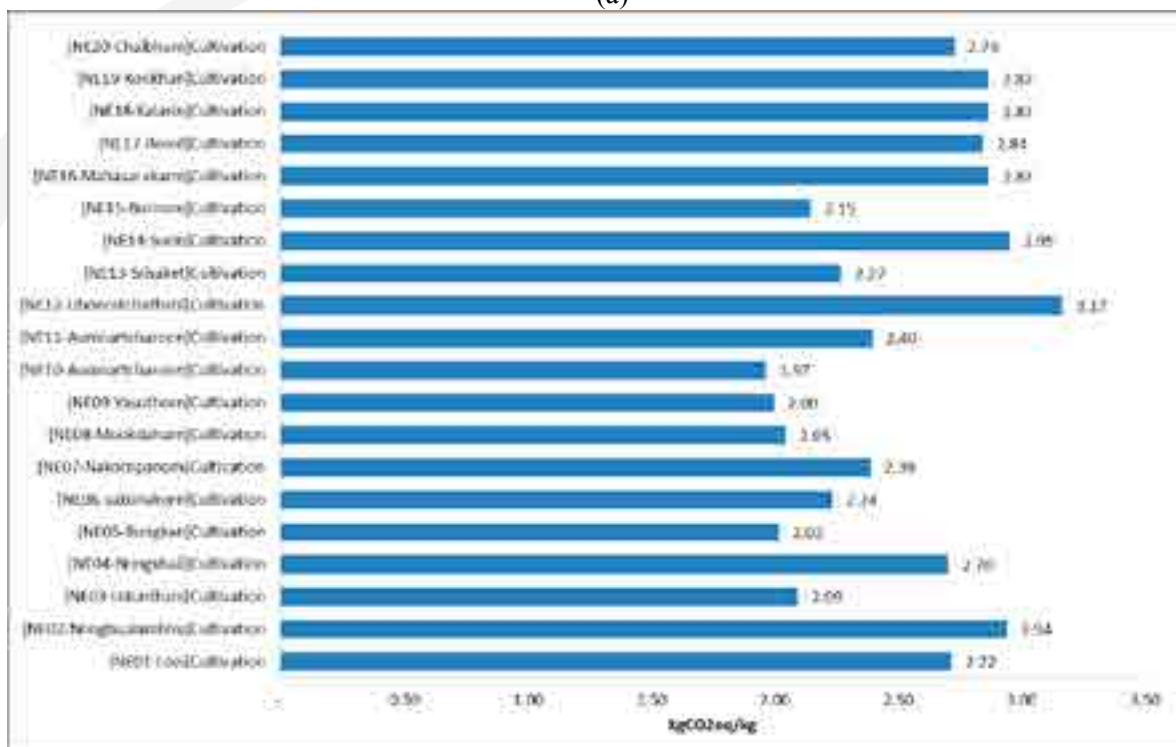
Scenario	Climate change (kg CO ₂ eq)	Terrestrial acidification (kg SO ₂ eq)	Freshwater eutrophication (kg P eq)	Terrestrial ecotoxicity kg (1,4-DB eq)	Freshwater ecotoxicity kg (1,4-DB eq)
N100%	46,116,390	998,345	317	176	4,046
N90%+NE10%	47,403,985	1,003,867	320	177	4,079
N80%+NE20%	48,691,581	1,009,389	322	179	4,111
N70%+NE30%	49,979,176	1,014,911	325	180	4,143
N60%+NE40%	51,266,771	1,020,434	327	182	4,176
N50%+NE50%	52,554,366	1,025,956	329	183	4,208
N40%+NE60%	53,841,961	1,031,478	332	185	4,241
N30%+NE70%	55,129,557	1,037,001	334	186	4,273
N20%+NE80%	56,417,152	1,042,523	337	187	4,305
N10%+NE90%	57,704,747	1,048,045	339	189	4,338
NE100%	58,992,342	1,053,567	341	190	4,370

Note: N – North, NE, North East (percentages indicate the areas planted in the North and North East)

The GHG emissions from rice fields in the North ranged between 1.15 – 1.70 kgCO₂e/kg whereas that in the North East between 1.97 – 3.17 kgCO₂e/kg (Figure 1). Sukhothai and Chiang Mai provinces performed the best in terms of climate change, due to lower fertilizer application rate and higher yield. In the North East, the lowest impact on climate change was found in Amnat Chareon, followed by Yasothon. It is worth noting here that the GHG emissions in Surin, Ubon Ratchathani, Khon Kaen, Phrae, and Chiang Mai provinces were based on the field measurements informed in technical reports (NSTDA, 2014). It was observed that the field measurements resulted in higher values of GHG emissions compared to the default values (using the IPCC tier-1 method).



(a)



(b)

Figure 1: GHG emissions per kg of rice produced in the North (a) and North East (b)

5. Conclusions

To make policy decisions on which areas should be expanded for the organic Hom Mali rice, the environmental performances of rice farming in different provinces should be taken into account. LCA offers quantitative environmental performances that could be useful for comparing the environmental performances among different areas. It was revealed that the expansion of organic rice farming 100% in the North was associated with the lowest impacts.

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89. Assessing methods to attribute soil greenhouse gas emissions to a crop in life cycle assessment of cropping systems

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ABSTRACT

There is an increased demand for a reduction in greenhouse gas (GHG) emissions and LCA has been widely applied to agricultural systems to estimate their Global Warming Potential (GWP). However, no consensus has been found on how to attribute soil GHG emissions in agricultural LCA. In this study, the objectives were: (i) to compare methods of attribution (year period, planting to planting, harvest to harvest), and (ii) to assess advantages and disadvantages of each method when used to attribute CO₂ and N₂O emissions to a crop in the LCA of cropping system with no winter crops. Soil CO₂ and N₂O emissions were estimated over 28 years using the biogeochemical DNDC model for 4 different scenarios based on a field experiment in Manitoba, Canada. Model results were used in the Crop.LCA tool to estimate global warming potential (GWP) on ha basis. Results showed no significant differences among methods when considering the full cropping system, however large differences were found on a year basis. Inter-annual variability was found to be higher than the difference across methods. Larger differences among methods were found for the cropping system where residues remained on the field. Thus a multimethod approach is suggested together with a long term LCA assessment to assess this system. The choice of methods to employ is a compromise between accuracy and applicability with regards to the LCA objectives.

Keywords: LCA, temporal variability, soil GHG emissions, cropping systems, DNDC

1. Introduction

There is an increasing demand for a reduction of greenhouse gas (GHG) to face the climate change challenge (Ingrao et al., 2016). Agriculture is responsible for 10-14% of the direct anthropogenic GHG emissions and an additional 12%–17% is due to land cover change, including deforestation (Paustian et al., 2016). Greenhouse gas emissions associated with agricultural activities can be highly variable because they depend on weather and climatic conditions, soil type, and agricultural practices (Miller et al., 2006).

Soil CO₂ emissions are caused by the decomposition of plant residues, mineralization of soil organic matter, decomposition of lime and urea hydrolysis. Different factors affect soil CO₂ emissions including soil temperature, soil moisture, type of residues and tillage (Brady and Weil, 2002; Paustian et al., 2016). Improved crop management often leads to an increase in soil C content which can largely contribute towards the reduction of atmospheric CO₂ concentration (Paustian et al., 2016). Management changes will tend to result in an initial rapid rate of soil C change slowly decreasing in time until a new equilibrium is reached, unless further management changes occur. After reaching this equilibrium, soil C content is affected mainly by climate variability, without any clear trend in net soil C change (Hutchinson et al., 2007; Paustian et al., 2016). This equilibrium can take a long time (e.g., 20-100 years) to reach depending on soil and climatic conditions (Goglio et al., 2015; Petersen et al., 2013; Smith et al., 2010).

In contrast to soil CO₂ emissions, soil N₂O emissions occur mostly through denitrification and nitrification, which have a shorter time frame (i.e., days, weeks) (Goglio et al., 2013; Lehuger et al., 2007). The applications of nitrogen fertiliser and animal manure, soil tillage and crop type can all affect N₂O emissions. However, the effects of these crop management practices are largely dependent on soil type and weather conditions (Saggar, 2010). Along with soil C, N₂O emissions have a large spatial and temporal variability (Goglio et al., 2013; Uzoma et al., 2015). Often the emissions remain low throughout the year with the exception of peaks where nearly 90% of N₂O emissions occur (Abdalla et al., 2009). These peaks generally occur after rainfall events, fertiliser and manure applications, during thawing and the decomposition of high N content residues (Goglio et al., 2013; Saggar, 2010; Wagner-Riddle et al., 2007). Several models have been used to assess soil N₂O and CO₂ emissions (Goglio et al., 2013; Kim et al., 2009; Zaher et al., 2013).

Life cycle assessment (LCA) is undertaken to account for all GHG emissions (e.g. CO₂ and N₂O) and the environmental impacts of agricultural systems. In cropping systems, it has been observed that

the impact of a crop is significantly affected by the previous crop (Brankatschk and Finkbeiner, 2015; Knudsen et al., 2014; Nemecek et al., 2015). Several studies have proposed different approaches to account for N₂O and CO₂ emissions (Goglio et al. 2015; Topp et al. 2011), including their temporal variability. However, no consensus has been found on how to attribute the timing of soil GHG emissions in agricultural LCA.

In this study, the objectives were: (i) to compare methods of attribution (calendar year, planting to planting, harvest to harvest), (ii) and to assess the advantages and disadvantages of each method when used to attribute CO₂ and N₂O emissions to a crop in the LCA of a cropping system with no winter crops.

2. Methods

The three methods of attribution considered were: the calendar year (Y), the period from planting to planting (PP), and the period from harvest to harvest (HH) (Figure 1). These methods were applied to 4 different cropping system scenarios. The Crop.LCA tool (Goglio et al., in preparation) was used to carry out a scenario assessment for different cropping systems on the basis of a field experiment described by Glenn et al. (2010, 2011, 2012) and Maas et al. (2013), located at the Glenlea Research Station (49.64°N, 97.16°W; height, 235 m, <2% slope) close to Winnipeg, Manitoba, Canada. The SOC content (0–0.2 m) was about 3.2% at the start of the study. The particle size distribution was 60% clay, 35% silt, and 5% sand.

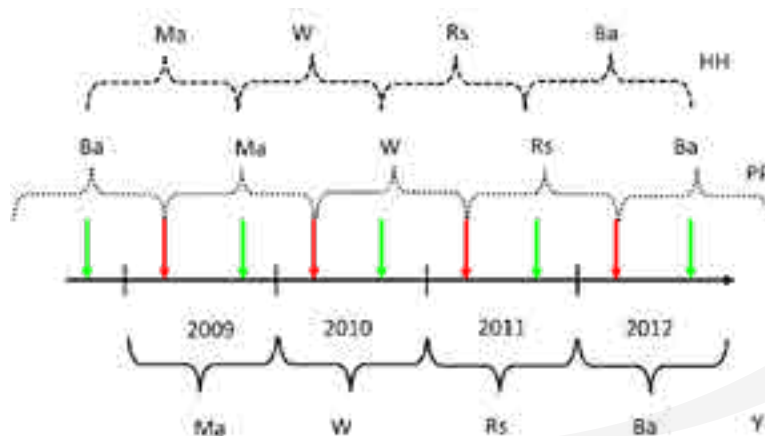


Figure 6 Description of the different methods to attribute soil GHG emissions to a crop (red arrows: planting date; green harrows: harvest date; Rs: rapeseed; Ba: barley; Ma: maize; W:wheat)

Crop management for the conventional (CONV) cropping system (Table 1) was similar to the annual cropping system described by Glenn et al., (2010) and Uzoma et al., (2015). The CONV, NT, and RES systems employed the same crop sequence, while in the legume (L) system, faba bean (*Vicia faba* var. minor L.) was substituted for maize (*Zea mays* L.) (Table 1). In the NT system, no tillage was carried out. Finally, in the RES cropping system, both straw and corn stover were left in the field (Table 1).

The scenario assessment used 28 years of climate data (1985–2012) to drive the estimates of the DNDC (Denitrification and Decomposition) model using site management and soils data from which the model was previously validated for estimating N₂O emissions (Uzoma et al., 2015). DNDC results for GHG emissions, grain and residue yield, and ammonia volatilization were then used as input to the Crop.LCA tool to perform the LCA (Goglio et al., in preparation) with 1 ha of land as the functional unit, in agreement with previous research (Goglio et al., 2014; Nemecek et al., 2011, 2015). One impact category was considered: 100 year horizon global warming potential (GWP), using the IPCC factors from the 5th assessment report (Myhre et al., 2013).

The system included the agricultural phase, all the upstream processes and farm transport (i.e. machinery production, transport, maintenance and repairs; production and delivery of fertilizers, pesticides, seeds and fuels and fuel consumption).

Data were collected from the field experiment, integrated with statistical data, expert opinion, databases, a survey of machinery manufacture, and products suppliers as in Goglio et al. (2014). For the HH method, it was assumed that both baling and collecting were carried out at harvest time.

GWP results on a ha basis were statistically assessed for the Y, PP and HH methods for each cropping system scenario separately, using R software (R Development Core Team, 2005). Each sample series was tested for normality. If the normality conditions were not met, the Friedman test was carried out together with paired nonparametric comparisons (Galili, 2013; Rosner, 2011). The standard deviation was computed for each cropping system x method combination; while the average difference between the various methods was computed for each cropping system.

Table 1 Summary of the average characteristics of the cropping systems considered in the scenario assessment (Note: Bold indicates differences between systems)

Cropping System	CONV	NT	L	RES
Crop sequence ^a	maize- spring wheat- rapeseed-spring barley	Maize- spring wheat- rapeseed-spring barley	faba bean^b- spring wheat- rapeseed- spring barley	maize- spring wheat- rapeseed- spring barley
Tillage	Yes	No tillage	Yes	Yes
Fertiliser				
N (kg ha ⁻¹ y ⁻¹)	112	112	72.2	112
P (kg ha ⁻¹ y ⁻¹)	16.6	16.6	6.6	16.6
K (kg ha ⁻¹ y ⁻¹)	18	18	0	18
S (kg ha ⁻¹ y ⁻¹)	18	18	0	18
Pesticide treatment number per year	1.5	2.5	1	1.5
Residue management	Straw and stover collected	Straw & stover collected	Straw and stover collected	Straw and stover left on the field

^amaize (*Zea mays* L.), spring wheat (*Triticum aestivum* L.), rapeseed (*Brassica napus* L.), spring barley (*Hordeum vulgare* L.), and faba bean (*Vicia Faba* var. *minor* L.); ^bfaba bean received no N fertiliser

3. Results

For each method, several advantages and limitations were assessed (Table 2). The applicability was assessed in agreement with Goglio et al. (2015) and (JRC, 2011). The Y methods has several advantages including the ease of use; however PP better represents soil C dynamics, while the HH method better considers crop management prior to planting (Brankatschk and Finkbeiner, 2015; Goglio et al., 2015; Nemecek et al., 2015). However, a higher level of applicability was attributed to the Y due to its ease of use since many biophysical models and emission factor accounting procedures provide output on an annual basis.

Figure 2 shows the overall results for GWP, soil CO₂ and soil N₂O emissions. It is clear that there is large variability between years and across the different cropping systems. This high inter-annual variability resulted in a statistical analysis that showed no significant differences (p<0.05) for the three methods across the entire time period. However, for GWP on a per year basis, the differences across the methods can reach up to 10,800 kg CO₂eq ha⁻¹y⁻¹ with higher values for the RES system; while the average differences among methods ranged between -430 and 41 kg CO₂eq ha⁻¹y⁻¹, representing at most 18% of the standard deviation (1130-4120 kg CO₂eq ha⁻¹y⁻¹) for all the methods and cropping systems. The estimated GWP per ha ranged from -3870 to 12100 kg of CO₂eq ha⁻¹ y⁻¹ (Figure 2a).

A similar pattern was found for soil CO₂ emissions, with no significant difference among the methods tested (Figure 2b). However, on average the difference among methods reached up to 450 kg CO₂ ha⁻¹y⁻¹, while the maximum difference reached 10,600 kg CO₂eq ha⁻¹y⁻¹ within the RES system. The average difference represented up to 27% of the standard deviation which ranged between 884-

Table 2 Summary of the main observations for the 3 different methods compared (Y: year method; PP planting to planting method; HH: harvest to harvest method)

Methods	Y	PP	HH
Advantages	Ease of use; In Canadian conditions no management activity occurs early or late in the year due to frozen ground	It includes residue incorporation due to tillage in the autumn or spring C inputs from residues to the appropriate crop.	Impacts for spring tillage operations including seedbed preparation are reflective of the following crop
Disadvantages	Highly affected by carry over from previous crops; autumn tillage carried out for the following spring crop are attributed to the previous crop	The impacts of the tillage operations carried out prior to seeding are attributed to the previous crop	It attributes residue incorporation due fall tillage and also decomposition to the following crop
Applicability	High	Medium	Medium

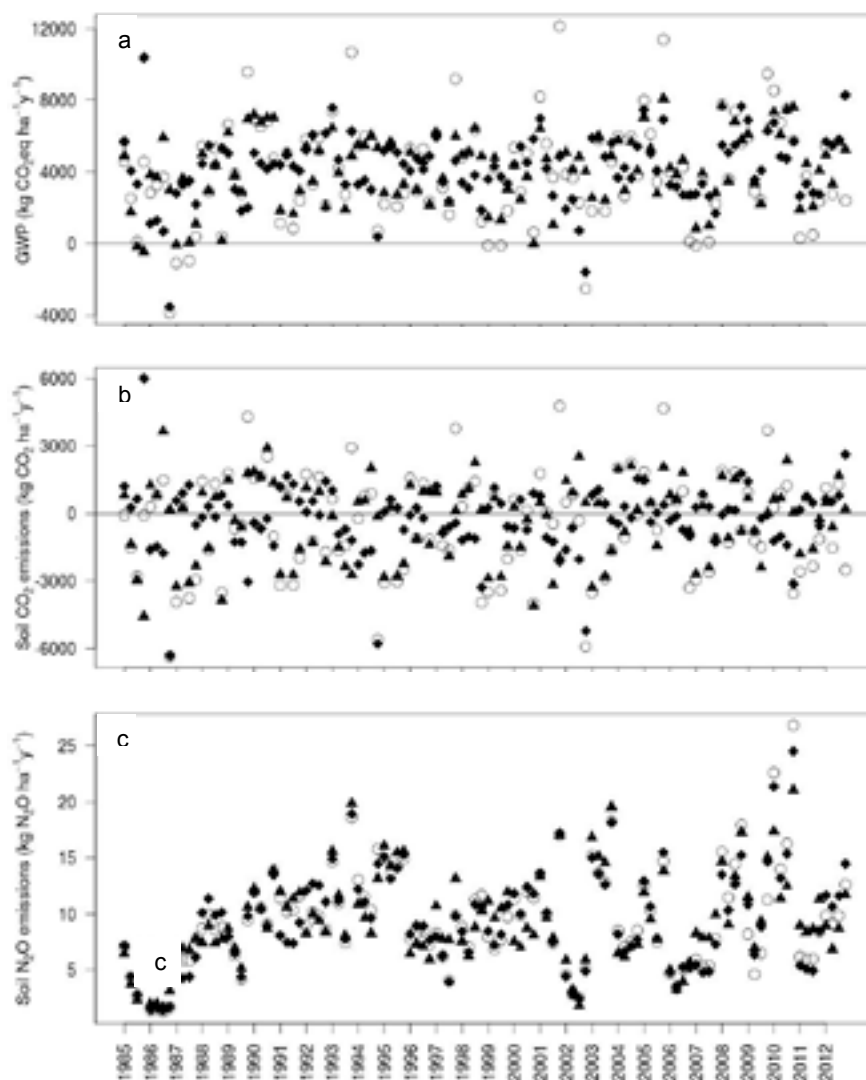


Figure 7 GWP (a), soil CO₂ emissions (b), soil N₂O emissions (c) for the four cropping systems on year basis with the 3 different methods tested (○ Year period (Y), ▲ planting to planting (PP), ◆ Harvest to harvest (HH))

3200 kg CO₂ ha⁻¹y⁻¹. Soil CO₂ emissions on a per ha basis were estimated between -6350 and 6000 kg CO₂ ha⁻¹ y⁻¹.

There was also no significant difference between the three methods for soil N₂O emissions (Figure 2c). However, the values on a yearly basis were clearly more similar than results for CO₂ and in contrast to GWP and soil CO₂ emissions, N₂O emissions for certain crop and cropping system combinations were found to be similar between the methods tested (Figure 2c). The inter-annual variability in N₂O emissions was very high for all methods, which was not surprising considering the site specific variability was also high (Uzoma et al., 2015). The standard deviation for each method (by plot) ranged between 3.1 and 5.2 kg N₂O ha⁻¹y⁻¹, at least 48 times larger than the average difference among methods (-0.082 to 0.084 kg of N₂O ha⁻¹y⁻¹). The simulated soil N₂O emissions ranged from 1.3 to 27 kg of N₂O ha⁻¹ y⁻¹.

4. Discussion

The main objectives were to compare different methods to attribute the timing of GHG emissions in agricultural LCA. The main advantages of the yearly method is the ease of use, since some methods for estimating GHG emissions only operate on an annual basis (such as Tier I and Tier II IPCC methodology, (Paustian et al., 2006)). Instead, the PP method had the advantage of including residues dynamics of the planted crop, and the HH method considered the effect of the associated crop management prior to planting. In contrast, the latter two methods had lower applicability, while the yearly method was affected by carry over effect from previous crops and attributes autumn tillage effects to the previous crop. A previous experiment was carried out by Topp et al. (2011) with soil N₂O emissions while here soil N₂O emissions together with soil CO₂ emissions were assessed. Furthermore, Topp et al. (2011) assessed both spring and winter crops, while here just spring crops were evaluated.

The results obtained show that over the long term all the methods are equivalent. The differences between methods was overshadowed by high inter-annual variability, however on a year to year basis large differences between methods were obtained for both GWP (up to 10,800 kg of CO₂eq ha⁻¹y⁻¹), soil CO₂ emissions (up to 10,600 kg of CO₂ ha⁻¹y⁻¹) and N₂O emissions (up to 5.8 kg of N₂O ha⁻¹y⁻¹). This suggests that an important accounting error may occur when only one year is assessed. Therefore, long term assessment should be performed to avoid uncertainties related to GHG attribution to different crops. This was in agreement with previous findings which highlighted the importance of carrying out both LCA of crops and cropping systems at the same time to better consider the carry over effects over long periods (Goglio et al., 2014; Nemecek et al., 2015).

The present results suggest that the residue retained system is more strongly influenced by the various methods on a year to year basis, thus an appropriate multimethod comparison should be carried out. Indeed soil C dynamics are slow and a long period of time is necessary to detect potential changes in crop management affecting the overall soil C content (Goglio et al., 2015; Paustian et al., 2016). This is generally better captured by the PP method since in the RES systems crop residues are tilled before planting time the next year. Alternative approaches suggested by Knudsen et al. (2014) consider residue decay, but they are more complex than the methods presented here. Thus, the choice of methods results in a compromise between the applicability and accuracy of the methods with regards to the objectives of the LCA, as previously discussed (Garrigues et al., 2012; Goglio et al., 2015). The present recommendations could be considered valid and reliable for spring crops in temperate/continental conditions. Due to the complexity of cropping system interactions and the dependency of GHG emissions on regional conditions (Brankatschk and Finkbeiner, 2015; Goglio et al., 2014; MacWilliam et al., 2014, 2014; Nemecek et al., 2015), the present recommendations cannot be extended for other cropping systems in different climates.

GWP values obtained here were larger in range (-3870 to 12,100 kg CO₂eq ha⁻¹y⁻¹) in comparison to previous research (-6277 to 5190 kg CO₂eq ha⁻¹y⁻¹) (Camargo et al., 2013; Dendooven et al., 2012; Dyer et al., 2010; Goglio et al., 2014; Kim et al., 2009; Pelletier et al., 2008) who assessed several crops including wheat, rapeseed and maize in North American conditions using either agroecosystem models or direct measurements for GHG emissions. Present CO₂ emissions had a lower range (-6350 to 6000 kg of CO₂ ha⁻¹ y⁻¹) than several studies for North American conditions (-17,100 to 2270 kg of CO₂ ha⁻¹y⁻¹) (Dendooven et al., 2012; Ellert and Janzen, 2008; Hernandez-Ramirez et al., 2011).

However, the soil N₂O emissions from the study site for which DNDC was validated (Uzoma et al., 2015) were highly variable but the modelled results were considered to be reasonably accurate in comparison with observations (Uzoma et al., 2015). Further perspectives would include the complete assessment of methods to account for soil GHG emissions together with smaller secondary N₂O emissions (i.e., NH₃ volatilisation and nitrate leaching).

5. Conclusions

The present research compared different methods to attribute the timing of soil GHG emissions in agricultural LCA of spring crops in a continental climate. The long term differences between the methods were negligible for spring crops in continental Canadian conditions. In contrast, on an annual basis, the methods showed large differences with regards to GWP and soil CO₂, particularly in systems with no residues harvested. In this situation, a multimethod comparison is recommended to avoid erroneous conclusions. In the choice of the methods, a compromise should be found between accuracy and applicability in relation to the objectives of the study. Present findings can be applied in similar climatic areas and for similar crops. Further perspectives include carrying out this LCA comparison with winter crops and in other climates, including secondary N₂O emissions.

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206. Life-Cycle Assessment of Table Grape Production in Eastern Austria

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ABSTRACT

In Austria, a domestic table-grape producing sector is emerging, with production being pioneered by small, family-owned vineyards. While life-cycle assessment (LCA) results are readily available for wine grapes, table grape LCAs have not been found. The objective of this work was the quantification of selected environmental impacts of table grape production in three small case study vineyards in Eastern Austria, enhanced by a comparison with a hypothetical reference vineyard and by an identification of optimization measures. The method is a cradle-to-gate, attributive life-cycle assessment (LCA) with a functional unit of one kilogram of table grapes at the first point of sale. Results demonstrate that impacts can vary substantially between vineyards. Climate change impacts (GWP100) range from 0.30 to 1.05 kg CO₂-eq/kg grape, mainly due to machinery operations, the production of packaging materials, and fertiliser application. Freshwater eutrophication impacts range from 0.09 to 0.18 g P-eq/kg grape, terrestrial ecotoxicity impacts range from 0.09 to 1.76 g 1,4-DCB-eq/kg, and human toxicity impacts range from 0.13 to 0.28 kg 1,4-DCB-eq/kg. Large uncertainties that preclude any differentiation between vineyards were found for eutrophication and human toxicity impacts. A much higher assumed grape yield in the hypothetical reference vineyard R results in substantially lower impacts, but with the yield assumption adjusted to match those in the example vineyards, the reference impacts also become comparable. Options to reduce impacts in the example vineyards include the use of modern, efficient tractors with fewer cultivation steps and a less material-intensive packaging system. To the authors' knowledge, the study for the first time presents LCA results on table grape production in small Austrian vineyards, and its findings may help to lower future impacts by pointing out effective improvement measures.

Keywords: Table grapes, attributive LCA, hot spots, sensitivity analysis.

1. Introduction

Table grapes consumption in Austria is dominated by imports (Statistik Austria 2016), but a domestic table-grape producing sector is emerging, with production being pioneered by small, family-owned vineyards. Potential advantages of local table grape production include a streamlined supply chain with more disease-resistant varieties, less storage and shorter transport distances, allowing for ripening at the vine and thus enhanced flavors (Ollig 2010). Table grape varieties are distinct from wine grapes; while the environmental impacts of wine production have been extensively studied (e.g. Bosco et al. 2011, Christ and Burritt 2013, Rugani et al. 2013), similar work on table grapes is lacking. The objective of this work was a cradle-to-gate, attributive life-cycle assessment (LCA) of table grape production in three small vineyards in Eastern Austria. Specific sub-objectives were (a) to quantify four potential environmental impacts (global warming potential with a 100-year horizon, GWP100; freshwater eutrophication potential, FEP; terrestrial ecotoxicity potential, TETP; and human toxicity potential, HTP), to compare these impacts to those of a hypothetical reference vineyard, and to identify options to reduce the potential environmental impacts in this emerging agricultural sector.

2. Methods

The three case study vineyards are situated in established wine production regions in the two eastern provinces of Austria, Lower Austria and Burgenland. Their characteristics are summarized in Table 1. In all three vineyards, the vines are vertically trained to a bilateral cordon, but other characteristics differ. Vineyard A is partially situated on a steep slope with sandy soil that is low in potassium and phosphorus. On a total area of one hectare, 2000 vines are planted. The area comprises three smaller vineyards, all at a distance of approximately 1.5 km from the farm. The expected yield of approximately 7.0 tons of grapes per hectare and year is an estimate since one of the three smaller vineyards had just been planted at the time of data collection. Fertilisation is currently limited to green manure, a perennial grass-clover mix (approximately one quarter grass). Additionally, all woody and leafy residues are mulched into the soil three times a year. The vineyard currently limits pesticide use to a 80% wettable sulfur fungicide dispersion. Tillage, mulching, and other cultivation requires 53 hours of tractor operation annually, and harvest requires 40 h/a with an old (1965) tractor model. Grapes are pruned and harvested manually. In this and the other vineyards studied, all grapes are

assumed to be stored after harvest in chilled storage halls, and no grapes of lesser quality are used as co-products for juice production. Forty percent of vineyard A's production is sold at local farmers' markets, the remainder is sold directly on the farm to mostly local customers living within 10 km of the farm. The packaging consists of cardboard trays of 0.5 or 1.0 kg capacity.

Table 1: Key characteristics of the three studied vineyards A-C and the hypothetical reference vineyard R. Data for vineyards A-C are based on questionnaires completed by vineyard owners; data for vineyard R are based on Richter (2010), unless noted otherwise.

Characteristic	Vineyard A	Vineyard B	Vineyard C	Vineyard R
Planted area, ha	1 ^a	8	0.66	1 ^b
Yield, kg/ha/a	7000 ^a	6500 ^a	7500 ^a	14000 (Ziegler 2011)
Management system	"Natural" (owner's specification)	Organic	Organic	Conventional
Fertiliser	Green manure, mulched 3x per year, no tillage	Green manure, mulched 4x per year, with tillage	Green manure, mulched 3x per year, with tillage	Mineral fertiliser ^c , green manure mulched 1x per year
Pesticides (kg/ha/a)	Sulfur (1.6)	Sulfur (11.7), copper (0.17), others ^{d,e}	Sulfur (15), copper (2), others ^e	Sulfur (12), phosphonic acid (3), copper (1) (Redl 2013)
Tractor operation (year produced, rated power in kW, hours/ha/a)	Tractor 1 (2010, 66, 40); Tractor 2 (1965, 48, 53)	Tractor 1 (2006, 63, 9.8); Tractor 2 (2006, 52, 8.2)	Tractor 1 (1987, 52, 45.5); Tractor 2 (2004, 55, 30.3)	Tractor 1 (2002, 45, 27) ^{b,f}
Packaging (kg grapes per container)	Cardboard trays (0.5 kg or 1.0 kg)	PET trays (0.5 kg) in corrugated cardboard one-way boxes (5kg)	PET trays (0.5 kg) in reusable PP crates (5-7kg)	LDPE bags (0.5 kg) in reusable PP crates (5-7kg) ^b
Transport to first sales point	60% sold through direct marketing (10 km by diesel car); 40% sold at farmer's markets (40%, 13 km by diesel car)	To supermarkets (150 km by a 3.5-7.5 ton lorry)	To supermarket (90 km by van < 3.5ton)	To supermarket (30km by van < 3.5 ton) ^e

^a approximate value; ^b authors' assumptions; ^c mass per ha and year: 40 kg N, 20 kg P₂O₅, 70 kg K₂O, 25 kg MgO (Ziegler 2011); ^d includes ethanol, potassium bicarbonate, tensides, pheromones (not modelled); ^e algae-based phyto-stimulants (not modelled); ^f model tractor from the ecoinvent v2.2 (ecoinvent Centre, 2010) database

Vineyard B is an organic vineyard situated at an approximate distance of 0.5 km from the farm on gently sloped sandy loess. The total area of 8 ha is split into 2 four-hectare sections, with 2800 vines per hectare yielding 6.5 t/ha/a. Green manure cover (80% legumes) is maintained except during the main vine growth period between May and July, with mulching four times a year. The vineyard applies pesticides and preventative plant protection chemicals, including wetting sulfur, copper, potassium bicarbonate, a wetting agent, and a foliar fertiliser. An algae-based plant strengthener and a pheromone agent could not be included in the LCA model due to a lack of inventory data. Two 2006 tractors with 63 kW and 52 kW are operated for a total of 18 hours per hectare and year. Grapes are manually harvested with 5-kg multi-use PP crates and packaged in 0.5 kg PET trays that are contained in 5-kg corrugated cardboard crates as requested by the customer, a distributor located at a distance of 150 km from the vineyard.

Vineyard C is an organic 0.66-ha vineyard situated at an approximate distance of 4 km from the farm on gently sloped sandy loam. 2244 vines yield 4950 kg grapes per year, or 7.5 tons per hectare and year. Green manure is re-seeded every other year and mulched three times a year, and woody and leafy residues are incorporated into the soil as well. Plant protection consists of wetting sulfur, copper, and an algae-based plant strengthener that could not be modelled due to a lack of data. Mulching and other cultivation, as well as part of the harvest are conducted with an older model tractor (1987) for a

total of 45 hours, and the other part of the harvest, as well as tillage operations are conducted during 30 h/a with a newer (2004) tractor model. Grapes are manually harvested directly into PP crates that contain 0.5-kg PET trays and shipped in a small van to the sole customer, a supermarket at a distance of 90 km from the vineyard.

A hypothetical, one-hectare conventional reference vineyard (vineyard R) was assembled from literature data, mainly from a German model cost calculation by Richter (2010). The vineyard is assumed to be located at a 5-km distance from the farm; it grows 4000 vines with a yield of 14 t/ha/a (Ziegler 2011). Permanent green manure is mulched once per year, as are all woody and leafy residues. This is in addition to mineral fertilisers (ammonium nitrate, diammonium phosphate, magnesium oxide, potassium chloride, and Boron). It was further assumed that the vineyard applies fungicides in the form of sulfur, copper, and phosphonic acid in the course of six applications a year. Tractor operation was modelled at 27 operating hours per hectare and year, with ecoinvent data that describe a 2002 tractor. Harvest is done manually into reusable PP crates that are also used for storage and transport. Grapes are assumed to be sold to a supermarket at a distance of 30 km, packaged in 0.5-kg LDPE-bags. This hypothetical vineyard differs from the three case study vineyards mainly in the much higher grape yield and in the use of mineral fertiliser. The influence of the higher yield assumption on the results is demonstrated below as part of a sensitivity analysis.

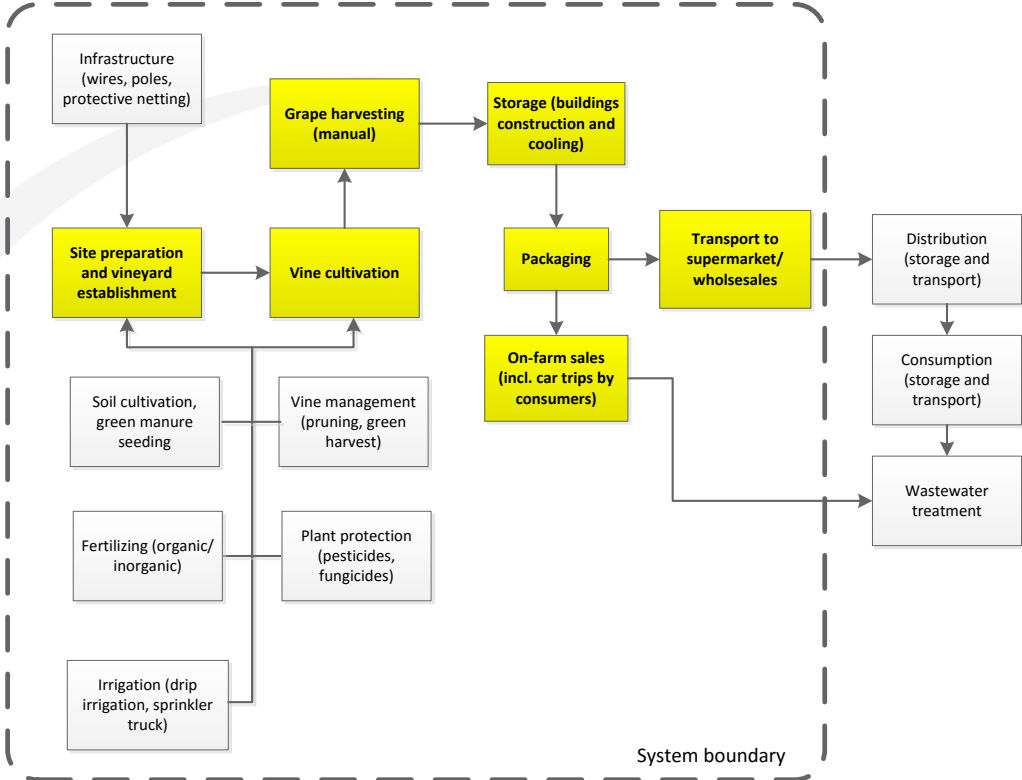


Figure 1: Simplified system diagram of table grape production in Eastern Austria

The life-cycle inventory model for the table grapes shows technical system boundaries (Figure 1) that model the cradle-to-gate process chain of table grape production. The models include the full production chains, from establishing the vineyard, to grape storage (2 days in chilled rooms) and packaging, to the transport to the first point of sale. An attributive LCA approach was deemed suitable since the focus was on assessing emerging local table grape production rather than on comparing such production with that of the currently prevalent imported table grapes. The functional unit was one kilogram of table grapes at the first point of sale, either to private customers or to a distributor/supermarket. The LCA model was implemented with the openLCA software (version 1.4.2; Green Delta GmbH, Berlin, Germany) combined with the ReCiPe(H) impact assessment method (Goedkoop et al. 2013). Four of ReCiPe(H)'s impact categories were chosen as representing

anticipated environmental impacts: the 100-year global warming potential (GWP100), the freshwater eutrophication potential with infinite time horizon (FEP), the freshwater ecotoxicity potential with infinite time horizon (FETP), and the human toxicity potential with infinite time horizon (HTP). Primary data were collected from interviews with vineyard owners, secondary data were obtained from literature and from the ecoinvent v2.2 (ecoinvent Centre, 2010) database. Parameter uncertainties (e.g. vineyard poles, wires, and nets; hours of tillage; N₂O emissions from fertilisation; and transportation distances), were propagated to total impact uncertainties using Monte-Carlo simulations with 10,000 runs for the vineyard total impacts, and 1,000 runs for the contribution analyses.

3. Results

A comparison of overall results between the four vineyards (Figure 2) shows that the climate change impact is higher for all three case study vineyards than that of the hypothetical reference vineyard R, with a GWP100 of 0.49 to 1.05 kg CO₂-eq/kg grape for vineyards A to C compared to 0.30 kg CO₂-eq/kg for vineyard R. This is largely due to the above-mentioned higher yield assumed for vineyard R that distributes impacts over a larger amount of grapes. Other reasons for higher GWP100 of the case study vineyards include the operation of older, less efficient tractors for more hours per year in vineyards A and C, which contributes 69% and 40.0% to the total GWP100, respectively, and relatively material-intensive packaging (PET polymer trays) by vineyards B and C (see Table 1).

Vineyard R impacts are lowest also with respect to the eutrophication potential FEP and the human toxicity potential HTP, with the exception of vineyard A's eutrophication potential (Figure 2), again due to the high yield assumption for the reference vineyard. However, large uncertainties blur the differences between vineyards in both impact categories.

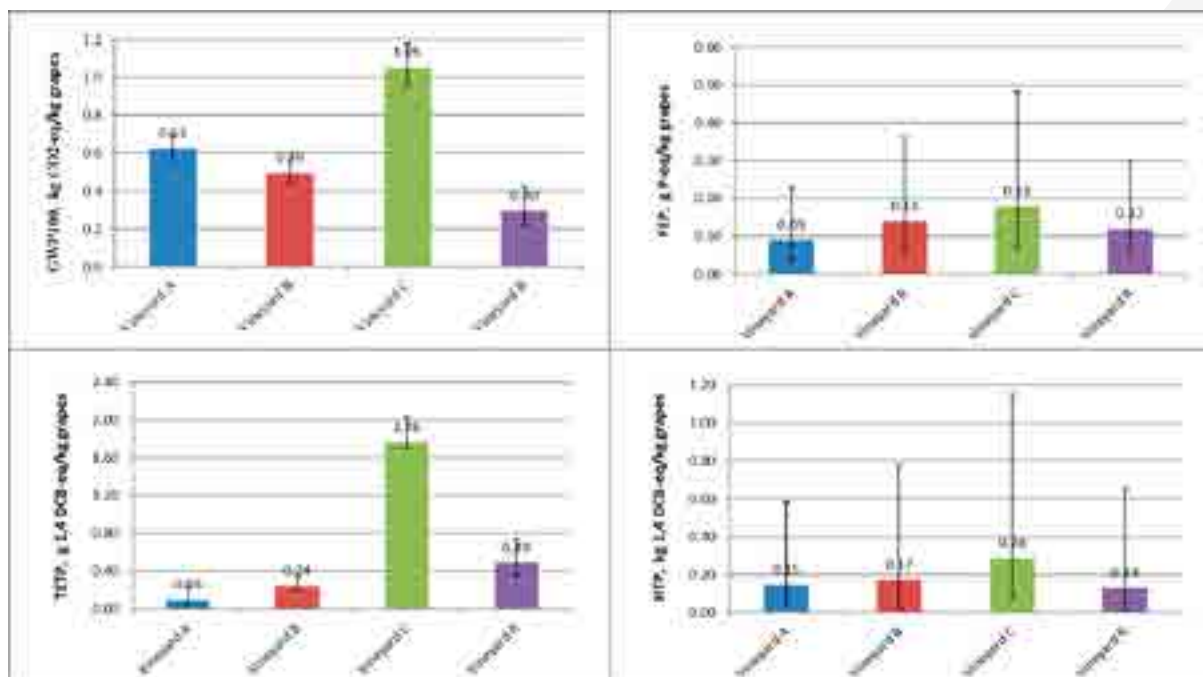


Figure 2: Potential impacts per kg table grapes for four selected impact categories of the three studied vineyards A to C and the hypothetical reference vineyard R. GWP = global warming potential, FEP = freshwater eutrophication potential, TETP = terrestrial ecotoxicity potential, HTP = human toxicity potential (error bars = 5th and 95th percentiles, Monte-Carlo simulations, n=10,000)

The terrestrial ecotoxicity impact of table grape production varies widely between vineyards, with vineyard C showing by far the highest result, almost twenty times that of vineyard A. This is explained by the main contribution to the TETP, the application of copper as a fungicide; Vineyard C applies the

highest copper amounts, while vineyard A reported that it does not apply any copper (s. Table 1). However, the use of copper is considered essential by Austrian experts; an assumed copper use by vineyard A of 1 kg/ha/a would increase its total TETP more than tenfold, from 0.09 to 0.98 g 1,4-DCB-eq/kg grapes.

The contribution analysis shows that dominant processes are vineyard- and impact-specific. For the climate change impact, vineyards A (Figure 3) and C (data not shown) are dominated by tractor operations (69% and 46% of the total impact, respectively) due to their high machinery operation hours and use of old tractors for some processes (see Table 1). For vineyards B and R (data not shown), potential contributions to climate change are more evenly spread, but diverse. In vineyard B, the largest contribution to climate change is the manufacturing of PET packaging trays (27% of total GWP100), while in vineyard R the highest contribution is due to the application of mineral fertiliser (21% of total GWP100), both from nitrous oxide emissions after field application and from ammonium nitrate production emissions.

Freshwater eutrophication is dominated by infrastructure manufacturing processes for all vineyards (machinery in vineyard A, materials such as steel support poles and wires in vineyards C and R), as well as by phosphate emissions from plant protection and from PET manufacturing for packaging vineyard B grapes. The terrestrial ecotoxicity potential is dominated by emissions of copper fungicide in all vineyards, with the exception of vineyard A, which does not use copper. There, 32% of a relatively low total TETP is due to hydrocarbons emitted during operation of the very old (1965) tractor, followed by machinery and diesel production emissions. Emissions from this tractor are also the largest contribution (28% of total impacts) to the human toxicity potential impacts of vineyard A. Vineyard materials manufacturing, the second-largest contribution to HTP, dominates the impacts in the other vineyards, with 40%, 32%, and 56%, of total impacts for vineyards B, C, and R, respectively.

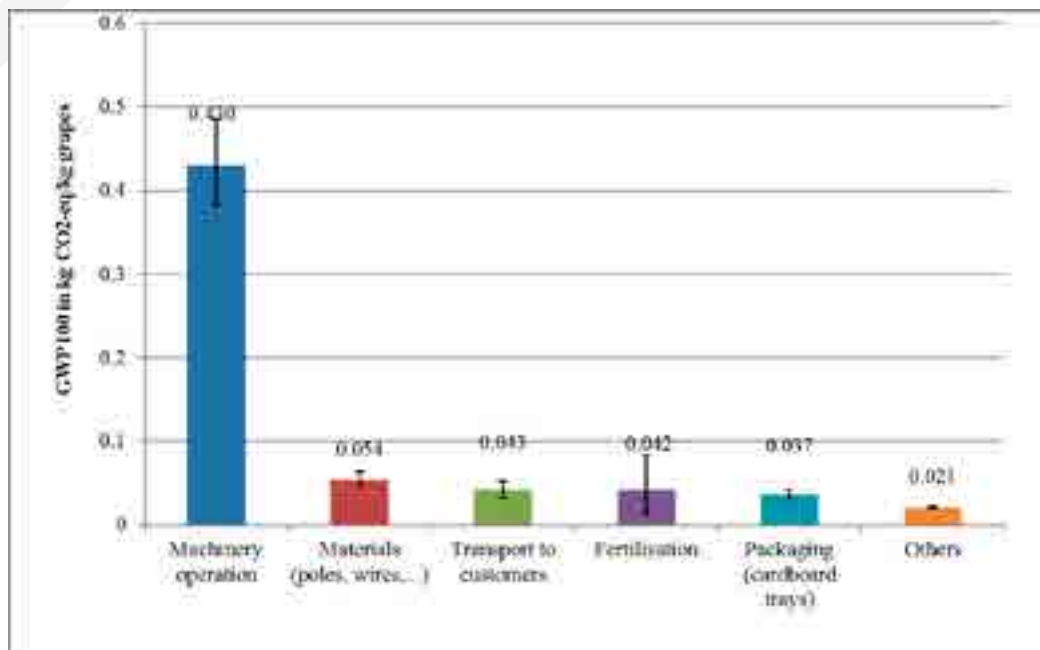


Figure 3: Contributions to global warming potential of vineyard A grapes (error bars = 5th and 95th percentiles, Monte-Carlo simulations, n=1000)

4. Discussion

The comparison between the three case study vineyards and with the hypothetical reference vineyard R demonstrates that impacts can vary substantially between vineyards. Not surprisingly, the grape yield largely determines relative environmental impacts; the yield may not only be limited by

local soil and climate conditions, but also by quality considerations. Comparison with carbon footprints in literature on wine grape production shows that the table grapes' footprints of 0.30 - 1.05 kg CO₂-eq/kg are comparable to the wide range of carbon footprints in published wine grape studies. For example, Bosco et al. (2011) give a values for four different wines from the Tuscany region in Italy that can be re-calculated to 0.11-0.32 kg CO₂-eq/kg (wine) grape, assuming 1.1 kg grapes used to produce 0.75 liters of wine. At the other end of the range, Neto et al. (2012) report a carbon footprint of 2 kg CO₂-eq for 0.75 liters of Portuguese *vinho verde* wine, which would correspond to 1.82 kg CO₂-eq/kg (wine) grape. As the dominating contributors, the two studies identified greenhouse gas emissions from the production and application of mineral fertiliser, as well as from machinery operations. A similar literature analysis for the impact category freshwater eutrophication potential (data not shown) indicates that the results calculated for the case study vineyards are comparable, but in parts lower, than literature results for wine grapes.

As part of a sensitivity analysis, the effects of three hypothetical changes to the vineyard models were calculated: First, the old tractors used for some operations in vineyards A and C (1965 and 1987, respectively) were replaced with the 2002 model from the ecoinvent 2.2 database. This resulted in a 28% reduction of GWP100, from 0.63 to 0.45 kg CO₂-eq/kg grape for vineyard A, and a 38% reduction for vineyard C, from 1.05 to 0.66 kg CO₂-eq/kg grape. Somewhat smaller reductions would affect mainly the human toxicity impacts in these vineyards. Second, the effect of altered packaging material for the grapes was investigated in vineyard B, by exchanging the climate-change dominating PET trays with lighter LDPE bags. This results in large reductions, particularly in the eutrophication, climate change, and human toxicity impacts (Table 2). It should be noted, however, that the choice of packaging material is currently made by the vineyard's customer and not by the vineyard owner.

Table 2: Sensitivity result – Vineyard B, effect of packaging dematerialization

Impact category	Packaging materials		Effect on impact category (% change)
	Baseline packaging ^a	Reduced packaging ^b	
GWP100 [kg CO ₂ -eq/kg]	0.50	0.34	- 0.15 (-30.5%)
FEP [kg P-eq/kg]	1.40*10 ⁻⁴	7.75*10 ⁻⁵	- 6.25*10 ⁻⁵ (-44.6%)
TETP [kg1,4-DCB-eq/kg]	2.40*10 ⁻⁴	2.10*10 ⁻⁴	- 0.30*10 ⁻⁴ (-12.5%)
HTP [kg1,4-DCB-eq/kg]	0.170	0.12	- 0.06 (-31.8%)

^a PET trays (0.5 kg) in one-way corrugated cardboard boxes (5 kg)

^b LDPE bags (0.5 kg) in reusable PP crates (5-7 kg)

Third, the effect of reducing the yield of the hypothetical reference vineyard R was estimated. A reduced yield of 7 t/ha/a was assumed (down from an original 14 t/ha/a; Ziegler 2011), which would be comparable to the case study vineyards' yields of 6.5 to 7.5 t/ha/a. As a coarse approximation, the fertiliser application was halved, but pesticide amounts were left unchanged. As a result, total impacts in all impact categories of vineyard R increased substantially, from an almost 50% increase in the climate change category, to almost doubling the terrestrial ecotoxicity potential impact (Table 3).

The LCA results presented here may be interpreted as estimates of selected potential environmental impacts; however, other environmental aspects of table grape production are not covered. For example, no data were available on the effect of table grape production on soil carbon dynamics. For wine production, Bosco et al. (2013) have shown that accounting for soil organic matter can change the vineyard portion of a wine's carbon footprint from a GHG source to a net sink, depending on the

Table 3: Sensitivity result – Vineyard R, effect of reducing yield from 14 t/ha/a to 7 t/ha/a, comparable to those of case study vineyards (6.5 to 7.5 t/ha/a).

Impact category	Annual grape yield	Effect on impact
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	Baseline (14 t/ha/a)	Reduced (7 t/ha/a)	category (% change)	
GWP100 [kg CO ₂ -eq/kg]	0.30	0.44	0.145	(+ 48.8 %)
FEP [kg P-eq/kg]	1.20*10 ⁻⁴	1.89*10 ⁻⁴	0.69*10 ⁻⁴	(+ 57.8 %)
TETP [kg 1,4-DCB-eq/kg]	4.90*10 ⁻⁴	9.60*10 ⁻⁴	4.70*10 ⁻⁴	(+ 95.9 %)
HTP [kg 1,4-DCB-eq/kg]	0.13	0.24	0.11	(+ 82.3 %)

site and on management practices. Other environmental impacts that were not included here but may be of interest, such as solid waste production and land use, have been summarized for wine production by Christ and Burritt (2013). Further limitations include plant protection agents that could not be modelled due to a lack of LCA data (pheromones and phytostimulants), as well as the exclusion of the effects of temporal variability (for example, large increases in copper use in precipitation-rich years, and yield fluctuations in general). While parameter uncertainties were propagated to the end results using Monte Carlo simulations, most model uncertainties (for example, the actual copper emissions from copper fungicide application) could not be included in the calculation due to a lack of reliable data.

5. Conclusions

Selected potential environmental impacts could be quantified with an attributional cradle-to-gate LCA for three Austrian table grape production systems and for one hypothetical reference system, from vineyard establishment to the first point of sale. The comparison between the three case study vineyards and with the hypothetical reference vineyard R demonstrates that variability between vineyards can be substantial. Climate change impacts expressed as GWP100 range from 0.30 to 1.05 kg CO₂-eq/kg grape, mainly due to machinery operations, the production of packaging materials, and fertiliser application. Freshwater eutrophication impacts range from 0.09 to 0.18 g P-eq/kg grape, with major contributions from manufacturing machinery and vineyard infrastructure, from preventive fungicides and fertilisers, and from PET tray packaging production. Terrestrial ecotoxicity impacts range from 0.09 to 1.76 g 1,4-DCB-eq/kg, caused to a large extent by copper fungicide application emissions. Human toxicity impacts range from 0.13 to 0.28 kg 1,4-DCB-eq/kg, with contributing processes similar to freshwater eutrophication. Large uncertainties that preclude any differentiation between vineyards were also found for the two impact categories of eutrophication and human toxicity.

A high yield assumption by the literature source for the hypothetical reference vineyard R leads to lower impacts relative to the three case study vineyards across all but one of the four categories considered. However, a reduced yield assumption that is in line with the three case studies, increases impacts to levels that surpass two of the three vineyards in all impact categories but climate change. With climate change, relatively low machinery use and light grape packaging leave the carbon footprint for the reference system below those of the case study systems even at comparable grape yields.

A comparison with literature on wine grape production shows that the table grapes' impacts are comparable or somewhat lower than those of the analysed wine grape studies.

Options to reduce the impacts of table grape production in the case study vineyards could be identified based on the contribution analysis. They include the use of modern, efficient tractors with fewer cultivation steps, and a less material-intensive packaging system. In the vineyard with the highest total GWP100, just replacing the old tractor with a modern model would reduce the grapes' total carbon footprint by 38%. In vineyard B, replacing the PET tray packaging with LDPE bags would substantially reduce impacts, but the choice of packaging material is made by the vineyard's customer and thus may not be easily changed.

To the authors' knowledge, the present study for the first time presents LCA results on the environmental impacts of table grape production in small Austrian vineyards, and its findings may help to lower future impacts by pointing out effective improvement measures.

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236. Including product multicriteria quality in life cycle assessment, application to grape

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ABSTRACT

Objective: In the context of protected designation of origin (PDO) wine production, wine organoleptic quality, and hence grape quality is a key target of vineyard management. LCA-based Improvement of Technical Management Routes¹ (TMR) by choice of more environmentally friendly techniques needs to take into account this quality dimension in addition to the yield function usually considered in wine and grape LCAs. The aim of the paper is to present and discuss two proposals for including multi-criteria quality into the eco-efficiency assessment of PDO grape production in order to support the choice, design and assessment of vineyard TMRs.

Methodology: We propose the design of a combined quality index Q relating an observed multi-criteria quality to a targeted quality. A set of logical rules of inference (if...then...) was used considering several levels of correspondence of the product to the quality target. From this Index, a Quality Functional Unit (QFU=Q) and a Mass x Quality Functional Unit (MQFU= yield x Q) were defined. For application of Q to grape, a typology of grape quality targets with the different quality criteria and their levels for each grape type was established with expert-oenologists as a basis for Q. We tested the sensitivity of the two indexes to a change of quality target. We implemented the two FUs in LCA calculations on five real viticultural TMRs representing the Middle Loire Valley diversity for production of a same type of wine

Main Results: The quality index Q was calculated for the five TMRs. The two FUs derived from Q were sensitive to a change of quality target. The LCA results of the five TMRs expressed per MQFU differed from results per QFU and were sensitive to a change of quality target. Results with QFU were close to those obtained with classical FUs (1 kg grapes and 1 ha vines.1year) due to minor differences in the value of Q,

Implications, meanings, conclusions: QFU appeared too restrictive while MQFU allowed accounting for the main function of the system: the production of a given quantity of quality grapes with a target of a given type of wine.

Key words: Environment, practices, winegrowers, LCA, impacts, Protected Denomination of Origin, wine

1. Introduction

In the context of Protected Designation of Origin (PDO) wine production, wine organoleptic quality, and hence grape quality is a key target of vineyard management. Life Cycle Assessment (LCA)-based Improvement of Technical Management Routes¹ (TMR) by choice of more environmentally friendly techniques needs to take into account this quality dimension in addition to the yield function usually considered in wine and grape LCAs. However, to our knowledge, none of the published wine sector LCA studies (Gazulla *et al.*, 2010; Vázquez-Rowe *et al.*, 2012; Fusi *et al.*, 2014; Villanueva-Rey *et al.*, 2014) accounts for quality criteria in spite of the importance of both grape quality in the final wine quality (Bravdo, 2001; Guidetti *et al.*, 2010) and wine quality for wine consumers (Lockshin and Corsi, 2012; Jourjon and Symoneaux, 2014).

Quality, in its broad sense, is scarcely considered in food and crop LCA studies. Müller-Lindenlauf *et al.*(2010) included predicted milk quality as an additional impact besides classical LCA environmental impacts of milk production. Nevertheless, the most frequent option in food LCAs has been to consider quality as one of the main functions of the product. LCA calculates the potential environmental impacts of the whole life cycle of a product related to a Functional Unit (FU) which corresponds to the product's main function. (Heller *et al.*, 2013). Hence quality has been included in the FU: Charles *et al.*(2006) used an FU including a single quality criterion for wheat: "1 equivalent ton grain with 13% protein". The multi-criteria nutritional value of various foods composing diets has more recently been considered in LCAs of diets through single indices resulting from the aggregation of the nutritional values of each foodstuff related to daily consumer needs (Kägi *et al.*, 2012; Saarinen, 2012; Heller *et al.*, 2013). Some authors even included qualifying and disqualifying nutrients in the score (Van Kernebeek *et al.*, 2014). For LCAs of meals, Inaba and Ozawa (2008) proposed a comprehensive food-value index constructed on the same principle but involving the taste, nutrient balance and health function of the dishes of a meal including weighting factors determined by

¹ Technical Management Routes (TMRs): logical chain of practices managed by a farmer in a field (Sébillotte, M., 1974)

consumer survey. Nevertheless, as pointed out by van der Werf *et al.* (2014), including quality considerations in FUs remains a major challenge for the LCA food community, especially for certified productions that favour quality over volume as is typically the case for PDO wine production. Like food nutritional quality, wine-grape quality is multi-criteria (Geraudie *et al.*, 2010). Its assessment is widely used for technical decisions: harvest date, allocating grapes to different cuvees, winemaking management, and the payment of grape providers. The most common quality indicators for white grapes are the sugar and soluble acids content, and also the polyphenol content for red cultivars. However, this assessment of grape maturity is more and more complemented with on-field sensory analysis (Winter *et al.*, 2004; Le Moigne *et al.*, 2008; Olarte Mantilla *et al.*, 2012; Siret *et al.*, 2013) especially for aroma, color or texture. The health of the berries is also a key quality determinant. *Botrytis* bunch rot is especially problematic for white wine production (Hill *et al.*, 2014).

The aim of this paper is to present and discuss a proposal for the inclusion of quality in the eco-efficiency assessment of quality grape production in order to support the choice and design of vineyard TMRs to preserve the environment while maintaining the targeted quality. We considered the concept of eco-efficiency (Huppes and Ishikawa, 2005) as relevant to express the objective of improvement of environmental performance while maintaining a targeted quality level. We used it taken as the ratio between, as numerator, emissions and resource use and as denominator, the service they provide, expressed by the FU (Kicherer *et al.*, 2007). In this research, a synthetic index of grape quality and a (quality x yield) index were designed and used as FUs. This paper presents i) the two indexes formula construction and the grape quality measurements methods; ii) The results of their application to five contrasting TMRs, eco-efficiency results for mass- and quality-based FUs compared and discussed, iii) a wider discussion on methods and perspectives and iv) a conclusion.

2. Material and methods

The research work was conducted in Middle Loire Valley PDOs, on five real TMRs (TMR1 to 5) that represent the regional diversity of vineyard management for PDO Chenin Blanc dry white wine production (Renaud-Gentié *et al.*, 2014). The cases were studies for harvest 2011.

2.1 Life Cycle assessment

Data for LCA were collected from winegrowers and completed with expertise, experimental results and databases. Impacts of grapes harvested were calculated on the basis of field operations implemented in 2011, occasional operations amortized according to the frequency of implementation and operations done during non-productive periods and amortized on 30 years. The LCA was conducted from cradle to field gate. Quantification of direct emissions linked to the use of all inputs and their distribution in environmental compartments were calculated with the models proposed by (Koch and Salou, 2014) for nitrate (NO₃⁻), nitrogen dioxide (N₂O), nitrogen oxides (NO_x), phosphorus, heavy metals, and fuel combustion. Pesticide emissions were calculated with PestLCI 2.0 (Renaud-Gentié *et al.*, 2014) and Ammonia (NH₃) according to (Hutchings *et al.*, 2013) Tier2 approach.

Due to the huge quantity of data generated in this study, we present here only the results for three impact categories that proved to give very different patterns in the results: “Global warming potential at term 100 year” (GWP 100a), calculated with IPCC (Solomon *et al.*, 2007) model, “Fresh water ecotoxicity potential” (FwEtoxP) calculated with USETox™ V1.03 (Rosenbaum *et al.*, 2008) characterization method and “Abiotic resources consumption” (Res) calculated with EDIP (2003) method. LCA results of the five TMRs were calculated per ha (not presented here) and in this paper, eco-efficiency per kg are compared with the results of the two new quality-based FUs:

2.2 Grape quality index

The quality of a product is defined by (ISO, 2005) as the “degree to which a set of inherent characteristics fulfills requirement” (we prefer “Target” to “requirement” for grape quality). Still, grape quality criteria do not always have a linear relationship with the target, but rather various types of relationship, depending on the nature of the criterion. For example, sugar content can be optimal (for a given targeted wine) between 200 and 220 g/l, refused under 200 g and above 250g/l, while be accepted with a lower satisfaction between 220 g/l and 250 g/l. We propose to solve this problem by a set of logical rules of inference for each criterion considering several levels of correspondence to the

target: $C_{ig} = 100\%$: perfect correspondence, $C_{ig} = 0\%$: refused. If secondary targets are acceptable, intermediate levels are added with lower but acceptable degree $C_{ig} = e_i\%$ of correspondence to the target, as many levels of correspondence as needed must be added. For a given targeted grape type, for an assessed grape g , described by n criteria, with $i=1$ to n , the degree of correspondence C_{ig} of the grape g to the quality target, for criterion c_i is calculated according to the following formula:

$$\text{Équation 1: } C_{ig} = \begin{cases} 100 & \text{if } c_{ig} \in A_i \\ e_i & \text{if } c_{ig} \in B_i \\ 0 & \text{if } c_{ig} \in D_i \end{cases}$$

with $A_i \cap B_i = \emptyset$ and $B_i \cap D_i = \emptyset$ and $A_i \cap D_i = \emptyset$
and with $A_i \cup B_i \cup D_i$ include all c_{ig} and where:

C_{ig} = degree of correspondence to the target of criterion i for grape g .

c_{ig} = value of criterion i for grape g

A_i = set of values of criterion i corresponding to the target quality

B_i = set of values of criterion i corresponding to a secondary target quality, considered as acceptable

e_i = value of degree of correspondence of a secondary target quality to the initial target for criterion i

D_i = set of values of criterion i considered unacceptable

The limits of sets A_i , B_i , and D_i and e_i are fixed considering that the secondary target is $(1-e_i)\%$ less satisfying than the primary target.

The quality index Q_g is the global degree of correspondence to the quality requirements for the grape g and is the result of the weighted average of the degrees of correspondence to the target of each criterion:

$$\text{Equation 2 : } Q_g = \frac{\sum_{i=1}^n w_i C_{ig}}{\sum_{i=1}^n w_i} \quad \text{with: } Q_g = \text{quality index of grape } g \text{ and } w_i = \text{weight given to criterion } i$$

Applying the previous definition of the quality index to wine grapes implies defining A_i , B_i , e_i and D_i , i.e. the primary and secondary target grape types and their inherent characteristics. These characteristics are the criteria describing the grape and the gap between the primary and secondary targets (grape types that will make an acceptable quality wine but not matching the initial target).

Expert knowledge elicitation has been used (Tobias and Tietje, 2007) with nine expert practitioners who frequently deal with Chenin Blanc grapes for different middle Loire Valley PDO dry wine type production. After individual face-to-face interviews with each of the experts, a consensus session between them enabled them to reach an agreement about the primary grape quality criteria, the main grape types and their characteristics. The sugar content, aroma maturity (green, fresh fruit, cooked fruit), health and color of the berries (green to golden) were identified as the key parameters that differentiate the types of Chenin Blanc grapes for middle Loire Valley PDO dry wines. The values of the criteria corresponding to the quantitative (for sugar and rot) or qualitative (for berry color and aroma) limits of A_i for each grape type are listed in Table 1.

Table 1: Chenin Blanc grape types suitable for dry still wine in the middle Loire Valley PDO context and their characteristics according to expert consensus

Berry color	Dominant aroma	Sugar content in potential % alcohol	% of rotted berries	Type	Type code
Green or yellow	Fresh fruits	11>>13	< 10% *	Fresh dry wine	FD
Golden	Ripe fruits	13>>14.5	< 10% *	Ageing dry wine, ripe aromas	ADR
Golden	Cooked fruits, jam, honey	14.5>>16	< 10% *	Ageing dry wine, over-ripe aromas	ADOR
Golden	cooked fruits, jam, honey	14>>16	Noble rot	Ageing dry wine, noble rot aromas	ADN

*if it is grey mould evolving into noble rot, then rot is accepted

We translated the qualitative values given by the experts for color and aroma into quantitative ones relative to the existing scales that our sensory analysis panel was trained to use.

The experts named the grape types according to the wine that they were the most suitable for. Correspondence between grape typology and single criteria assessment was carried out through the inference rules presented in Table 2. These rules were determined for the type of wine targeted by the growers in this study: type ADR = dry quality wine for ageing with ripe aromas as main target and types FD = fresh dry quality wine and ADOR = ageing dry wine with over-ripe aromas, as acceptable alternative targets (Table 2).

Table 2: Rules of inference determined for grape type ADR as the primary target and types FD and ADOR as secondary targets with both $e=50\%$

Criterion number	Measured parameter (scale or unit)	A_i	$e\%$	B_i	$e\%$	D_i	$e\%$
c_1	Berry color (/10)	$4 < c_1 < 9$	100	$2 < c_1 < 4$	50	$2 > c_1$ or $c_1 > 9$	0
c_2	Dominant aroma	$c_2 =$ ripe fruits	100	$c_2 =$ fresh fruits or $c_2 =$ cooked fruits, jam, honey	50	$c_2 =$ vegetal, or earthy/mouldy	0
c_3	sugar content (potential %alc.)	$3 \leq c_3 \leq 14,5$	100	$11 < c_3 < 13$ or $14,5 < c_3 < 16$	50	$11 > c_3$ or $c_3 > 16$	0
c_4	rot (%)	$c_4 < 10$	100			$c_4 > 10$	0

A_i = set of values of criterion i corresponding to the target quality, B_i = set of values of criterion i corresponding to a secondary target quality, considered acceptable, e_i = value of degree of correspondence to the initial target quality, D_i = set of values of criterion i considered unacceptable

2-3 Quality criteria measurements

Based on these criteria, quality assessment was carried out on the grapes sampled in the 5 selected plots at harvest time in October 2011. The sugar content was measured on a representative sample of 200 berries. The health of the berries was visually assessed on each bunch of 40 vines. An additional representative sample of 300 berries was used for berry sensory analysis from which the berry color assessment results were extracted. Must sensory analysis results were used to determine aroma. The musts were obtained by pressing grapes harvested on the same 40 vines at the same date. The berries and musts were assessed on a 0 to 10 continuous scale for each parameter by a trained expert panel of 11 judges for the berries and 13 judges for the musts. The attributes selected as corresponding to the experts grape typology criteria - berry color, must vegetal aroma, white fruit (for fresh fruit) and prune (for cooked fruit) aromas- were found discriminating in the analysis of variance with P-values lower than 0.01.

2.4. Application of quality Functional Units to LCA results

The environmental impact for each impact category relative to the quality index, Q_g was obtained by dividing the “per ha” LCA results by Q_g (%). The mass x quality (MQ_g) FU was derived from the calculation of a mass-quality index, MQ_g , by multiplying the annual grape yield by Q_g (%). The environmental impact results, in this case, were obtained by dividing the “per ha” LCA results by MQ_g giving results per MQ_g FU.

3. Results

3.1. Quality measurement results and quality and mass x quality index calculation

The results of the sensory analysis concerning aromas showed no differences between the TMRs in Fisher’s Least Significant Difference test. These results were analyzed to identify the dominant aroma, in accordance with the expert description of grape types. The five TMRs yielded grapes dominated by a fresh fruit aroma. Table 3 reports the construction of the quality index Q_g based on the results of measured quality criteria and the dominant aroma. Two levels of Q_g appear for the five TMRs (62.5 and 75) due to a difference in berry color, those of TMRs4 and 5 being more golden.

Table 3: Quality results and Quality- and Mass x Quality indexes calculation (ADR grape type target, for the 5 TMRs in 2011)

Criterion number	Parameter (scale or unit)	TMR1		TMR2		TMR3		TMR4		TMR5	
		c_{iTMR1}	$e\%$	c_{iTMR2}	$e\%$	c_{iTMR3}	$e\%$	c_{iTMR4}	$e\%$	c_{iTMR5}	$e\%$
c_1	Color (/10)	3.95	50	3.40	50	3.59	50	5.25	100	4.16	100
c_2	Dominant aroma	Fresh fruit	50	Fresh fruit	50	Fresh fruit	50	Fresh fruit	50	Fresh fruit	50
c_3	Sugar content (pot. %alc.)	12.31	50	12.03	50	11.8	50	12.28	50	12.31	50

c₄	Rot (%)	14*	100	0.8	100	5	100	2.3	100	0.6	100
Q_g	Quality index		62.5		62.5		62.5		75		75
Yield 2011	(kg/ha)		6440		5250		7500		5880		5250
MQgADR	Mass x Quality index		4025		3281		4688		4410		3938

* turning to noble rot

The TMRs can be divided into two groups with the same Q_g : TMRs 1 to 3 and TMRs 4 and 5. The mass-quality index (MQg) results from $Q_g \times$ grape yield (Table 3), TMR 2 showed the smallest MQg while TMR3, due to a very high yield, had the highest MQg.

3.2. Comparison of TMR eco-efficiency according to the three different Functional Units

The results of the comparison between the five TMRs depend on the FU chosen (Figure 2). The hierarchy was the same between TMRs 1, 3 and 5 for the 3 FUs in the 3 impact categories. The most important change between the FUs concerned TMR2's GWP 100a and FwEtoxP impacts which were 13 to 30% higher relatively to other TMRs with the Q_g FU than with the other FUs. TMR 3 remained the most eco-efficient for GWP 100a and Res whatever the FU, TMR1 remained the least eco-efficient for Res. TMR4 remained in an average position in GWP 100a and Res impact categories, and TMR5 remained the least eco-efficient for GWP 100a.

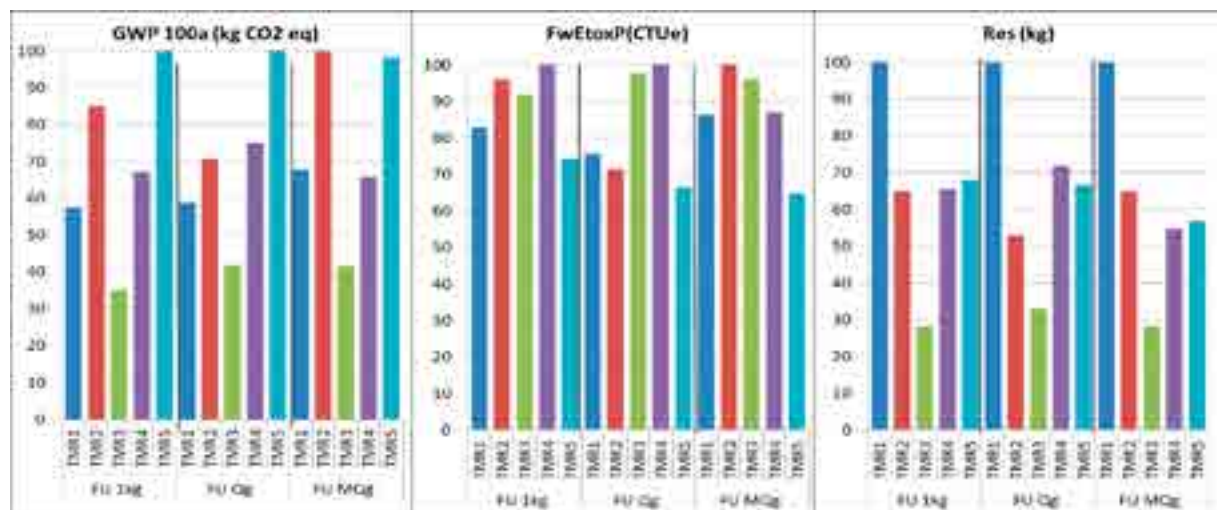


Figure 2: Eco-efficiency results of the five TMRs for GWP 100a, FwEtoxP and Res impact categories, according to three different Functional Units (FUs): 1 kg grapes, quality index: Q_g , 1 kg grapes $\times Q_g$: MQg. Results are in % of the impact of the most impacting TMR

4 Discussion

The quality index Q_g results for the five TMRs showed only minor differences mainly because of the standardization of harvest dates. Higher differences would have caused more contrast between the eco-efficiency results. The eco-efficiency results showed that the higher the Q_g , the better the eco-efficiency results in Q_g FU. This is true for the MQg FU but modulated by the yield. The yield had the same influence on eco-efficiency: for the same per ha impact, the higher the yield, the better the eco-efficiency for mass and mass quality FUs. Consequently the TMRs that combine high yield and high Q_g have the greatest gain in eco-efficiency when changing from 1 ha FU to MQg FU.

To our knowledge, the appreciation of grape quality related to a defined target has not been formalized to date in a specific indicator. This process is carried out spontaneously by the production stakeholders when they harvest or process the grapes, but the targets are often not precisely described, being more an objective fixed on an unconscious scale based on experience. This approach is generic to any grape, provided the criteria and thresholds are adapted to the cultivar and the regional, or even

local, and annual context. It can also be applied to any other product. However, this first proposal of an indicator construction might be improved in the future by the use of fuzzy logic (Zadeh, 1965), to avoid the threshold effects, and enable a gradual progression of “e” from the primary target to secondary ones and refused grapes (Coulon-Leroy *et al.*, 2012; Guillaume and Charnomordic, 2012).

Between $e = 100\%$, perfect correspondence to the primary target, and $e = 0\%$, off target, we fixed the value $e = 50\%$ for a grape corresponding to an acceptable secondary target. This threshold could be adapted, for generic situations, “e” can be determined with the experts who contribute to the determination of grape types and criteria. For specific studies, “e” should be adapted with the final user, because it can be dependent on the markets of the wine estate which determine the difference of commercial value between the primary and secondary target grapes or the corresponding wines.

The use of quality FUs in the assessment of TMR environmental performance improvement has the advantage that the advisor or the decision maker keeps in mind the quality objective of the production. These FUs are more appropriate than mass and surface FUs when considering the central function of grape production, especially in a premium wine production context like PDOs. The Q_g FU reflects only a quality level, without any reference to the yield. Using this FU reflects that the grape production exclusive -or very primary- objective is quality, whatever the yield. This can be the case in some specific situations (ultra-premium quality wines, or high quality oriented part of a vineyard being only a part of the income of the farm for example). However, in most situations, both quality and yield are needed to secure the income from the vineyard activity and to satisfy the markets in quantity. The second FU, (MQ_g), mixing mass and quality, permits to account for both objectives. According to (Heller *et al.*, 2013), this type of FU is well suited to comparing agricultural production methods. Table reports the main aspects we propose to account for in the choice of FU in quality viticulture.

Table 4: Advantages of four Functional Units for quality grape production TMRs

Functional unit	Advantage/usage
Surface area : 1ha of vineyard	minimizes impacts when cultivating a given surface area, accounts for multi-functionality of viticulture (landscape, ecosystem services), adapted to communication of LCA results to winegrowers
Mass : 1kg of grapes	minimizes impacts of a mass of grapes, considers the economic importance of yield, adapted to communication of LCA results to consumers
Mass with a quality level: 1kg grape x Q_g	minimizes the impacts of a mass of grapes, considers the central function of quality wine TMRs, avoids decreasing the quality when improving environmental performance,
Quality level : Q_g	avoids decreasing the quality when improving environmental performance

However, surface-based FUs are complementary to quality FUs to account for multi-functionality of viticulture, and also to communicate the results to the producers in a unit that meets their usual technical decision unit, 1 ha. These quality based FUs cannot totally replace per ha FU for another reason, the variability of grape composition due to climatic conditions (Jones and Davis, 2000). A climatic accident (like heavy rain before harvest) can cause a severe decrease of yield and grape quality which cannot be attributed to the TMR. In this last case, yield and surface FUs will be more reliable than MQ_g FU. Moreover, yield also varies for climatic reasons (Makra *et al.*, 2009), so MQ_g FU cumulates two sources of variations linked to climatic conditions which can be different climatic events for yield and grape quality. The climate of the year is an important variation factor in different aspects accounted for in grape LCA (Vázquez-Rowe *et al.*, 2012; Renaud-Gentié *et al.*, 2014), and the TMR itself is adapted to the climatic conditions by the growers. Accordingly, before planning important changes in the TMR on the basis of eco-efficiency, these results must be considered in the climatic context of the year in which they were obtained and LCA must be conducted on several climatically contrasting years unless a way is found to simulate the effects of

climate on these different parameters. Finally, one may also consider conducting the LCA separately from the quality assessment and combine both assessments afterwards (Beauchet *et al.*, 2014).

The effects of yield variability on LCA results may also be quantified through a sensitivity analysis of LCA results per ha, exploring the range of the known variability of yields instead of using the mass FU.

5. Conclusion

We have proposed a new grape quality assessment approach for inclusion in the eco-efficiency assessment of quality vineyard technical management routes. The quality indicator Qg expresses the degree of correspondence of the harvested grapes to the quality target assigned to the TMR. A typology of grapes was established with experts as a basis for this Qg indicator. The five contrasting vineyard TMRs, representing the middle Loire Valley diversity gave different quality-based eco-efficiency performances close to those obtained with classical FUs (1 kg of grapes and 1 ha of vines per year) due to minor differences in Qg.

Two functional units for life cycle assessment of TMRs were derived from this indicator. A quality FU: Qg, and a mass x quality FU: MQg including the yield. The QgFU alone appeared too restrictive while including the yield in this quality FU accounted for the main function of the system, i.e. the production of a given quantity of quality grapes for a given type of wine. Even though PDO wines do not respond to industrial quality standards, the wine growers and winemakers have a quality target in mind which is adjusted every year to the quality potential given by the vintage conditions. This adjustment of the quality target can be made whenever necessary. However including quality in TMR evolution or eco-conception demands further work that may involve the use of fuzzy logic in the indicator construction to avoid threshold effects and for grape quality prediction knowing the TMR.

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133. Using Acoustic Diversity in Life Cycle Assessment of Agriculture: Case Studies of Oil Palm Production in Indonesia

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ABSTRACT

Recent important accomplishments in the assessment of biodiversity impact due to land use and land use change within the framework of life cycle assessment (LCA) include the publication of global characterization factors (CFs) and the establishment of procedures to use expert knowledge. However, the CFs are too rough to be used in management decisions and the use of expert knowledge, which can be considered as a method to resolve the problems with using the CFs, has some difficulties in dealing with the subjective information. In this paper, we appraise the possibility of constructing physical biodiversity indicators for LCA of agriculture using rapid acoustic assessment. In order to measure biodiversity in different land use (palm oil plantations, enclave forests, and natural forests), we conducted field recording and digital information analysis; the Acoustic Diversity Index (ADI) was utilized for quantifying each sound file. Case studies were carried out in South Sumatra and West Kalimantan. In average, the ADIs for plantations were lower than those for the forests, although the results were dependent on recording locations and dates. In addition, we realized that ADIs at dawn and dusk are important in assessing diversity and that paying attention to periodicity or rhythm of nature is crucial. Biodiversity assessment using acoustics is expected to establish the relationships between management practices and biodiversity, which will be useful in comparative LCA of agricultural production systems.

Keywords: biodiversity, Acoustic Diversity Index (ADI), plantation, forest, land use

1. Introduction

Global characterization factors (CFs) have recently published for assessing biodiversity impact due to land use and land use change within the framework of life cycle assessment (LCA). For example, de Baan et al. (2013) presented an approach to measure the potential regional extinction of nonendemic species caused by land occupation and transformation impacts within each WWF ecoregion and allocated the total damage to different types of land use per ecoregion. Furthermore, Chaudhary et al. (2016) calculated vulnerability scores for each ecoregion in order to take endemism into account, in addition to the quantification of regional species loss due to land occupation and transformation.

However, if we use the CFs to compare biodiversity impacts of several agricultural production systems in the same ecoregion, the impact scores per product unit are dependent only on crop yields. The reason is that the CFs were prepared for each ecoregion and for each land use type (e.g., agriculture, pasture, urban, and managed forest) as explained in Michelsen et al. (2014). It implies that the CFs are too rough to be used in assessing management practices and making management decisions. Furthermore, an environmental labelling policy using the CFs is not incentive compatible, because farm managers can improve indicator values (biodiversity scores per product unit) by increasing crop yields rather than practicing environmentally-friendly management.

Methodological development to use expert knowledge is another important achievement in the assessment of biodiversity. For example, Jeanneret et al. (2014) proposed a method to assess agricultural production systems, in which biodiversity scores are calculated as the products of the impacts (subjective rating) of management options on indicator-species groups, and habitat and management coefficients (subjective rating) for indicator-species groups. Fehrenbach et al. (2015) developed a method using the concept of hemeroby (naturalness), which has the hierarchical evaluation structure of criteria and metrics. Categorical scores are given to each metrics and overall average values are calculated. Lindner et al. (2014) proposed a combined use of geographical information and subjective evaluation. In their method, biodiversity impact is defined as the product of the ecoregion factor and the additive reciprocal of biodiversity potential, the average value of single attribute value functions that explain the relations of management intensity on biodiversity.

In essence, these methods based on subjective judgements can be classified as a constructive approach based on multi-criteria assessment (Beinat, 1997) and the methods mentioned above result in the development of constructed scales under the situation where natural measures do not exist (Keeney, 1992). Therefore, although these methods can resolve the problems with the use of the

global CFs in agricultural production systems, they are based on human preferences and do not provide physical facts. The use of preferences does not imply that the methods are useless. Rather, it means that the methods should be appropriate for prescriptive purposes in decision support contexts.

In this paper, we propose another way. We appraise the possibility of constructing physical biodiversity indicators, instead of developing indicators using subjective judgments. In this case, our intention is to develop an approach that makes comparative LCA of agricultural production systems possible, as in the case of the latter methods, rather than estimating environmental impacts caused by land-use categorical changes at the global scale. Our strategy is to construct physical indicators using labour-saving techniques and, therefore, we employed rapid acoustic assessment (Sueur, 2008), which is a digital information analysis without identifying flora and fauna based on simple field recording.

2. Methods

2.1. Assessment framework

We use the general framework for land use change impact assessment within LCA (Figure 1), which distinguishes between land transformation and land occupation by depicting a three-dimensional figure with time, land quality, and area. Our purpose in this study is to appraise the possibility to measure the quality using acoustics.

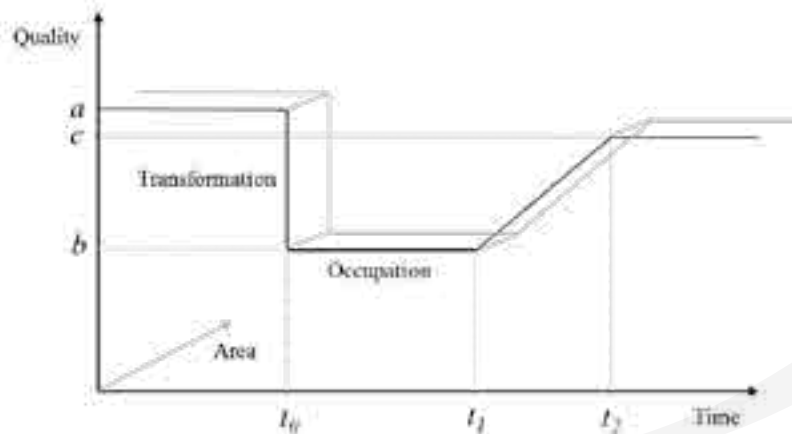


Figure 1: Simplified illustration of land use change impact assessment framework within LCA

Production systems analysed in this study using the above framework are oil palm production systems in Indonesia. Oil palm plantations between the time t_0 and t_1 (quality b) are compared with forests (quality a), which can be considered as a reference. Although pristine forests are precisely equivalent to the reference situation (quality a), we used natural forests near the plantations and enclave forests (forests within plantations), because the recording before transforming the land into plantations is not feasible and it was actually difficult to find pristine forests near the plantations analysed in this study.

2.2. Data collection

Sound data were collected in the Dawas plantation, South Sumatra and the Ngabang plantation, West Kalimantan. In the latter case, the enclave (secondary) forest in the Parindu plantation was also included. We made recordings on August 2015 using three digital recorders (SONY PCM-D100). Since natural sounds exceed the limit of the conventional CD quality (sampling frequency: 44.1 kHz, quantifying bit number: 16 bit), as well as human audible bandwidth (until about 20 kHz), the recorders were set at the mode of 192 kHz and 24 bit. Although it was difficult to record the sounds until 96 (192 divide by 2) kHz, because of the performance of the built-in microphones, we confirmed that the frequencies until at least about 50 kHz were observable. We recorded stereo sounds at the microphone angle of 120° , which are equivalent to recordings at two adjacent places. The recorders

were held horizontally at 1.5 m height. The recording locations are shown in Table 1. Outdoor loudspeakers in the plantations were not used during the recording.

Table 1: Locations for recording

Location	Dawas		Ngabang-Parindu	
	1st trial (Aug. 10-11)	2nd trial (Aug. 12-13)	1st trial (Aug. 19-20)	2nd trial (Aug. 20-21)
Plantation	2° 31' 59.31" S 103° 48' 12.39" E	2° 31' 0.96" S 103° 48' 46.99" E	0° 22' 12.47" N 109° 54' 31.71" E	0° 21' 19.72" N 109° 54' 31.32" E
Enclave forest	2° 31' 36.33" S 103° 48' 17.54" E	2° 31' 34.28" S 103° 48' 16.43" E	0° 15' 35.36" N 110° 18' 46.46" E	0° 15' 35.36" N 110° 18' 46.46" E
Natural forest	2° 30' 59.10" S 103° 48' 25.53" E	2° 30' 52.89" S 103° 48' 24.82" E	0° 22' 6.65" N 109° 47' 27.86" E	0° 22' 6.65" N 109° 47' 27.86" E

The actual land use history for the plantations are as follows. The Dawas plantation was transformed from secondary and rubber community forests in 2006, while the Ngabang and Parindu plantation was transformed from primary forests in 1980. The first generation of oil palm trees in the Ngabang plantation was planted in 1982 and 1983. After some unproductive phases, oil palm trees were replanted in 2005, 2009, 2010, 2011, and 2012. In the recorded location, the trees were replanted in 2005. The first generation in the Parindu plantation was planted in about 1980.

In order to synchronize the time information in the recorded files among the three recorders, the identical sounds were recorded in the three recorders. Using the sounds, the time stamps in the recorded files were adjusted in milliseconds. Each recorded data file (2 GB, the maximum size of sound files), which lasts about 30 minutes, was separated into one-minute files (30 pieces). The pieces that contain noises such as human voices during device setting were not used in the analysis.

2.3. Acoustic index to measure biodiversity

The Acoustic Diversity Index (ADI) (Villanueva-Rivera et al., 2011) was utilized for quantifying each sound file. In this index, the spectrogram of an audio file was divided into many frequency bands and the proportion of sound in each frequency band, which represent a specific "species", was used to calculate the Shannon index. We used the package 'soundecology' in the statistical software *R* for calculate the index. Although the default maximum frequency in calculating the index was 10 kHz, we used the value of 50 kHz, because the sounds until 50 kHz were actually recorded in the files.

3. Results

We calculated average ADI values for day and night separately, because there were differences in the ADI values during the day and those at night (Table 2). The time for sunrise and sunset was calculated using the latitude, longitude, and elevation at the recording point. In Dawas, the results were different from what we expected. Although the ADI value during the day for natural forest was higher than that for plantation in the 1st trial, it is not applicable to the other cases. In contrast, anticipated results were available in Ngabang-Parindu. The descending order of the ADI values was natural forest, enclave forest, and plantation in many cases.

Table 2: Average ADI values at day and night for each location

Location	Dawas				Ngabang-Parindu			
	1st trial		2nd trial		1st trial		2nd trial	
Day/night	Day	Night	Day	Night	Day	Night	Day	Night
Plantation	2.464	3.183	2.785	3.144	2.289	2.456	2.831	2.220
Enclave forest	2.763	2.529	2.712	2.575	3.199	2.885	3.257	2.810
Natural forest	3.068	3.136	2.742	2.839	3.068	2.976	2.953	3.049

Since we recognized that the ADI can identify the higher activities of animals during dawn and dusk, we calculated average ADI values for dawn, defined as two hours after sunrise, and dusk, defined as two hours after sunset. In this case, the ADI values for natural forest were higher than those

for plantation in most cases. The differences in the ADI values between day/night and dawn/dusk were larger for natural forest than those for plantation. It implies that part of natural rhythm was disappeared in plantations.

Table 3: Average ADI values at dawn and dusk for each location

Location	Dawas				Ngabang-Parindu			
	1st trial		2nd trial		1st trial		2nd trial	
Dawn/dusk	Dawn	Dusk	Dawn	Dusk	Dawn	Dusk	Dawn	Dusk
Date	Aug. 11	Aug. 11	Aug. 13	Aug. 12	Aug. 20	Aug. 19	Aug. 21	Aug. 21
Plantation	2.961	3.435	2.904	3.401	2.397	1.946	2.752	3.156
Enclave forest	2.858	3.422	2.717	3.799	2.747	3.522	2.879	3.507
Natural forest	3.432	3.312	3.423	3.570	2.838	3.730	3.030	3.636

In addition, the results suggest that the differences between the two types of forests were smaller than those between the plantations and forests. It means that the both forests have a generality as forests and that the differences in the quality explained in Figure 1 between plantations and forests were detected, although there were exceptions.

4. Discussion

We demonstrated the possibility of acoustics in biodiversity appraisal within the framework of LCA, although further studies are necessary for the use of acoustic diversity in, for example, environmental labelling policies based on LCA. We will discuss how the approach using acoustic diversity can be further developed.

4.1. Definition of the reference state

One of the reasons why the ADI values for the forests were in some cases lower than those for plantations may be attributed to the degraded condition of natural forests surrounding oil palm plantations. Although we recorded the sounds in the two types of forests (natural and enclave forests), because we thought that the acoustic diversity in forests are dependent on locations and because it was difficult to find the pristine forests near the plantations, we have to be explicit about the difference between natural vegetation and potential natural vegetation in biodiversity assessment of land use (Souza et al., 2015). The both types of forests in this study should be recognized as the latter vegetation. Since most natural forests are used by local people for hunting animals and for logging precious woods, further investigation is necessary for understanding the relationship between land use intensity (forest management) and biodiversity.

4.2. Selection of acoustic indices

Another reason for the lower value cases can be related to the definition of acoustic indices. Because of the difficulty in finding a single index summarizing all biodiversity information, Sueur et al. (2014) recommend the complementary use of several α -diversity indices. In this study, we complementarily applied the Acoustic Complexity Index (ACI) (Pieretti et al., 2011). In average, the ACIs for plantations were larger than those for natural forest, although we did not judge that it implies biodiversity in plantations is richer than that in natural forests. The important differences were found in scale parameters (e.g. standard deviations), rather than location parameters, at dawn and dusk, although this trend is also applicable to the case of ADIs. That is, in the morning and evening, standard deviations for the natural forest were larger than those for the plantation. Further research on acoustic indices is necessary in clarifying the relationship between sound states and indicator values.

4.3. Implications to inventory analysis and impact assessment

Inventory data for land use and land use change contain the information about occupation and transformation (“from” and “to”) in the section for “inputs from nature”. The results suggest that

acoustic indices can be used as a proxy for the quality of biodiversity (the vertical axis in the assessment framework in Figure 1). However, applicability of acoustic diversity is not limited to this way of inventory construction.

One is the description as “outputs to nature”. In this case, understanding of multifunctionality plays an important role and further consideration on ecosystem services is necessary. Another way is the establishment of impact assessment based on the relationships between management practices (information such as fertilizer and pesticide application in the section for “inputs from technosphere”) and biodiversity.

5. Conclusions

The results of this study indicate that acoustic diversity can be applicable to biodiversity assessment within the framework of LCA of land use and land use change and that attention has to be paid to periodicity or rhythm of nature. Although this research direction will be important in establishing the relationships between management practices and biodiversity on the basis of site-specific conditions, there may be difficulty in applying to the locations where biodiversity is sparse. For example, crop production systems in temperate regions might be difficult to use acoustic indices, although acoustic analysis can be useful in, for example, identifying animal species.

An important implication of this study is that in assessing biodiversity, the concept of landscape and soundscape becomes crucial. Although LCA has already been used at the regional levels, agricultural provision of ecosystem services at the landscape/soundscape level can be embedded in LCA.

Although rapid biodiversity appraisal using acoustics is promising in the sense that it can gather site-specific biodiversity data without using much time and cost, there are limitations due to acoustics; the recorded data are limited to sounds produced by animals when moving, communicating, and sensing their environment. Further justification of methodology through the identification of animal species in the recorded sites and the integration with the other sensing technologies will be important research topics.

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176. Combined Nutritional and Environmental Life Cycle Assessment of Fruits and Vegetables

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ABSTRACT

Nutritional health effects from the ‘use stage’ of the life cycle of food products can be substantial, especially for fruits and vegetables. To assess potential one-serving increases in fruit and vegetable consumption in Europe, we employ the Combined Nutritional and Environmental LCA (CONE-LCA) framework that compares environmental and nutritional effects of foods in a common end-point metric, Disability Adjusted Life Years (DALY). In the assessment, environmental health impact categories include greenhouse gases, particulate matter (PM), and pesticide residues on fruits and vegetables, while for nutrition we consider all health outcomes associated with fruit and vegetable consumption based on epidemiological studies from the global burden of disease (GBD). Findings suggest that one fruit/vegetable serving increase may lead to substantial nutritional health benefits even when considering uncertainty; 35 μ DALY/serving_{fruit} benefit compared to a factor 10 lower impact. Replacing detrimental foods, such as trans-fat and red meat, with fruits or vegetables further enhances health benefit. This study illustrates the importance of considering nutritional effects in food-LCA.

Keywords: LCA, fruits, vegetables, nutrition, human health

1. Introduction

Dietary risks are leading the global burden of disease (GBD) with about 12 million annual attributable deaths globally, illustrating the strong relationship between dietary patterns and human health (IHME, 2015). Diet- and food-related life cycle assessments (LCA), up to date, mainly focus on human health impacts associated with environmental emissions. The ‘use stage’ of food products, although part of a product’s life cycle, does not typically consider nutritional effects that occur with consumption and can have substantial effects, positive and/or negative, on human health (Stylianou et al. 2016). Incorporating a nutritional assessment in diet- and food-related LCA would provide a comprehensive and comparable human health effect evaluation of food items and diets that could yield more sustainable dietary decisions.

The nutritional value and beneficial human health effects associated with fruits and vegetables consumption is widely recognized and evident by numerous recommendations urging consumers to increase their fruit and vegetable daily intake (USDHHS and USDA 2015; Nordic Council 2014). However, current conventional fruit and vegetable production methods require the application of pesticides which yields residues that have the potential of inducing human health impacts, a continuous concern requiring a constant monitoring and evaluation. As a result, increased consumption of fruits and vegetables – although considered as a healthier dietary option – could result in higher exposures to a wide variety of pesticides, alongside other environmental health related impacts associated with corresponding increase in production and distribution. The aim of this study is to assess the overall human health trade-offs between potential environmental and nutritional effects associated with one serving increase of fruits (141 g) and one serving increase of vegetables (123 g) over the average European consumption.

2. Methods

2.1. Framework for comparing environmental and nutritional effects of food

The Combined Nutritional and Environmental Life Cycle Assessment (CONE-LCA) framework evaluates and compares in conjunction environmental and nutritional effects of food items or diets expressed in a common end-point metric, Disability Adjusted Life Years (DALYs) (Stylianou et al. 2016). In this case study, the assessment starts from one serving of fruits and one serving of vegetables as a functional unit (FU) that are associated with environmental health impacts due to life cycle emissions of e.g. greenhouse gases (GHG) and particulate matter (PM) as well as chemical intake from pesticide residues on vegetal food. Nutritional impacts and benefits are assessed in parallel based on published epidemiology data that directly link fruit and vegetable consumption to

nutritional health outcomes such as cardiovascular diseases and neoplasms, starting from the GBD. Figure 1 illustrates the general CONE-LCA framework along with the framework used in the case study investigated in this paper (represented in the red dashed box).

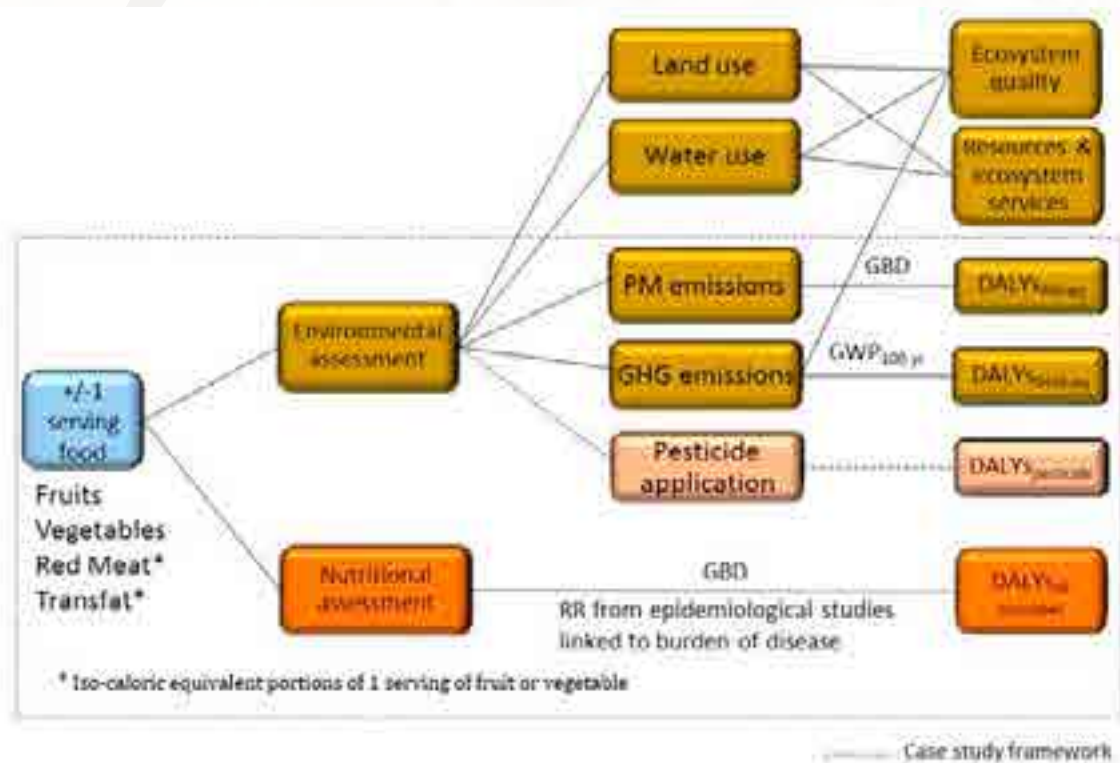


Figure 1. Graphical representation of the CONE-LCA framework. The dashed box denotes the scope of the presented case study

2.2. Case study: fruits and vegetables consumption in Europe

2.2.1. Dietary scenarios

The European Food Safety Authority (EFSA) Comprehensive European Food Consumption Database (EFSA, 2015) reports the average adult European diet. According to the latest data the current population-weighted European daily diet is consisted on average from 195 g of total fruits (fresh and processed) and 218 g of total vegetables (fresh and processed). These intakes correspond to a 1.2 and 1.8 servings of fruits and vegetables daily intake, respectively, which are below the dietary recommendation guidelines (USDHHS and USDA 2015; Nordic Council 2014).

To assess a potential dietary shift towards dietary guidelines, we investigate the case of one serving increase of fruits (141 g) and one serving increase of vegetables (123 g) over the average European consumption and evaluate the corresponding health effects. To consider for more realistic dietary scenario assessment, in addition to the increase in fruit or vegetable consumption, we also evaluate two substitution scenarios based on a default iso-caloric equivalent basis as a first proxy of a) trans-fat and b) red meat, two high burden dietary risk factors in the GBD (IHME, 2015).

One serving of fruits or vegetables has a nutritional energy content of respectively 102 or 74 calories, respectively. Hence, we investigated the following per person daily dietary scenarios:

- A. Add a serving of fruits (or vegetables), with no change to the rest of the diet.
- B. Add a serving of fruits (or vegetables) while subtracting an equal caloric quantity of trans-fat.
- C. Add a serving of fruits (or vegetables) while subtracting an equal caloric quantity of red meat.

2.2.2. Environmental assessment

The environmental assessment in our analysis follows a traditional LCA approach. Food group-specific emission factors for GHG and ammonia (NH₃) were retrieved from the work by Meier and Christen (2012), accounting for production, processing, packaging and transportation to retail. Other PM-related emission (primary PM_{2.5}, NO_x, SO₂) were extrapolated from GHG as described by Stylianou et al. (2016) since such information is not routinely reported in food LCA studies. Emissions are coupled with characterization factors (CF) to give human health impact in DALYs/FU. More specifically, CFs from Gronlund et al., (2015) and Bulle et al., (manuscript in preparation) were used for PM-related and a 100-year horizon global warming health impacts, respectively.

In regards to the pesticide residue exposure, human health impacts are determined based on the work by Fantke et al. (2012). Human health impacts have been quantified by crop class accounting for human exposure resulting from 133 pesticides applied in 24 European countries in 2003 and individual substances distinct environmental behavior and toxicity. Active ingredients found in pesticides were then associated to publically available consumption data. Adjusting for current European fruits and vegetables consumption (EFSA, 2015) and under a linear assumption, the human health impact estimate from pesticide residues on fruits and vegetables is 2.5×10^{-7} and 2.4×10^{-6} DALYs/year/person, respectively.

2.2.3. Nutritional assessment

For the nutritional assessment, there are numerous epidemiological studies investigating the association of fruit, vegetable, trans-fat, and red meat intake with various health outcomes. In our study, we focus on the various health outcomes considered in the GBD for each of these dietary risk factors. More specifically, cardiovascular diseases are the main health outcome associated with low fruit (86%), low vegetable (100%), and high trans-fat (100%) consumption while high red meat consumption is associated with diabetes (60%) and colorectal cancer (40%). We combine the total European burden reported by the GBD for each food group (IHME, 2015) with the corresponding current consumption (EFSA, 2015) to estimate the overall nutritional health effect, benefit or impact, per FU, accounting for the respective theoretical minimum risk intake (as defined by the GBD in the work by Forouzanfar et al., 2015).

3. Results

3.1. Environmental assessment: PM-related health impacts

Figure 1 illustrates the PM-related human health impact in μ DALY/serving corresponding to the iso-caloric food portions. Our analysis indicates that one serving of fruits is linked to a total of 0.065 g PM_{2.5}-eq, corresponding to a health impact of 0.08 μ DALY, mainly due to NH₃ (38%). The iso-caloric red meat equivalent is associated with substantially higher health impact (about 7 times), with NH₃ as the main PM-precursor contributor at 85%. For the vegetable serving we estimate PM-related health impact of 0.03 μ DALY/serving, again mainly attributable to NH₃ emissions (40%). The iso-caloric red meat equivalent had 6.5 times higher impact than a serving vegetable. The PM-related health impacts for the trans-fat substitutions are considered negligible.

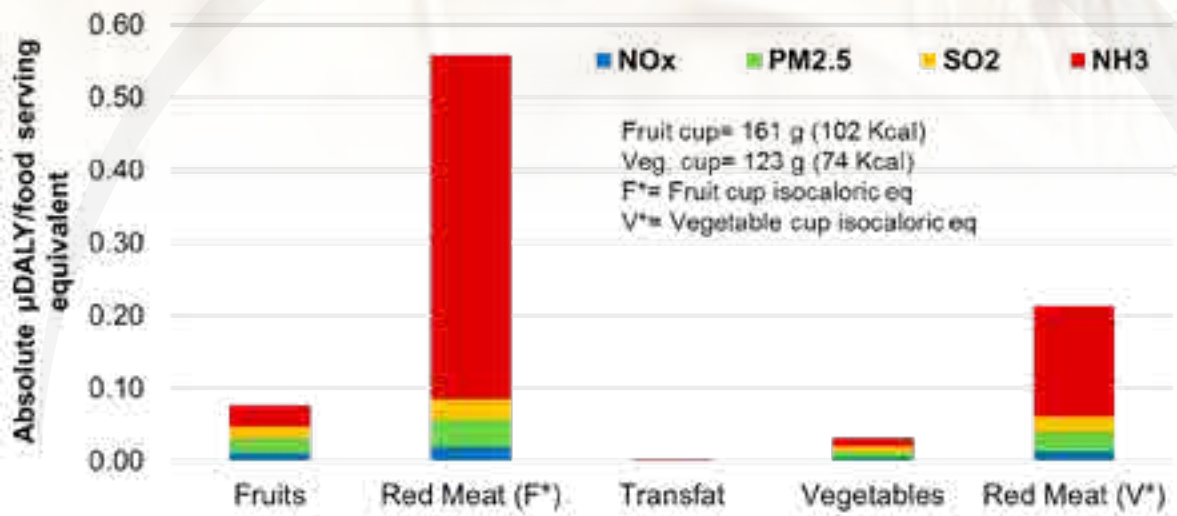


Figure 2. Particulate matter related human health impact measured in associated μ DALY/serving with an iso-caloric equivalent portion of distinct food intakes: (1) fruits, (2) vegetables, (3) trans-fat, (4) red

3.2. Nutritional assessment

A linear dose–response relationship relates food intake, expressed in g/person/day to all cause outcomes impact in DALYs/person/day. We use such dose–response functions to estimate the nutritional health burden attributable to food intake shift from the current consumption. For fruit consumption, we found that one serving increase in intake over current consumption would result in a benefit of 34.7 μ DALY (Figure 2). The analogous estimate for one additional serving of vegetable is a benefit of 17.2 μ DALY. Using the same approach for the considered substitutions, the fruit (or vegetable) iso-caloric reduction in trans-fat and red meat portion is associated with reductions in health impacts of 0.5 (0.4) and 1.5 (1.1) μ DALY, respectively.

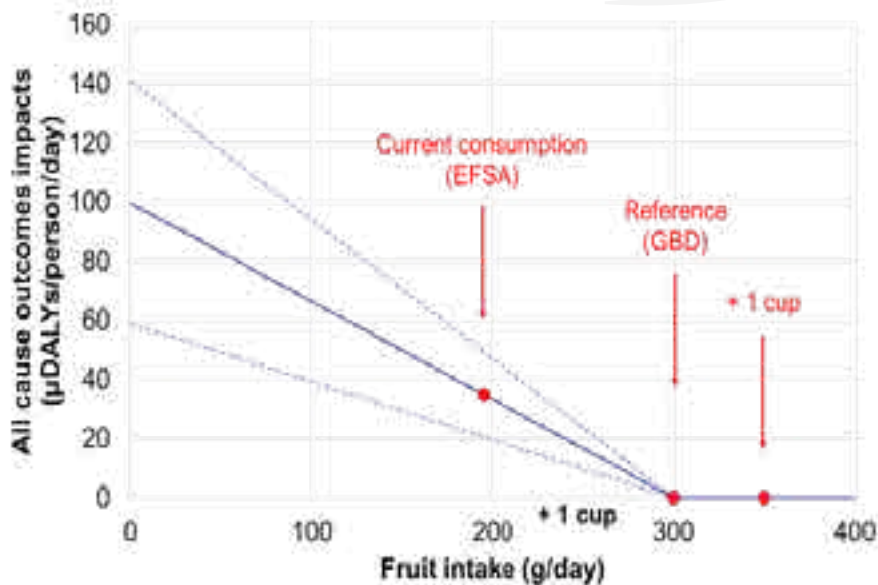


Figure 3. Dose–response function for fruit intake and all cause outcomes, with 95 % confidence intervals shown as dashed lines.

3.3. Overall comparison

Figure 3 represents the overall environmental and nutritional human health trade-offs associated with one serving of fruits without and with substitution scenarios. Adding one serving of fruits to the present European diet may lead to a considerable nutritional health benefits ($35 \mu\text{DALY}/\text{serv}_{\text{fruit}}$). The nutritional benefit is moderately enlarged when we consider the substitution scenarios since the substituted food items are associated with negative health effects and reduction in intake results in avoided human health impact. Overall environmental health impacts are substantially smaller, about an order of magnitude lower, compared to the nutritional benefits in each scenario. Benefits exceed impacts even when considering an uncertainty factor of 400 for the impacts of pesticide residues. Similar results are found for the case of adding one serving of vegetables to the average diet.

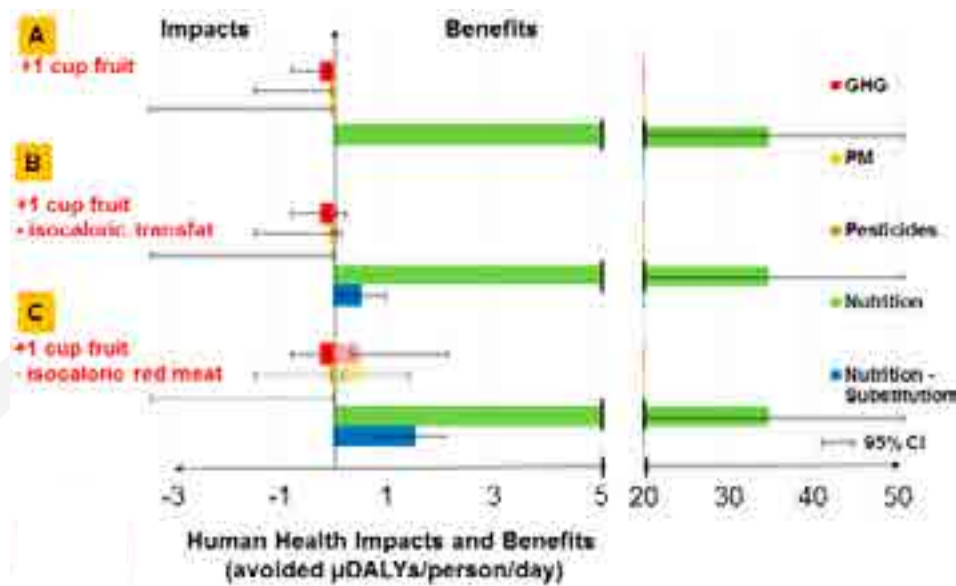


Figure 4. Comparison of daily environmental impacts and nutritional effects for a potential one serving fruit increase consumption, accounting for iso-caloric substitutions of trans-fat or red meat.

4. Discussion

In this paper, we use the CONE-LCA framework that enables a comparison between environmental and nutritional human health effects in a common end-point metric within a LCA context. In addition to the traditional environmental mid-point categories that are linked to human health impacts in LCA (GHG, PM), we also consider pesticide exposure in this case study since we are investigating consumption of fruits and vegetables. Although we limited our analysis to only three relevant environmental impact categories contributing to human health, it should be emphasized that the CONE-LCA framework can be extended to other human health-related environmental impact categories.

Specific to this case study, nutritional human health benefits associated with the addition of one serving of fruits or vegetables to the current European diet exceeded by far the corresponding environmental impacts in all three dietary scenarios. In scenarios B and C, where we considered potential substitution from trans-fat and red meat using an iso-caloric basis as a first proxy, the nutritional benefit was further reinforced due to avoided health impacts related to reductions of harmful food items. We acknowledge that such substitution choices come with limitations in terms of

scenario comparison and results interpretation. However, under the as assumption of an increase in healthy dietary choice consumption such as fruits and vegetables, an ideal substitution would occur from unhealthy food products such as trans-fat and red meat. We acknowledge that the trans-fat reduction as suggested in scenario B could be considerably hard to implement in practice. Although the content of trans-fat in food products has reduced and started to be labelled in food packaging (nutrition facts label), it still remains difficult to actually monitor and reduce daily intake due to the number of food items that contain trans-fat. Specific to our case study, the trans-fat substitution with fruits would require a reduction of 11.3 grams of trans-fat that could be achieved, for example, by removing 1.4 pieces of chocolate icing doughnut or 4 table spoons of margarine in stick form from the daily diet. To identify and assess realistic scenarios, substitutions should ideally build on detailed market-based and consumption-based surveys.

Finally, it should be mentioned that these are initial findings that depend on toxicological studies for the pesticide residue assessment and on epidemiological studies for the nutritional assessment. In addition, our findings are highly dependent on the quality and uncertainty of the data used. Hence, our findings should be interpreted within the context of this study and with caution. In the future our study aims to also consider epidemiological data that associate pesticide exposure to human health so that human health effects are assessed in a consistent manner with nutritional effects.

5. Conclusions

The present CONE-LCA framework enables us to compare in conjunction environmental impacts and nutritional effects on human health using a common end-point metric. The preliminary results of this case study indicate that nutritional health effects of food items, and specifically of fruits and vegetables, during the ‘use stage’ can be substantial and exceed by far any potential environmental impacts. In addition, our results emphasize the importance of affordability and accessibility to fruits and vegetables for the general public.

6. Acknowledgments

This work was based on an approach funded by an unrestricted grant of the Dairy Research Institute (DRI), part of Dairy Management Inc. (DMI).

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ORAL PAPERS

METHODS AND DATA 1

4. Methods and data (1)

234. Chapitre 8 ♦ Lca Comparison Results With Midpoint Method Associated With Fuzzy Logic Vs. Endpoint Method

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ABSTRACT

The Life Cycle Assessment (LCA) quantifies impacts, on the environment components, of the emitted or consumed substances. At a "midpoint" level, the end-user has to deal with results in many impact categories. A new multi-criteria assessment method provides a solution to aggregate, together, those results. This study's objective is to compare two methods giving a final indicator provided by the results of the LCIA method, in order to help end-users optimize the use of the calculation results. The LCIA studies have been done on ten grape production plots, with five contrasted vineyard management routes and on two contrasted years. The "ReCiPe" characterization methods were used to compare "endpoint" results, with "midpoint" results associated with an aggregative method: CONTRA, i.e. 'Transparent Construction of Decision tree' (Bockstaller, 2016) developed for viticulture. The "midpoint + CONTRA-QUALENVIC" method is a new multi-criteria decision analysis method (MCDA), using expert systems, associated to fuzzy logic. In the presented results, both methods have provided a similar ranking for 80% of the individuals. The presented results show that the "endpoint" method does not clearly differentiate the first five individuals. Both methods provide also intermediary criteria. The results obtained by the two methods are consistent, going in the same direction and provide good representation of the indicators results. However, "midpoint + CONTRA-QUALNVIC" method has the advantage of giving the possibility to go back to details of the "midpoint" impact origin provided by the LCIA.

Keywords: Life Cycle Impact Assessment (LCIA), Multi-criteria Decision Analysis, ReCiPe characterization factors, Aggregated results, Vineyard plots.

8.1 INTRODUCTION

The Life Cycle Assessment (LCA) tends to be the environmental assessment method most commonly used worldwide. The LCA makes possible the quantification of substances' impacts on the environment. Depending on the assessment objectives and the end-users,

different impacts can be studied. Cause and effect chains will determine the path of the substances emitted or consumed in the environment. The LCA quantifies potential effects on the environment for a quantity of substances emitted from an anthropogenic system ("midpoint" characterization) or the damage of those substances on the environment ("endpoint" characterization) (Bare et al. 2000; Payraudeau and van der Werf 2005).

The "midpoint" impact categories are often preferentially studied by end-users because results seem more relevant (Bare et al. 2000; Payraudeau and van der Werf 2005). This method allows characterizing directly the effect of a substance on the environment. The end-user chooses impact categories depending on the challenges his industry is facing. The end-user can choose global impact categories and/or local impact categories. The "midpoint" impact categories take into account most of the environmental compartments. The "endpoint" impact categories are less used by end-users, because they think that the method provides more uncertain results. The "endpoint" impact categories give up to three intermediate results to explain the environmental impacts caused by the studied activity.

At a "midpoint" level, it is difficult for the end-user to deal with the results of all impact categories considered in a study. The "midpoint" impact categories are too numerous, which makes difficult for him to analyze the result (good or bad result). Furthermore, with all these impact categories, the end-user doesn't know which ones really matters for him and where to start the improvement. LCA proposes some solutions to reduce the number of impact categories. However, by choosing a limited number of impact categories, the end-user will have only a partial view of the problems.

A new aggregating method named CONTRA (transparent construction of decision tree) (Bockstaller, 2016) is a multi-criteria decision analysis (MCDA) method using fuzzy logic. A version of CONTRA has been developed for viticulture as CONTRA-QUALENVIC (in the frame of the QUALENVIC Project) to be used, among other applications, in environmental evaluations of the technical management routes (TMR). Use of "midpoint" results associated with "CONTRA-QUALENVIC" method (development of CONTRA for viticulture) allows an end-user to have an overview of the impact categories results. The end-user can also get a result based on methods which characterize the effects of all substances emitted by processes on different compartments of the environment. By combining CONTRA method with "midpoint" results, the end-user will not lose the main information provided by the "midpoint" method and will ensure robustness of its impact categories results with accurate and reliable results.

The present work's objective is to study the interest of associating a joint assessment method using fuzzy logic with LCA midpoint categories in comparison with an "endpoint" LCA method, in order to help the end-user optimize the results' use. The first method uses "CONTRA-QUALENVIC" to aggregate "midpoint" LCA results. The second method is the "endpoint" characterization that gives one to three impact results on human health, ecosystems and resources. Both methods were compared by being applied on real cases of grape production systems. These new multi-criteria assessment methods can be a solution to aggregate, together, the environmental impact categories results.

The final results using each method, must give the best compromise possible, between (i) a good overview of the impact categories results and (ii) the possibility to go back to details of the impact origin.

8.2 METHODS

To analyze the two methods, the results of LCAs have been studied for ten individuals. An individual corresponds to the study of one plot during one production year. The considered cases are five plots of Chenin Blanc grape dedicated to produce dry white wine with a Protected Designation of Origin (PDO) area, located in middle Loire Valley, France and studied during two contrasted climatic years. These plots were chosen for their contrasted vineyard technical management routes (TMR) (Renaud-Gentié et al., 2014). The first studied year (2011) is considered as a favorable year for grape production (favorable climate and low pest pressure) and the second studied year (2013) is considered as an unfavorable year (Renaud-Gentié, Renaud et al. 2014). During 2011, the winegrowers have limited their pesticides sprayings and their mechanical interventions on the plot, whereas, during 2013, they have used a lot of pesticides sprayings and mechanical interventions on the plot.

The functional unit for the LCA of those individuals was the same in both assessment methods: 1-hectare area cultivated during one year of grape production. The system boundaries for all studied plots, took into account all operations' impacts (direct and indirect) that occurred on the grape production on the studied plots. All practices, conducted on grapes outside of the plot, are not considered in this study. The life cycle studies considers four main phases of grape production: Three non-productive phases, amortized on 30 years: (i) land preparation and planting of vines; (ii) first three years of life of the vine; (iii) destruction phase of the vine which is the end of studied plots life; and one productive phase: the annual grape production phase.

The LCA has used the SimaPro software (V8.0.5.13) with the EcoInvent (V3) as main database. Inventory data were collected from different sources. First, all practices on the plots, during all phases of the studied case's Life Cycle, were collected by interviews of the winegrowers. For each practice, many information were collected (i.e. tool and tractor names, practices' duration and characteristics names). Secondary data on the tools and the tractor were collected from literature (i.e. weights, lifetime, consumption) (Rouault et al. 2016). The EcoInvent database was used to complete the Life Cycle Inventory. Some direct field emissions were calculated: the emissions of nitrogen through the study of ammonia (NH₃) with Tier2 approach (Hutchings et al., 2013), nitrate (NO₃⁻) with SQCB (Faist-Emmenegger et al., 2009), nitrogen dioxide (N₂O) and nitrogen oxides (NO_x) with Ecoinvent (Nemecek and Schnetzer, 2011). Others emissions were also calculated, for example phosphorus with SALCA-P method (Nemecek et al., 2007), heavy metals with SALCA-ETM (Freiermuth, 2006), pesticides with PestLCI 2.0 (Renaud-Gentié et al., 2015 ; Birkved and Hauschild, 2006) and fuel consumption (Renaud et al. 2011; Rouault et al. 2016).

The ReCiPe characterization method (Hierarchist/ V1.12 / Europe ReCiPe H/A) was used for all Life Cycle Impact Assessments, "midpoint" and "endpoint". This way, we ensured to have the same assessment calculation approach, for each emitted substance in the environment and for each consumed substance from natural resources.

The "CONTRA-QUALENVIC" method was applied on a selection of "midpoint" results, firstly, to aggregate those results into four intermediary results and, then, to get one final score

(Figure 1). The “CONTRA-QUALENVIC” method followed the following steps: First, we had defined a decision tree that determined the aggregation criteria; then, the weightings of the different criteria were established; finally, threshold values (upper and lower) were defined for each criterion. These threshold values correspond to the values of the impact category or component, below or above which, the score is unfavorable (scores such as 0/10) or favorable (scores such as 10/10, meaning a low environmental impact). The decision tree and the weightings of criteria were established based on a consensus among experts. In the absence of other references at the moment, the threshold values were derived from minimum and maximum values, of the ten LCA performed for this study, for each chosen impact categories. At the aggregated level, threshold values on aggregated criteria are between 0 and 10, to obtain a final score between 0 and 10.

The “midpoint” and “endpoint” ReCiPe methods use different characterization factors (CFs) for one identical substance emitted in the environment.

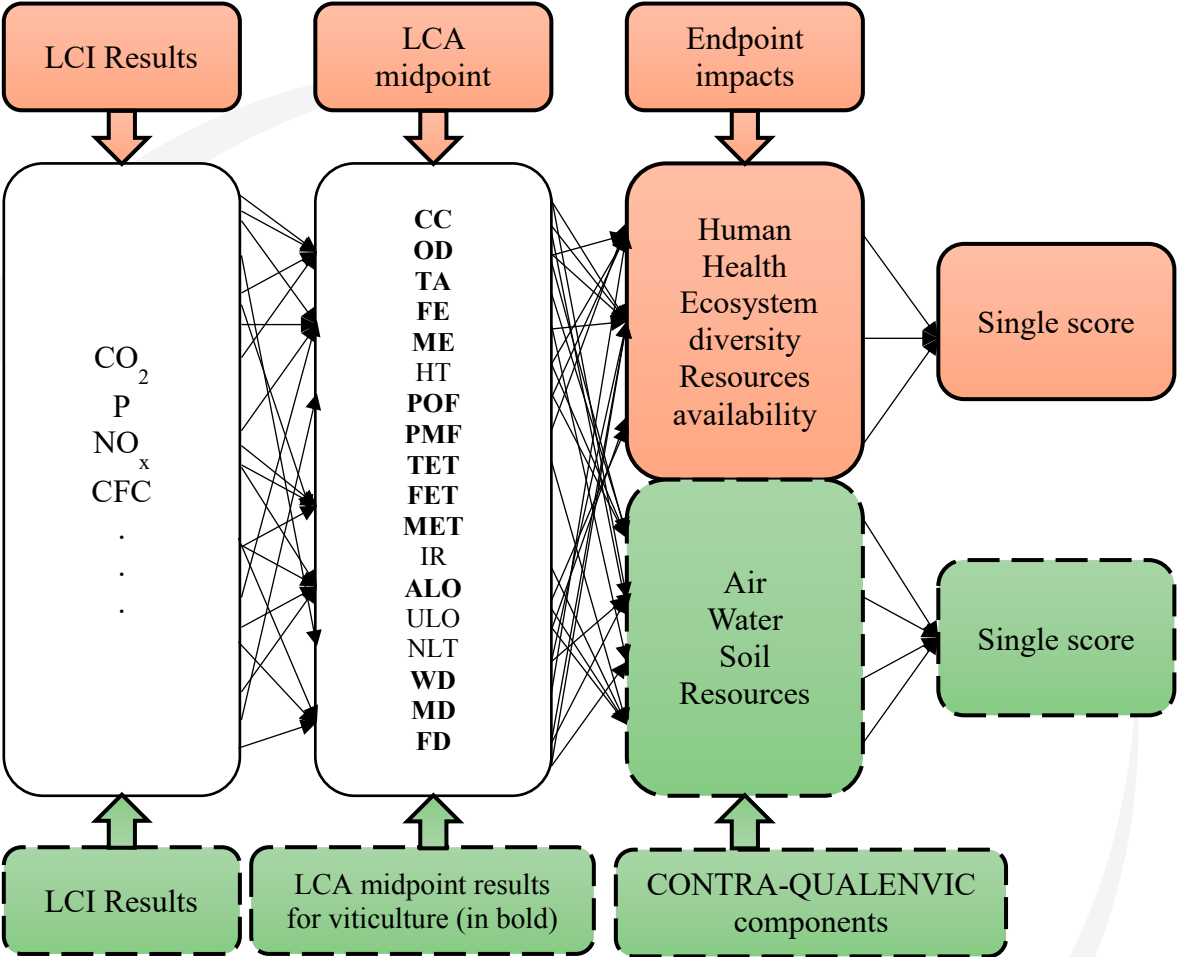


Figure 1: Relations between the LCI parameters, midpoint impact categories, endpoint categories, CONTRA categories, endpoint single score and CONTRA single score. (CC: Climate change, OD: Ozone depletion, TA: Terrestrial acidification, FE: Freshwater eutrophication, ME: Marine eutrophication, HT: Human toxicity, POF: Photochemical

oxidant formation, PMF: Particulate matter formation, TET: Terrestrial ecotoxicity, FET: Freshwater ecotoxicity, MET: Marine ecotoxicity, IR: Ionising radiation, ALO: Agricultural land occupation, ULO: Urban land occupation, NLT: Natural land transformation, WD: Water depletion, MD: Metal depletion, FD: Fossil depletion)

The ten individuals were analyzed through both of the two tested methods (“midpoint LCIA+CONTRA-QUALENVIC” vs. endpoint LCIA). The “midpoint LCIA+CONTRA-QUALENVIC” method focuses on effects and the Endpoint method focuses on damages. But both approaches are based on LCIA results.

Results’ analysis will evaluate each studied individual through each tested method. For each tested method, individuals are classified and the differences between the scores of individuals are measured.

8.3 RESULTS

The two tested methods provide a “single score” approach. This approach can help the end-user by simplifying the impacts analysis due to the reduced quantity of indicators (from one to four). But both tested methods provide different characterization factors, which have different units, and provide different information: The “midpoint” method gives independent results with numerous impact categories; the combining “midpoint” method to the “CONTRA-QUALENVIC” method gives, also, aggregated results. In comparison, the “endpoint” method focuses on the environmental damages characterization, which is more complicated for the end-user to work on, to improve their environmental performance.

The results of the present study (*Table 1*) show that “midpoint + CONTRA-QUALENVIC” is a new way of interpreting the LCA results. Although the two tested methods are different, the comparison is still possible and provides new information.

In the presented results sheet, both methods provided a similar ranking for 80% of the individuals (*Table 1*).

Table 1: Summary table of the comparative analysis of results of the two methods evaluated.

Individuals	Final notations		Ranking on final notations		Gap Notation between two individuals on final notations		Intermediary notations for CONTRA method using midpoint results (notations between 0 and 10)				Intermediary notations for Endpoint results (unit: Points)		
	CONTRA method (notations between 0 and 10)	Endpoint method (unit : Points)	CONTR A method	Endpoint method	CONTRA method (result between 0 and 1)	Endpoint method (result between 0 and 1)	Air	Water	Soil	Resources	Human Health	Ecosystems	Resources
11TMR3	9.9	467	1	1	1.0	1.0	10.0	9.9	10.0	9.7	65	371	30
13TMR3	8.4	503	2	3	0.8	0.9	8.1	7.2	9.4	8.0	81	379	42
11TMR4	6.9	508	3	5	0.7	0.8	6.6	9.1	6.3	5.6	64	393	50
11TMR1	5.7	474	4	2	0.6	1.0	8.2	4.9	8.1	3.7	59	366	49
13TMR1	3.7	504	5	4	0.4	0.9	4.8	1.5	6.3	7.3	73	388	42
11TMR2	3.7	572	6	6	0.4	0.6	2.7	9.1	5.2	3.1	73	444	54
11TMR5	3.2	585	7	7	0.3	0.5	2.4	9.4	3.5	3.7	86	435	63
13TMR2	2.1	602	8	8	0.2	0.5	1.6	4.8	5.7	1.9	86	451	64
13TMR5	0.5	635	9	9	0.0	0.4	0.1	0.9	2.3	1.3	106	448	80
13TMR4	0.1	727	10	10	0.0	0.0	0.4	0.5	0.2	0.3	212	431	84

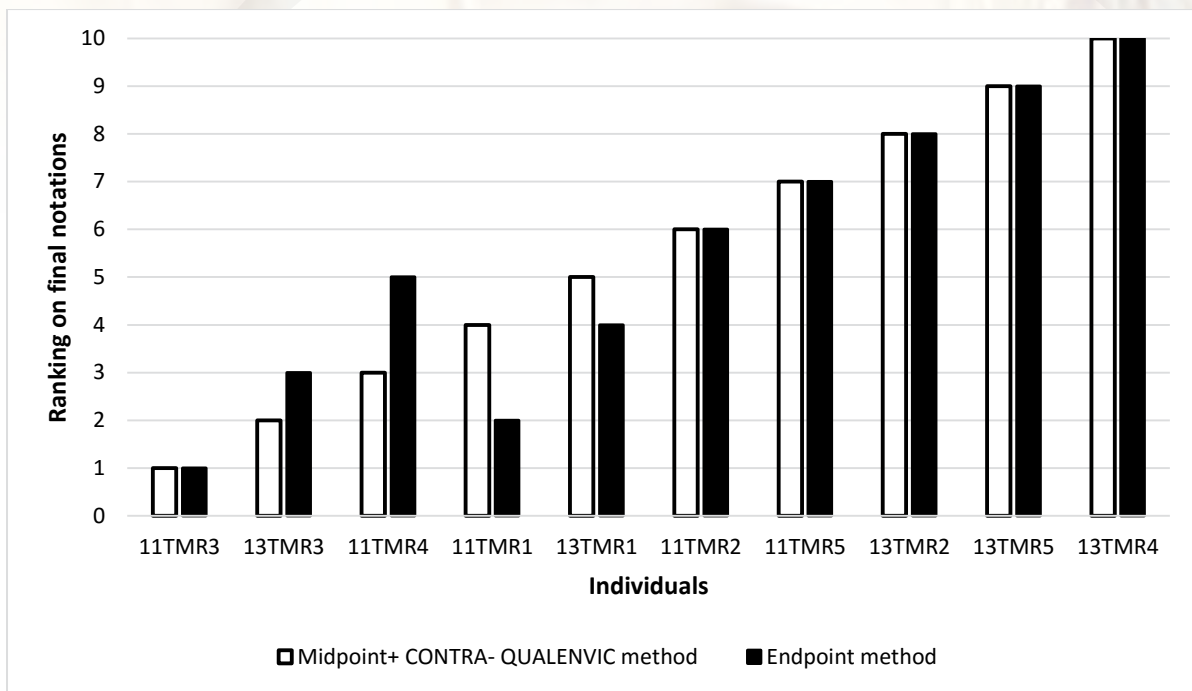


Figure 2: Ranking on final notations of the five studied plots over the two years studied.

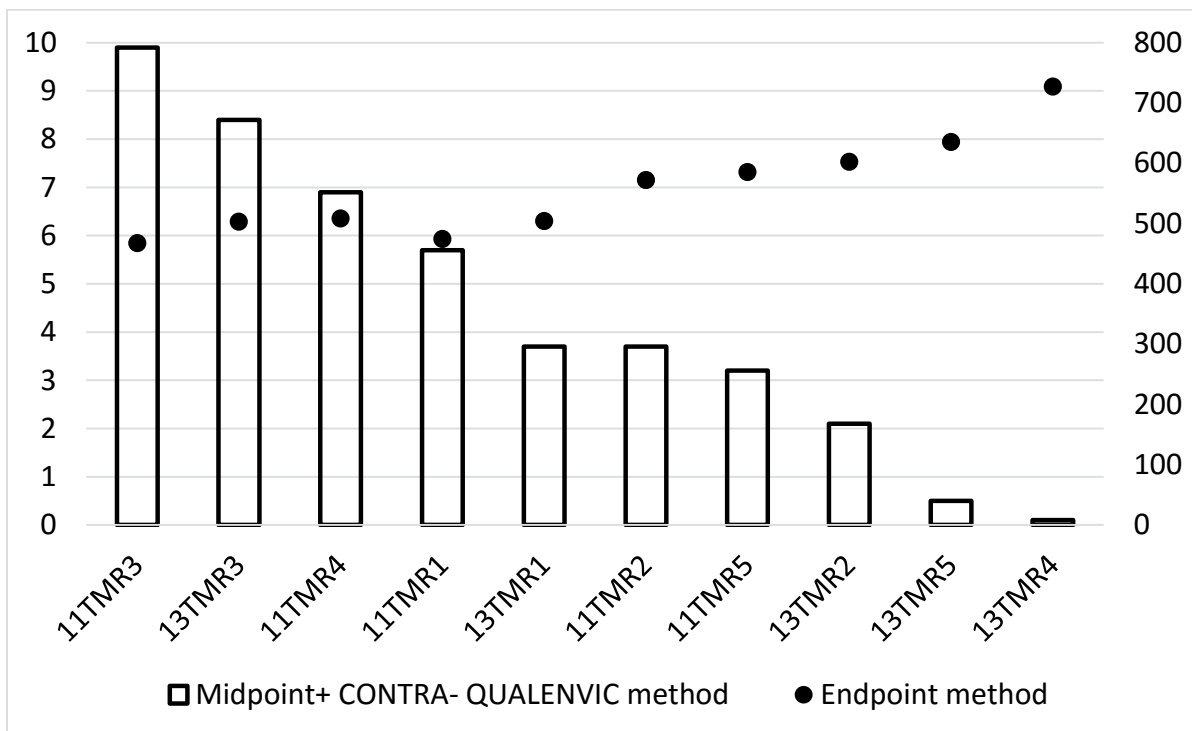


Figure 3: Final notations for both studied methods (midpoint + CONTRA-QUALENVIC and endpoint) the five studied plots over the two years studied.

Only two individuals don't obtain the same ranking in the two methods (figure 2): (i) 2011TMR1 (Technical Management Route n°1 pour the 2011 studied year) is classified 2nd with the "endpoint" method and 4th with the "midpoint + CONTRA-QUALENVIC" method; (ii) 11TMR4 is classified 5th with the "endpoint" method and 3rd with the "midpoint +

CONTRA-QUALENVIC” method. The presented results show that the "endpoint" method does not clearly differentiate the first five individuals (*Table 1*, column ‘Gap Notation between two individuals on final notations’).

Figure 3 illustrates the fact that both methods are not expressed with the same objective. The “endpoint” method (or the "midpoint" method, alone) is developed to focus on environmental impacts while the end-user measures the environmental performance of his activity. The “midpoint + CONTRA-QUALENVIC” method reunites the two approaches presents with a different logic of analysis; The "midpoint" impact results are transformed into aggregated performance results, and by doing so, the evaluation is established on the global environmental performance of an activity.

Both methods provide also intermediary criteria, but the comparison between the two methods is not easy at this point. This is due to the fact that i) the intermediary criteria are different, ii) the calculation methods are different and iii) the “midpoint + CONTRA-QUALENVIC” method evaluates the effects of emitted substances on the environment while endpoint method evaluates their damages.

8.4 DISCUSSION

Both methods can be compared on three criteria: transparency, uncertainty and usefulness for the decision-maker.

The LCA requests an evaluation method as transparent as possible (Bare et al. 2000; Payraudeau and van der Werf 2005), to help end-users. The “midpoint” method is usually considered more transparent than the “endpoint” method (Bare et al. 2000). If we consider that the “CONTRA-QUALENVIC” method is also a transparent method (the user can see all aggregating decision rules) it could be possible to consider that the “midpoint + CONTRA-QUALENVIC” association method is more transparent than the “endpoint” method.

The “endpoint” method is based on characterization factors which are applied on the “midpoint” results. The uncertainty of the models depends on the used characterization method and on its model parameters (Payraudeau and van der Werf 2005). It reveals that the “midpoint” method has less uncertainty than the “endpoint” method (Bare et al. 2000). The Research Group of the European Commission (Hauschild et al. 2013) and the ILCD (International Reference Life Cycle Data System) (Sala et al. 2011) have strongly recommended the use of "midpoint" categories, even if the endpoint characterization factors have been improved. However, the CONTRA method’s level of uncertainty has not been measured yet, except for few comparisons done with other aggregated methods, such as Choquet integral or DEXi (Botreau et al., in prep.). In the CONTRA-QUALENVIC method, where the practitioner decides on the construction of the decision tree, the weightings and the threshold values

The end-user can enjoy having a final score with both tested methods. But, with the “endpoint” results, it is more difficult to communicate on endpoint impact categories (because the units are difficult to interpret, like DALY (Bare et al. 2000). The results, given by the midpoint method, seem easier to understand (eg: kg eq. CO₂). Also, direct environmental impact is preferred to the effect indicators (Payraudeau and van der Werf 2005). The

aggregated criteria, proposed by the “midpoint + CONTRA-QUALENVIC” method, are the compartments of the environment (eg: water, soil), they are unambiguous for the end-user who can prioritize the environmental systems on which he wants to act first. Using final results makes it easier for the end-user to compare an LCA to another. This gives a first view of all the results. The “midpoint + CONTRA-QUALENVIC” method has been presented to end-users in the context of than the QUALENVIC national project. Their first feedbacks were enthusiastic and they were looking forward to using this method.

One additional comparison point between the "midpoint + CONTRA-QUALENVIC" method and the “endpoint” method can be highlighted, which is the way to consider the evaluation. The end-user will prefer to talk in terms of performance rather than talking of impacts, this can be more rewarding for him. And the objective will be to improve these performances.

Our results show that the "midpoint + CONTRA-QUALENVIC" method allows to directly interpret LCA results, by studying the effects of consumed or emitted substances on the environment. The information delivered by the method "CONTRA-QUALENVIC", based on the aggregation of categories "midpoint", allows the end-user to quickly and comprehensively analyze the study's main result. The end-user can also easily compare the study's results with the results provided by another study, such as the comparison of production systems. Moreover, the end-user can orientate his analysis on the environmental compartment (air, water, soil or resources) that he considers as his biggest challenge at the moment.

The rankings analysis shows that the results offered by "CONTRA-QUALENVIC" method are consistent with the use of a 'single score' via LCIA "midpoint" characterization method.

The differences in ratings with the “midpoint + CONTRA-QUALENVIC” method, shows some robustness, which suggests that the differences may not be caused by “a potential error”. The differences between the marks are relevant. The methodology used for the construction of aggregations was tested by Botreau et al. (in prep.) and he has found that the “CONTRA-QUALENVIC” method give better results compared to others tested methods.

Setting the “CONTRA-QUALENVIC” method can take some time. The determination of the decision-tree, the weighting and the threshold values must be the result of a consensus between expert panels. Meetings to determine those parameters can take few days. But once it is done, the evaluation can be quickly conducted.

However, using only ten individuals, we cannot validate fully the "midpoint + CONTRA-QUALENVIC" method.

Whatever the method (« endpoint » or «midpoint + CONTRA-QUALENVIC”), the results discriminate well the individuals, according to the plots, the practices and the typical years, which is rejoicing. More results on others plots and on more typical years would be needed to consolidate these first reassuring results.

8.5 CONCLUSIONS

The combination of "CONTRA-QUALENVIC" method with "LCA midpoint" method provide relevant, consistent and robust results for the LCA results analysis, by the end-users.

Comparing method "midpoint + CONTRA-QUALENVIC" with "endpoint" LCA method, by using the concept of the single score value, highlights the pertinence of using the CONTRA method in the LCA methods with midpoint categories. The results obtained by the two tested methods are consistent. This "midpoint + CONTRA-QUALENVIC" method ensures that we will evaluate the effect of substances and that we will get a single score at the end of the evaluation. Moreover, this is a transparent method that gives intermediate results. This method makes easier the comparison between LCA "midpoint" results, for different individuals, by using one single score note. This score can then be compared to other types of assessments results. More results on others individuals would be needed to consolidate these first encouraging results

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249. LCA and Risk Assessment of Recycled Phosphorous Fertilisers

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ABSTRACT

An efficient phosphorus (P) recycling from urban areas is becoming an increasing issue due to the scarcity of natural P deposits. In order to assess the environmental performance of different approaches of P recycling, a LCA assessment and risk assessment studies were carried out. Generally, we found the supply of recycled P-fertilisers (RPFs) to be competitive as compared to conventional sources in terms of LCA results analysed in this contribution: per kg P and with exclusion of infrastructure processes, the highest abiotic depletion potential is shown for conventional mineral fertilisers based on phosphate rocks due to the finite mineral P resources. For fossil primary energy demand, the recycled fertilisers (struvites and an ash-based fertiliser analysed) had highest impacts per kg P. Relatively high acidification and eutrophication potentials from the supply of P-fertilisers are related to composts, triple-superphosphate and struvites. For the global warming potential per kg P, compost presents the worst results again. However, if co-products of organic fertilisers are considered (i.e. N- and K-contents and the humus sequestration potential), most organic fertilisers are in advantage for a number of indicators – with the exception of conventional composting. The low emission compost and the stabilised sewage sludge present moderate to good overall results. The best relative results for all indicators were found for biogas digestate.

To assess the risk of soil contamination related to the long-term application of RPFs, accumulation scenarios in soil were calculated with a mass balance approach for the potentially toxic elements (PTEs) Cd, Cr, Cu, Ni, Pb, and Zn and for the persistent organic pollutants (POPs) polychlorinated biphenyls (PCB), polycyclic aromatic hydrocarbons (PAH) and polychlorinated dibenzodioxins and dibenzofurans (PCDD/F) in composts, digestates and other RPFs derived from sewage sludge. For all calculations, a fertiliser application over 200 years equivalent to 11 kg P ha⁻¹ yr⁻¹ was assumed. Dependent on PTE mobility in soil due to pH and precipitation excess F, an accumulation or depletion compared to the soil background values was found. Highest accumulation was found in scenario pH 7 F 0,1 m yr⁻¹, lowest in scenario pH 5 F 0,3 m yr⁻¹. Fertilisers like composts, with low P content compared to PTE load, had a higher accumulation potential than fertilisers like struvite, meat and bone meal, sewage sludge ash, sewage sludge and digestates, rock phosphate and triple super-phosphate. Only Cd accumulation with TSP was higher than that with compost. For POPs no accumulation risk in soil was found.

Keywords: life cycle assessment, . potential toxic elements (PTEs), contaminants, recycled P fertilisers

1. Introduction

Phosphorous (P), an essential element in plant nutrition and agricultural management is a globally limited resource. Due to a permanent export of nutrients with biomass harvests and because of the fact that phosphorous (P) is not fixed from the air such as nitrogen and must be imported as a bought-in fertiliser, soils of organic and conventional farms are in danger to loose P soil contents. Negative P balances were found for eight of the 27 EU countries for the average agricultural area for the period 2005 to 2008 (on average -4.8 kg P per hectare; Eurostat 2015). The average for all agriculturally utilised land in the EU-27 shows a positive balance of 1.8 kg P per hectare. Thus and due to the fact that P from fossil deposits is more and more depleted, it is important to close farm's P-cycles and the societal cycle by recycling residential organic wastes, e.g. organic household wastes or (stabilised) sewage sludge and its products. The focus of this contribution is the analysis of important

environmental impacts of the production, supply and fertilisation of selected recycled P fertilisers (RPFs) by a life cycle assessment (LCA) and the accumulation potential of potentially toxic elements (PTEs, often referred as heavy metals) and three persistent organic pollutants in soil after long-term fertiliser application.

2. Methods

We analysed and compared a number of recycled P-fertilisers: Firstly organic fertilisers produced from urban organic wastes, which are composts (with different composting techniques and substrates) and biogas digestates¹, as well as stabilised sewage sludge. Secondly a recycled fertiliser based on sewage sludge (SS) ash, two different struvites (with different technologies; from SS) and meat and bone meal. Thirdly, the conventional and commonly used references rock phosphate (assumed to be imported from Morocco for the LCA) and triple-superphosphate (TSP; based on phosphate rocks mined in Morocco and partly produced in Morocco as well as the European Union for the LCA).

LCA – data, impact categories, system boundaries and sensitivity analysis: LCA-relevant data on P concentration in the different substrates (e.g. household wastes, sewage) and their dry matter content were taken from Möller & Schultheiss (2014) and Möller (2015). For the demand on inputs for the production process, i.e. energy (carriers) and chemicals, data from studies and databases were used including measured data from pilot plants, e.g. Remy & Jossa (2015) and Ecoinvent (2014). Emissions from the production of fertilisers, for example during composting were taken from Ecoinvent (2014) or calculated from Pardo et al. (2014). For emissions from biogas plants, we assumed a leakage of 2% of CH₄ but no losses of N₂O and used IPCC (2006) guidelines for calculation. Credits for substituted energy from biogas plants were derived with the German energy mix according to Ecoinvent (2014) data.

Five LCA impact categories were calculated for each of the P-fertilisers and referred to the functional unit of 1 kg P: abiotic resources depletion potential (ADP; Sb-eq), fossil primary energy demand (FED; MJ), global warming potential (GWP, CO₂-eq), acidification potential (AP; SO₂-eq) and the eutrophication potential (EP; PO₄³⁻).

ADP considers the use of resources from the lithosphere and also includes fossil energy carriers. The latter are also included in the FED. The EP covers potential losses of phosphate (PO₄) and nitrogen (as NO₃, NH₃, NO_x) during the production processes and distribution of fertilisers with their potential negative impacts on aquatic and terrestrial ecosystems. The AP analyses acidifying substances (SO₂, NH₃ and NO_x) during the production and transportation of fertilisers and their inputs. The GWP reflects emissions relevant for the climate change: methane (CH₄), nitrous monoxide (N₂O) und carbon dioxide (CO₂).

All recycled P-sources are evaluated in the software SimaPro (v 7.3; PRé Consultants) along their whole life cycle until application and they are compared with each other as well as with the conventional references of mineral P fertilisers. For characterisation, the method "CML" (Guinée et al., 2002; with updated characterisation factors) was selected.

¹ Fertilisers that may be used in organic farming are designated with „(OF)“

System boundaries were chosen to include the phases of supply and production of RPFs. This includes acquisition of inputs such as raw materials and (co-) products from previous processes (e.g. materials for chemical processes), the demand on energy carriers, etc. Infrastructural processes i.e. capital goods such as buildings and roads for transports were excluded. The collection of substrates (e.g. organic wastes from households) was considered for the recycled P-fertilisers where it is specifically needed for the production processes. In cases where the collection of substrates is independent from production and has to be done in any case (e.g. the transportation of urban household wastes to a collection site), this first transportation was not considered for the LCA of the APFs. For these cases, the effort for the disposal of wastes was attributed to the end of life stage of the previous main product, i.e. to its main use phase; the disposal of biogenic household waste, for instance, is therefore attributed to food production.

Short as well as long process chains were found for the APPs' calculation: (1) for stabilised SS, for instance, the process chain covered by the LCA contains only the two processes of transports to a farm and field application. Emissions from SS until transports to a farm were allocated to the disposal phase (especially for the case of anaerobic treatment, where the potential net-energy gain from the process is also accounted for the disposal phase of SS and not to the fertiliser). (2) The example of the RPF biogas digestate covers more processes. The system boundary starts with a treatment of the substrate and includes the demand on auxiliary material or energy. On the contrary, the process of digestion leads to biogas which is used as fuel (to produce electricity and heat); consequently energy is produced that replaces other energy and a credit is accounted for this substitution. As above for the example SS, the household wastes' collection (transport) before their digestion is not accounted for, but allocated to the food waste disposal (food's end of life). (Modern) biogas plants are built as airtight systems, hence, losses of CH₄ and N-containing gases (NH₃, NO_x, N₂O) are assumed to be low. However, a small proportion of gases passes off and is accounted for within the process of the biogas plant. System boundaries end with the inclusion of energy demand for the process of fertilising: The application (spreading) of RPFs onto fields is also accounted for within the system boundaries for all P-fertilisers. Further emissions from fertilising or from the fertilised soil are not accounted for, to obtain comparability due to varying emissions, which highly depend on used techniques or processes for P recovery, the content and availability of (reactive) nitrogen and carbon in the fertilisers, etc. However, for the organically-based RPFs which contain other nutrients such as nitrogen (N) and carbon (C), an additional calculation addressed the credits for a humus sequestration potential and for substituted N- and K-fertilisers, the latter according to Ecoinvent (2014). The credit for the humus sequestration potential was calculated based on the assumption that a long-term organic fertilisation (50 years) increases the content of humus at most by 10% of the initial value (for compost) until humus saturation. Credits were specifically determined for organic fertilisers considering their respective P- and C_{org}-contents. Due to different types (liquid or solid) and varying application machinery as well as different concentrations of P in a certain fertilisers (for dry matter) and due to the RPFs specific water contents, impacts of transports and fertilising vary among the RPFs. For a useful comparison of the different RPFs, comparable transport distances were used. For an industrial production of a P-fertiliser (this is the case for recycled fertilisers, for meat and bone meal, rock phosphate and TSP), in total a 500 km transport by a lorry (16-32 t; fleet average concerning loading) was assumed from the production to a regional storehouse. Furthermore, a 20 km transport from the storehouse to the farm by tractor and trailer was assumed for these industrial P-fertilisers. For regionally produced RPFs (composts, biogas slurry or untreated SS), in total 50 km with tractor and a trailer (or a tanker for liquids) were assumed. For transports of auxiliary inputs (which are also considered within the category "transport"), e.g. chemicals for recycled RPFs, each 100 km of lorry and freight train were assumed.

Sensitivity analyses were done for the highly effective impacts of C- and N-containing biogenic emissions from the composting processes. We compared data from Ecoinvent (2014) to data calculated for composted food wastes based on Pardo et al. (2014).

Risk Assessment: Four accumulation scenarios in soil were calculated over a 200-years period for the potential toxic elements Cd, Cr, Cu, Ni, Pb, and Zn in a mass balance approach for the selected fertilisers described above. For all calculations a fertiliser application rate equivalent to $11 \text{ kg P ha}^{-1} \text{ yr}^{-1}$ was assumed to counterbalance the average annual P loss on organic arable farms without livestock (own calculations based on a literature review). The mass balance model for PTEs was according to Smolders (2013) and for POPs according to Amlinger (2004). Four scenarios were calculated for PTEs with pH 5 and 7 and a precipitation excess (F) of 0.1 and 0.3 m yr^{-1} . The iterative model included the input sources fertiliser, atmospheric deposition, liming, output via leaching and crop offtake as well as the soil background concentration (Table 1). PTE mobility in soil is different for each element and depends on many parameters. The main PTE output flux is leaching, it depends on soil pH, precipitation excess and PTE concentration in soil. Leaching is the outflow of PTEs dissolved in pore water from the topsoil and is a function of water percolation through soil expressed as precipitation excess F in m yr^{-1} . The dissolved PTE in pore water changes with total PTE concentration in soil and reflects the element behaviour in soil matrix. To predict the PTE concentration in pore water, a distribution coefficient (K_D in L kg^{-1}) was used. K_D is a measurement for the relative partitioning of an element between the solid and the solution phase. To estimate crop offtake, transfer factors for wheat grain and the average European wheat yield (5 t ha^{-1}) was used. Scenario results for PTEs in soil were compared with threshold values proposed by Gawlik and Bidoglio (2006). The iterative mass balance model for POP accumulation in soil considered the input fertilizer and atmospheric deposition, the soil background concentration and the half-life time of compound in soil (Table 1).

Table 1: Average soil background concentrations in European agricultural topsoil, atmospheric deposition, lime input and proposed threshold values for PTEs and half-life times for POPs.

		Cd	Cr	Cu	Ni	Pb	Zn
Soil background concentration	mg kg ⁻¹	0.28	94.8	17.3	37.3	32.6	68.1
Atmospheric deposition	g ha ⁻¹ yr ⁻¹	0.36	9.30	34.0	10.0	11.9	227
Input with lime	g ha ⁻¹ yr ⁻¹	0.14	0.92	1.38	0.78	0.94	2.00
Proposed threshold values in soil	mg kg ⁻¹	1.00	75.0	50.0	50.0	70.0	150
		PCB	PAH	PCDD/F			
Soil background concentration	mg kg ⁻¹	0.01	0.43	0.000006			
Atmospheric deposition	g ha ⁻¹ yr ⁻¹	0.35	2.67	0.000002			
Half-life time in soil	years (mean)	14.2	12.2	29.7			
Proposed threshold values in soil	mg kg ⁻¹	0.30	1.00	0.01			

PCB: polychlorinated biphenyls, PAH: polycyclic aromatic hydrocarbons, PCDD/F: polychlorinated dibenzodioxins and dibenzofurans

3. Results and Discussions

Detailed results on fossil energy demand and global warming potential: Compost-P results in medium results concerning FED, whereas the two struvites, biogas slurry (due to its transport) and meat and bone meal show much higher demand (Figure 3). Although the same transport distance (50 km by tractor) was assumed for all regionally produced RPFs, digestates shows a substantially higher impact of transportation than other RPFs. This is due to the higher water content of digestates, i.e. that a higher amount of fertiliser (water) has to be transported. However, a part of biogas' FED is substituted by the net-energy produced within the biogas process. Stabilised and regionally transported liquid SS, rock phosphate, the ash-based RPF and TSP present low to medium FEDs.

Results for the GWP do not consider infrastructure's impacts but the co-products for the organic fertilisers (humus sequestration, N- or K-fertiliser effect). Highest GWP was found for composts as a consequence of N- and C-losses during composting (Figure 3). For the sensitivity analysis with data from Pardo et al. (2014) vs. Ecoinvent (2014), we found a nearly equal impact of the biogenic emissions on the GWP and hence no effect on varying results. However, the performance of organic fertilisers improves substantially with consideration of the substitution potential of co-products and even results in negative net emissions per kg P – except for standard composting. Especially due to the co-products' effect, the best net emission result was found for digestates. Furthermore, the effect of potential CH₄ leakage from the biogas process is almost offset by substituted emissions through the net energy gain. Contrarily, liquid digestates show a comparably high impact of transportation on the GWP due to the significantly lower P-content (i.e. the higher water content). Regionally transported (liquid) stabilised SS show a relatively low GWP. Reasons for the low environmental impacts of SS

are the comparably high P content in the sludge and the assumed relatively low transport distances for a regional application and mainly the substitution potential of co-products. The ultimate substitution potential depends upon which N fertiliser is replaced with the organic (P-) fertiliser, the long term nitrogen use efficiency of the N in the organic fertiliser and how successfully – depending on the humus saturation of the soil $-C_{org}$ is sequestered in a long-term. As compared to organic RPFs such as digestates and especially composts, struvites or mineral P-fertilisers do not (or hardly) contain C, N or K anymore. Consequently, the organic fertilisers (with the exception of standard composting) are favourable in terms of most LCA results when they are related to their overall fertilising value.

Comparably low GWPs were also found for the conventional references rock phosphate and TSP. The latter has a slightly higher environmental impact due to the beneficiation processes for the raw material phosphate rocks. The analysed RPF based on SS is in advantage as compared to the struvites due to an efficient use of the by-product heat from mono-incineration of SS (for the drying of the fertiliser product). The GWP of struvites (especially for #2) is comparable to net emissions of the low emission compost. The two struvites based on SS need relatively high amounts of chemicals and energy embodied in these auxiliary materials (including their transport activities) as well as high energy demands for the precipitation process itself. Thus they show medium GWP results. The GWP for meat and bone meal is dominated by the fossil energy demand, which is mainly a consequence of energy needed for drying, extracting and especially for pressure sterilisation of the livestock material.

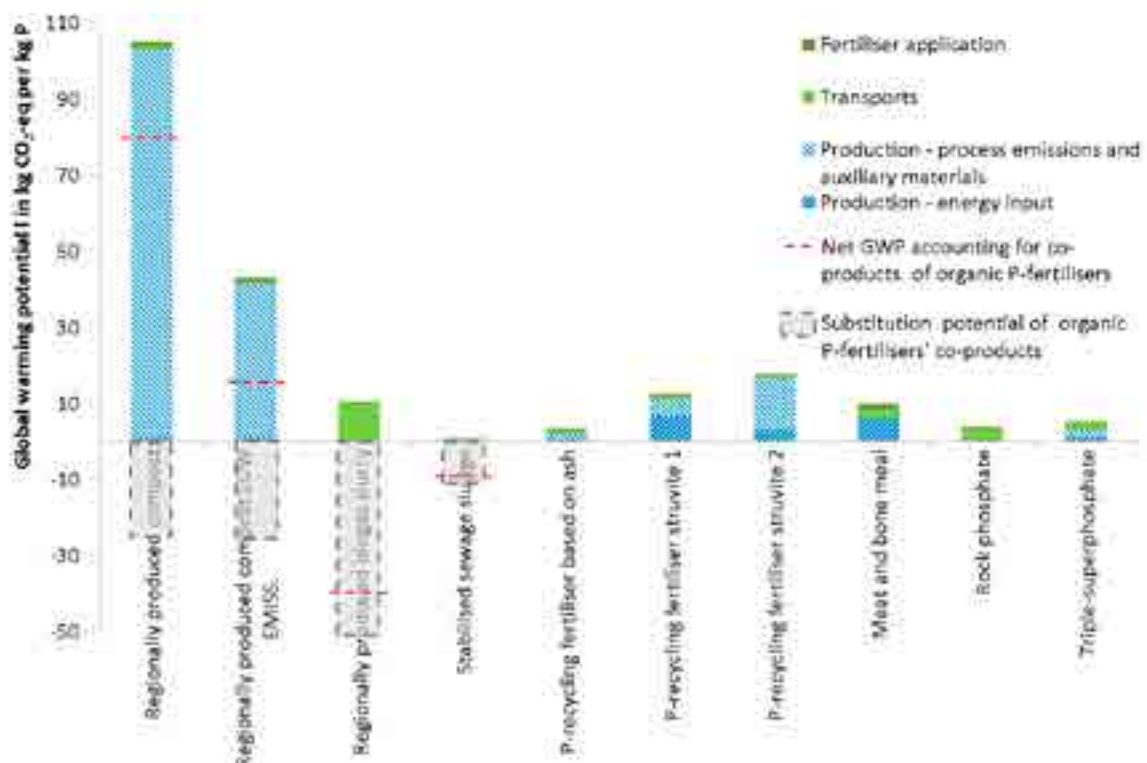


Figure 3. Global warming potential (GWP, greenhouse gas potential) in kg CO₂-eq per kg of P without infrastructure emissions but with co-products from the organic fertilisers (humus sequestration, N- or K-fertilising) taken into account.

P in ash-based RPFs as well as in rock phosphate is hardly available for plants, especially in alkaline soils. Thus, a good LCA result can be paired with a trade-off concerning the quality of the RPF and vice versa. The LCA results represent one important contribution to analyse strengths and weaknesses of P-fertilisers and recycling pathways but they have to be accompanied by other indicators such as plant availability, the effectively recyclable proportion of limited nutrients and the risk assessment on accumulation of contaminants (see chapter after the next).

Relative advantages and disadvantages of specific fertilisers for all five LCA-indicators:

Figure 4 presents an overview of all indicators analysed in this study, i.e. ADP, EP and AP in addition to the FED and the GWP as relative advantage between the best/lowest result per indicator (0) and the worst/highest result per indicator (1). A high column indicates high environmental impacts. Whereas fertilisers based on phosphate rock have a high ADP, struvite partially a high FED and EP and compost very high AP, EP and GWP, the RPFs based on SS show lower values in all categories. The low emission compost and the stabilised SS present moderate to good overall results. The best relative results for all indicators were found for biogas digestate (see Fig. 4).

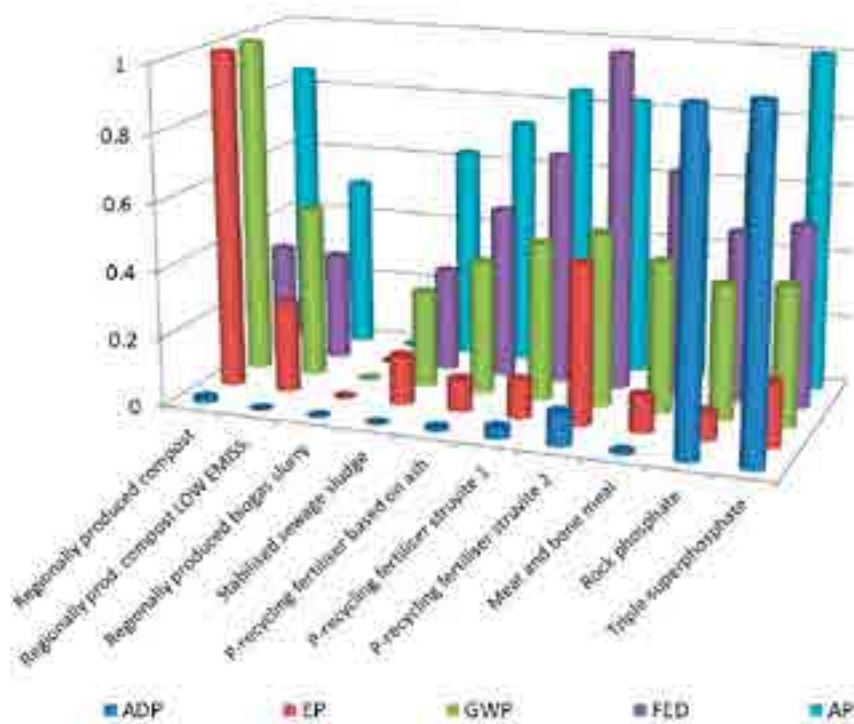


Figure 4. Relative values per kg P for abiotic resources depletion (ADP; in kg Sb-eq), fossil energy depletion (FED; in MJ fossil primary energy), global warming potential (GWP; in kg CO₂-eq), acidification potential (AP; in kg SO₂-eq) and eutrophication potential (EP; in kg phosphate-eq) from selected P-fertilisers between the best/lowest result per indicator (0) and the worst/highest result per indicator (1).

If the infrastructure is taken into account, the relative comparison would also show a slightly different picture. The most important change would be found for the indicators FED and ADP: after inclusion of infrastructure, highest values would result for recycled fertilisers (struvites or ash-based

RPF) due to their impacts of the chemical inputs and the highly technical production plants. However, even if the more comprehensive analysis on ADP including infrastructural processes shows a better result for the phosphate rock-based fertilisers, RPFs should be further developed and used before the fossil P deposits are depleted.

Results on the risk assessment: PTE accumulation in soil depended mostly on pollutant-to-nutrient ratios in fertiliser and the behaviour of the respective PTE in soil. Fertilisers with low P concentration in dry weight, like composts, had a high PTE accumulation in soil. The risk assessment calculation indicates the highest PTE accumulation rate for a soil with pH 7 and a low precipitation excess ($F = 0.1 \text{ m yr}^{-1}$), followed by scenario pH 7 $F 0.3 \text{ m yr}^{-1}$ and pH 5 $F 0.1 \text{ m yr}^{-1}$. The smallest accumulation potential showed a soil with pH 5 0.3 m yr^{-1} . This means, that PTEs in a soil with low pH and a high precipitation excess are leached to the water body. Exceedance of thresholds was found only for Cr, the other five PTEs remained below the threshold. The average European soil background concentration of Cr was already higher than the proposed threshold (244 kg ha^{-1}). Although in pH 5 scenarios a Cr depletion was found after 200 years, the threshold was not reached. In scenarios with pH 7, an accumulation was found for the composts (328 kg ha^{-1}), cattle manure and the digestate (OF), other fertilisers remained close to soil background concentration (308 kg ha^{-1}). Cd showed the highest accumulation potential in soil pH 7 for TSP (1.56 kg ha^{-1}) followed by composts and rock phosphate, cattle manure (1.23 kg ha^{-1}), while the other fertilizer were similar to the background concentration of 0.9 kg ha^{-1} . Cu was depleted or close to soil background concentration (56 kg ha^{-1}) with TSP, struvite and rock phosphate, meat and bone meal OF and increased with digestates, ashes, cattle manure, SS, untreated SS-ash and composts (88 kg ha^{-1}). For Ni we found a strong depletion in scenario pH 5 $F 0.3 \text{ m yr}^{-1}$ to approx. 17 kg ha^{-1} for all fertilisers, in scenario pH 5 0.1 m yr^{-1} between 71 to 61 kg ha^{-1} . Ni accumulated with compost fertilisation in soil pH 7, other fertilisers were close to soil background concentration (121 kg ha^{-1}). For Pb, no big difference was between the scenarios; green waste compost had again the highest accumulation potential (136 kg ha^{-1}), followed by compost catering waste, compost biowaste, digestate OF (111 kg ha^{-1}), SS ash, cattle manure and SS. Struvite, ashes, meat and bone meal OF, TSP and rock phosphate was close to the soil background concentration of 105 kg ha^{-1} . Zn was depleted in scenario pH 5 $F 0.3 \text{ m yr}^{-1}$. For soils with pH 5 and $F 0.1 \text{ m yr}^{-1}$, an accumulation for composts was found. In soil pH 7 all fertilisers led to Zn accumulation, especially composts (287 kg ha^{-1}), followed by SS ash, digestate OF, SS, most SS ash-based RPFs, cattle manure and digestate catering waste. Close to soil background concentration (221 kg ha^{-1}) remained meat- and bone meal OF, struvite, one SS ash-based RPF, TSP and rock phosphate.

For POPs no accumulation in soil was found, the calculated values after 200 years were lower than the soil background concentration.

4. Conclusions

In order preserve the essential element P for future agriculture, it is important to close nutrient cycles with recycled P-fertilisers. This needs the further development and improvement of suitable recycled P-fertilisers as well as an appropriate political framework (including farmers' associations guidelines, etc.) and their broad application in practical agriculture. From an LCA perspective, the different fertilisers show specific advantages and disadvantages, also depending on the system boundaries and the method of the assessment. Furthermore, a number of trade-offs between different indicators' results, including results from the risk assessment and other critical factors, was found.

However, it can be noted that – in the light of the results of this contribution – SS and its products should be used on behalf of the needs of future generations. SS with low pollutant contents could eventually be used directly, especially if it is not used for foods that are produced directly for human consumption. For SS with higher contamination levels, various processing options exist, which lead to a highly valuable products at equitable costs and equitable environmental effects (struvite or ashes for application in alkaline and acidic soils, respectively). Additionally, other treatment methods for organic wastes, e.g. anaerobic digestion to be used for energy-dense substrates and low emission composting on woody materials or the use of meat and bone meal, are recommended from the LCA perspective on behalf of P-supply for future generations.

The risk of the soil contamination with PTEs increased with increasing soil pH, decreasing precipitation excess and increasing pollutant-to-nutrient ratios in fertilisers. In general, for typical fertiliser application rates a low accumulation potential of toxic elements was found, especially for the recycled P fertilisers based on ashes, struvite and meat and bone meal. But even if the accumulation in soil is little, it should be considered that PTEs applied to the soil are partially transferred to other environmental compartments like water or dust where they may cause further negative environmental impacts. The POPs did not accumulate in soil and values after 200 years of fertiliser application remained below the soil background concentration.

5. Acknowledgements

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260. A Participatory Life Cycle-Based Sustainability Study On Using Unmanaged Land for Mānuka Oil Production On the East Coast of New Zealand

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ABSTRACT

This study investigated the sustainability impacts of cultivating mānuka (a native tree species) and subsequent essential oil production in the Waiapu catchment on currently unmanaged land owned by Ngāti Porou, the indigenous Māori tribe in the area. A participatory Life Cycle Sustainability Assessment (LCSA) was undertaken with three members of Ngāti Porou involved with decision-making regarding Māori land. Mānuka oil provided a regular and steady distribution of costs and profit, and employment, due to the annual cycle of branch trimming, distillation, and product creation. The highest greenhouse gas emissions were during harvesting/trimming, and the limited data on carbon sequestration suggested that a majority of the carbon lost during land use change was re-sequestered in the mānuka plantation. Culturally, the mānuka scenario was largely seen to enhance Ngāti Porou's aspirations. The participatory LCSA approach was seen by the participants as useful because it engendered engagement with issues such as Māori ownership of different life cycle stage activities (e.g. oil distillation), how to make better use of by-products and waste materials in the life cycle, and how to enhance cultural aspirations. Distinct representation of a cultural indicator (and not merely embedding culture within social indicators) was appreciated because it made cultural aspects more visible in the decision-making process.

Keywords: mānuka plantation, Life Cycle Sustainability Assessment, cultural assessment, Māori

1. Introduction

Forests cover a significant proportion of New Zealand's land and comprise protected nature reserves of indigenous tree species (covering 26% of the land area, 6.8 million hectares) (Ministry for Primary Industries, 2015), and exotic planted forests (7% of the land area, 1.77 million hectares). Radiata pine (*Pinus radiata* D.Don) accounts for 87% of the exotic planted forests (Ministry for Primary Industries, 2015). However, there is a growing interest in cultivation of alternative species to radiata pine (Fairweather & Hock, 2004; Rotarangi, 2012), and in particular indigenous species such as rimu (*Dacrydium cupressinum* Lamb.) and mānuka (*Leptospermum scoparium* J.R.Forst. & G.Forst. (Myrtaceae)).

Māori are the indigenous people of Aotearoa (New Zealand), and own a significant amount of forested land: 520,000 hectares of exotic forests (30% of New Zealand's exotic forest land), and 600,000 hectares of indigenous forests (Holt & Bennett, 2014). Māori forest owners have a particular interest in indigenous species but at present there is limited information available about alternative forestry options (Te Puni Kōkiri, 2011).

To address this knowledge gap, a Life Cycle Sustainability Assessment (LCSA) was undertaken of three potential forestry scenarios (radiata pine, rimu and mānuka) and three associated products (pine house framing, rimu flooring, and mānuka essential oil). This paper discusses the mānuka scenario and use of these LCSA results to support decision-making about currently unmanaged land owned by Ngāti Porou, the indigenous Māori tribe in the Waiapu catchment on the East Coast of North Island, New Zealand. Mānuka is a native tree species that is commonly utilised for the production of honey. However, the foliage can also be harvested to produce mānuka essential oil which is used for medicinal (anti-bacterial, anti-fungal, anti-viral) purposes (Porter & Wilkins, 1998). This is an emerging industry in New Zealand and there are currently no established large-scale commercial mānuka plantations focused on essential oil production (although wild-growing mānuka branches are harvested and used by a small number of oil producers). Therefore the LCSA study reported here involved modelling the life cycle for mānuka essential oil using estimated data provided by experts.

The LCSA study was undertaken within a wider participatory process that involved collaborative engagement of the LCSA researcher with three members of Ngāti Porou who are regarded as community leaders. Over a period of three years, a series of nine significant semi-structured interviews and several informal discussions with the Ngāti Porou participants were held. The focus of the interviews was the co-development of the life cycle scenarios including choice of forestry scenarios and associated products, and LCSA indicators. More details about the process can be found in Pizzirani et al. (forthcoming).

2. Goal and Scope, and Inventory Analysis

The purpose of the LCSA was to evaluate the potential benefits and disadvantages of cultivating mānuka on currently unmanaged land owned by Ngāti Porou on the East Coast of North Island, New Zealand. The life cycle of mānuka essential oil was selected for the study because this product has potential to develop into a significant economic activity for the region; mānuka oil from this region has been shown to naturally have the most potent (and therefore the most valuable) medicinal qualities and may therefore be more valuable than mānuka oil from other regions (Douglas et al., 2004). The functional unit was defined as “production and use of mānuka essential oil from one hectare of currently unmanaged land in the Waiapu catchment”; this was modelled as 9,712 mānuka essential oil-filled 10mL bottles (i.e. 5% of total oil produced). The system boundaries were set to include establishment and cultivation of the mānuka over a 15-year rotation (including harvesting of the branch material each year from year 2 to year 15), through the distillation process to bottling of the oil, and final waste management of the used bottle.

During the participatory process, the group decided to focus on the following indicators for the study: economic costs and profits, employment (hours of labour), greenhouse gas emissions, carbon sequestration, and cultural impacts. It was recognized that a wider range of indicators was desirable but this was not possible in the present study due to time constraints.

Primary data were collected during interviews with industry experts, and represented the majority of the data used for modelling the nursery, site preparation, tending, harvesting, distillation, and packaging processes. Secondary data sources (including modelled data, reports, and the v3.1 ecoinvent database) were used for the glass production and landfill processes, and production and use of fertilisers and energy (diesel, petrol, aviation fuel, electricity); cost of soil analysis; tractor diesel use; above- and below-ground carbon loss when unmanaged land is converted; and tree carbon sequestration rates. A full description of the data used in the analysis can be found in Pizzirani (2016). For the cultural impacts, the Cultural Matrix Indicator was used (as described in Pizzirani et al. (in review)).

3. Inventory Analysis

The processes included in the study are shown in Figure 1. The mānuka life cycle involves nine main processes:

1. Nursery: the life cycle begins in the tree nursery where seedlings are grown over a period of 12 months.
2. Site preparation: the unmanaged land is prepared for mānuka cultivation; this involves spraying of herbicides to kill existing vegetation, fertiliser application, and tilling of the soil.
3. Planting: a high number of seedlings are planted (15,000 seedlings per hectare) with the purpose of producing a substantial amount of branch material each year.

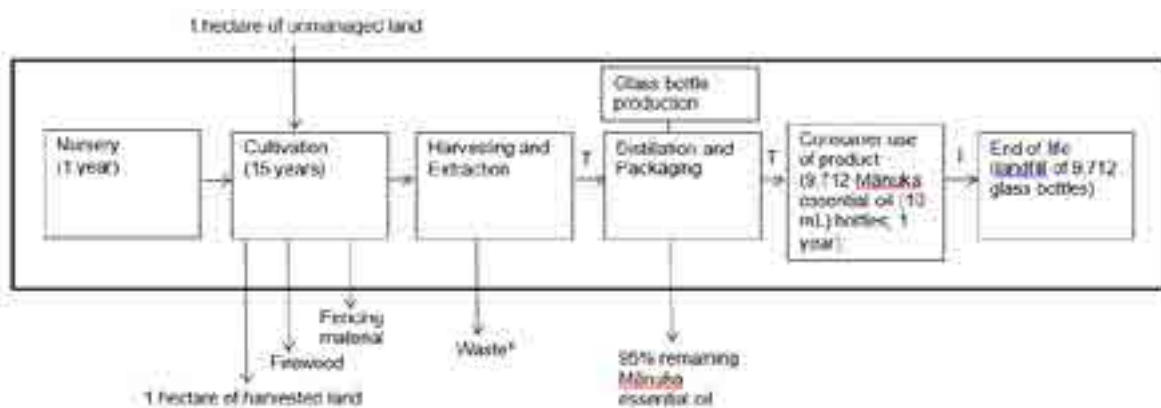


Figure 1: Life cycle of mānuka essential oil

*Distillation produced steamed branch matter that may be considered waste yet is also viable mulch material which could be sold as a co-product. (T indicates freight transport)

4. Tending: the tending processes involve the replacement of dead or underperforming trees to ensure a consistent annual growth rate of branch material per hectare. In year 1, it is assumed that 10% of the planted seedlings will need replacing, 7% in year 2, and 5% each year in years 3 to 13 of the rotation; in total, 10,800 seedlings per hectare are replaced throughout the rotation.
5. Harvesting/trimming: at year 2 the seedlings are mature enough to be trimmed. The top 2/3 of the tree is trimmed using a forage harvester with the branch material being deposited into an adjoining truck with a trailer which takes the material to the distillation site². An average amount of just under 39 tonnes per hectare per year is harvested in years 2 to 15, equalling a total of 544 tonnes of branch material per hectare per 15-year rotation. During the final year of the rotation, the full mānuka tree is harvested (not just the top 2/3 of the tree). Thus, at year 15 there is a higher yield of harvested branch material which equates to higher volumes to be distilled – this leads to higher costs and GHG emissions (from the increased time required for harvesting and distillation), and higher profits (from the increased quantity of oil produced).
6. Distillation: the distillation process involves steam distilling the mānuka branch material to produce mānuka essential oil. Approximately 3.5 kg of essential oil is produced for every tonne of mānuka branch material distilled (P. Caskey, personal communication, February 2,

² For the purposes of this study, it was assumed that the mānuka cultivation site is located adjacent to the distillation site. Therefore, no impacts associated with the transport of the branch material were included in this study.

2015)). Therefore, if 544 tonnes of mānuka branch material is produced during a 15-year rotation, then 1,904 kg of essential oil is created during the distillation process.

7. Packaging: it is estimated that 5% of the oil is packaged in 10mL bottles (P. Caskey, personal communication, February 2, 2015). Thus, 5% of the total oil produces 97,125 mL, or 9,712 10mL mānuka essential oil-filled bottles (as 1kg of oil equals 1020.41mL of oil (assuming the density of mānuka essential oil is 0.98 (New Zealand Manuka Bioactives, 2016)).
8. Consumer use: use of the oil has cultural value but is not associated with any other impacts (transportation varies with individual supply chains, and so this is not included in the analysis, and carbon sequestration in the oil occurs only for a short time period prior to use of the oil).
9. Disposal: the empty mānuka essential oil glass bottles are assumed to be disposed of in a landfill.

For this study, as noted in Step 7 above, it was assumed that only 5% of the total oil was packaged into 10mL bottles due to a limited market for this product; the remaining 95% of the oil was assumed to be sold in bulk to other industries for use in beauty and health care products. The impacts associated with the mānuka establishment, cultivation and distillation, were allocated between the essential oil and bulk oil supply on a mass basis.

The fencing material and firewood by-products from the mānuka cultivation (as highlighted in Figure 1) were regarded as “free” by-products from the mānuka plantation: any firewood would be collected from the land regardless of whether it was unmanaged land or mānuka plantation, and the fencing material is used within the product system to protect the mānuka plantation.

For the calculation of costs, the actual material costs (e.g. fuel, machinery, fertiliser) and labour costs (including overheads) were aggregated; the associated data were gathered from industry experts. In addition, an opportunity cost was included for the land use. The rationale for its inclusion was that, when the cultivation of mānuka was chosen over another option (e.g. land rental), there was a forfeited economic gain from the option not pursued, and this should be represented in the mānuka study. The opportunity cost in this research was based on the alternative option of land rental (Prokofieva & Thorsen, 2011). It was determined by finding out the price of current flat land for sale in the case study region, and taking 4% of this price as a reasonable representation of the minimum return on investment amount (P. Hall, personal communication, 15 March, 2014).

For the calculation of profit, the life cycle from the nursery through to the distillation facility was assumed to be owned and operated by one “entity” (i.e. the participants and the Ngāti Porou iwi). The remainder of the life cycle (i.e. packaging of the mānuka oil and waste management) was assumed to be operated by external entities. This meant that there were two points where profit was realised: after the distillation process and after the packaging process. For the distillation process, profit was calculated by subtracting the total costs (inclusive of the nursery process through the distillation process) from the amount paid (i.e. value) for the mānuka oil after distillation. For the packaging process, profit was calculated by subtracting the aggregated cost of purchasing the essential oil from the distillers (as well as the cost for packaging materials and labour) from the profit realised per packaged bottle of mānuka essential oil.

For the employment indicator, labour hours for each process were estimated based on industry expert knowledge.

Greenhouse gas emissions were calculated for energy production and use (i.e. diesel, petrol, jet fuel, electricity), and the production and use of pesticides, herbicides, and fertiliser. The greenhouse gas emissions also included the loss of carbon (above-ground and in the soil (to 30cm)) associated with land use change. The initial loss of above-ground carbon when scrubland with woody biomass was cleared, was 47,850 kgCO₂eq per hectare (Wakelin & Beets, 2013) and the loss of soil carbon (to 30 cm depth) when converting from scrubland to the mānuka plantation was 63,800 kgCO₂eq per

hectare (based on Hewitt et al., 2012). According to the PAS 2050 (BSI, 2011a), the total GHG emissions from land use change are to be allocated evenly across the next 20 years. Therefore, 75% of the total loss of carbon was allocated to the full mānuka rotation.

For calculation of (above-ground) carbon sequestration during the mānuka plantation, the weighted average carbon sequestration in above-ground biomass over the rotation was calculated based on data in Scott et al. (2000) i.e. the amount of additional carbon sequestered each year of the rotation was calculated and then weighted by the length of time it was sequestered. This came to 24,545 kg carbon; assuming the land continues to be used for mānuka cultivation in future, this can be regarded as a form of ongoing carbon storage (equivalent to a weighted average of 90,090 kg CO₂eq). The same approach was used for carbon sequestration in the nursery seedlings.

For the Cultural Matrix indicator, the three participants individually and subjectively scored each process in the mānuka life cycle according to how they perceived it may impact on their identified cultural aspirations. The Cultural Indicator Matrix uses a scale of -2 (a degrading effect on aspirations) to +2 (a flourishing effect on aspirations); a score of 0 is considered to have a neutral or maintaining effect on aspirations. The results were calculated as averages and represent the participants' impressions regarding the mānuka life cycle.

4. Impact Assessment Results

The LCSA results (Table 1) showed that there was a total life cycle cost of \$150,353 and a total profit of \$48,081; the life cycle stages associated with highest costs were Distillation and Packaging, and the life cycle stage at which the greatest profit occurred was Packaging. The life cycle stage contributing the highest number of hours of employment was the Packaging stage. The highest GHG emissions (5,082 kgCO₂eq) were associated with the use of diesel during the Harvesting/trimming process. Overall, carbon sequestration in the mānuka plantation was equivalent to about 80% of the total loss of carbon during conversion from unmanaged land to mānuka plantation; however, this result should be interpreted with caution given the paucity of data on carbon sequestration. The Cultural Indicator Matrix result of 0.97 indicated that the participants believed the mānuka life cycle to have an overall enhancing effect on Ngāti Porou's aspirations; the most favourable process was the Nursery while the least favourable process was the Landfill.

Table 1. LCSA impact assessment results for 9,712 mānuka essential oil-filled 10mL bottles.

LCSA impacts for 9,712 mānuka essential oil-filled 10mL bottles	Total cost	Value added	Employment (hours)	GHG (kgCO ₂ eq)	C-sequestration (kg C)	C.I.Matrix
Nursery	1404	238	52	42	82	1.81
Site prep	68	0	0	4196	0	1.43
Planting	292	0	13	0	0	1.52
Tending	481	0	8	68	1229	1.43
Opp cost	780	N/A	N/A	N/A	N/A	N/A
Harv/trimming	1500	0	27	5082	0	1.43
Distilling	24841	3714	109	1220	0	0.62

Packaging	120917	44130	324	220	0	0.90
Use	N/A	N/A	N/A	0	0	0.90
Landfill	71	0	0	188	0	-1.33
TOTAL	150353	48081	532	11015	1311	0.97

An alternative way of presenting the costs and profits, GHG emissions, and employment results is along timelines (Figure 2). It can be seen that the mānuka scenario has a steady occurrence of costs as well as profit due to the annual trimming, distilling, and product creation activities. This is also seen in both the GHG and employment results as both indicators are largely influenced by specific life cycle activities i.e. the labour-intensive nursery, harvesting, and distillation processes. Figure 2 also shows the Cultural Indicator Matrix results by activity as opposed to time (because the Cultural Indicator Matrix values cannot be arbitrarily divided by the number of years a process occurs).

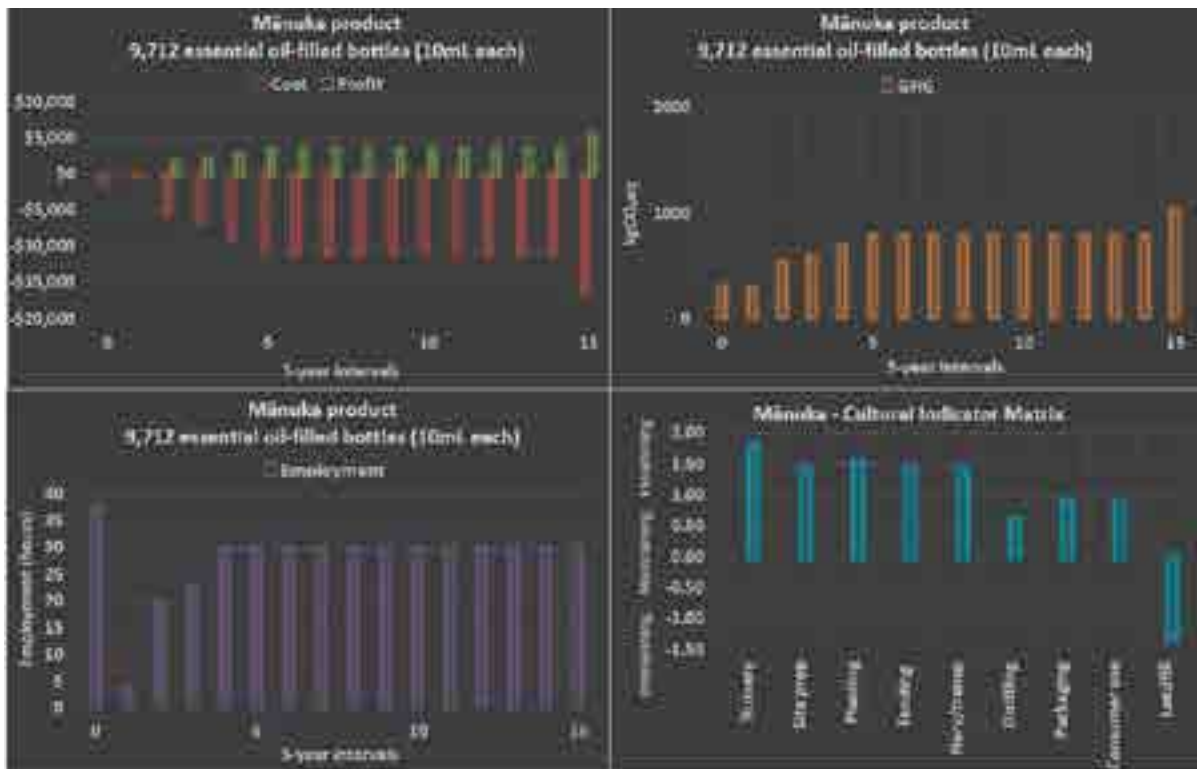


Figure 2: LCSA results for to 9,712 mānuka essential oil-filled 10mL bottles i.e. 5% of the total essential oil produced per hectare of mānuka produced over 15 years. **The Cultural Indicator Matrix results are presented per life cycle process and are not scaled to the quantities of material utilised or activities undertaken in development of a product.**

4. Discussion

The results showed that the greatest economic costs, and greatest profit, occur at the packaging stage. This suggests that if Ngāti Porou can retain ownership of the supply chain through to sale of the packaged bottles, they are likely to realize the greatest profits. Furthermore, given that only 5% of the mānuka oil is assumed to be packaged in this way, there is scope for much greater profit for Ngāti Porou if they can identify a larger market for these small packaged bottles as opposed to bulk sales to beauty and health care product manufacturing companies.

Employment was also highest at the packaging stage (assuming use of manual labour for labelling, filling, etc.). This was regarded as a positive impact by the participants who are striving to create more employment in their region. However, the economic cost of employment may alternatively be considered a burden and one that may be reduced during the packaging process by automating the labelling, filling, and packaging of the bottles instead of completing this task by hand.

The highest GHG emissions were associated with the distillation process; this process required 15 litres of diesel to steam distill one tonne of mānuka branch material. Conversion of the distillery to utilize alternate forms of energy may decrease not only the GHG emissions but also the economic cost of this process. In addition, utilisation of the by-products of the mānuka life cycle (e.g. mānuka wood material produced during cultivation for use as firewood or fencing; steamed mānuka branches after distillation for use as mulch) may increase the potential for economic gain.

Culturally, the mānuka essential oil life cycle was seen by the participants as having an overall enhancing effect on their tribal aspirations. However, the Cultural Indicator Matrix results highlighted that the distillation process had one of the lowest scores across the range of activities in the life cycle. In a subsequent discussion about the results, the participants explained that they had little to no interaction and/or knowledge about the distillation process, and scored it according to how they perceived the distillation process. For example, the participants scored the distillation process as having a fairly neutral impact on their employment and healthy ecosystem aspirations. However, an existing combined mānuka distillery and honey production plant in the area already provides over 100 jobs and has numerous environmental protection protocols in place. Therefore, the distillation activity may actually have more a positive impact on Ngāti Porou's cultural aspirations than previously believed by the participants. This result showed the need for increased engagement between the distillers and Ngāti Porou.

Overall, the LCSA results enabled the participants to identify hotspots i.e. areas in the life cycle-value chain where value does not currently exist for Ngāti Porou. These areas were reviewed and opportunities for creating more value were discussed. They included retaining ownership of the supply chain through to a final product (from both economic profit and employment perspectives), and capitalising on the by-products of mānuka cultivation and distillation to reduce the overall greenhouse gas emissions (as well as increasing profit). For the Cultural Indicator Matrix results, it was recognized that the individual scores of the participants for the different life cycle processes needed to be reviewed in more depth in order to develop a better understanding of why the participants had different scores.


5. Conclusions

The LCSA results show that mānuka oil provides a regular and steady distribution of costs and profit due to the annual cycle of branch trimming, distillation, and product creation. This can be contrasted with other forestry scenarios where there is a more uneven distribution of costs and profits due to their longer rotations (e.g. pine and rimu). Culturally, the mānuka scenario is largely perceived as enhancing Ngāti Porou's aspirations (Table 2).

The use of the LCSA approach was seen by the participants as useful because it engendered engagement with issues such as Ngāti Porou ownership of different life cycle stage activities (e.g. oil distillation and subsequent packaging into final products), how to make better use of by-products and waste materials in the life cycle, and how to enhance cultural aspirations. Inclusion of a cultural indicator (and not merely embedding culture within social indicators) was appreciated because it made cultural aspects more visible in the decision-making process.

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67. A preliminary methodological framework to assess potential contributions of food to sustainable transformation

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ABSTRACT

The paper presents the methodology applied for the development of a framework method to assess and evaluate sustainability impacts along the life-cycle of products with a focus on food. The sustainability evaluation framework method covers the three sustainability dimensions: the socio-ethical, ecological and economic dimension. Building on and adapting established methods a main new aspect of the method is that not only potential negative impacts are assessed but also potential positive impacts regarding sustainable development. Furthermore the method will evaluate the contribution of products and services to sustainable development. Thus, an evaluation scale is to be set. Including literature review and stakeholder consultations, the selected evaluation scale builds on the Sustainable Development Goals (SDGs), created by the United Nations (UN). During the development process a focus was set on the balancing between precision, validity and effort, as the framework method shall be applicable with acceptable efforts while still being viable. Furthermore with the goal of an easy communication of the method's results the options of aggregations are to be discussed. This framework method shall help users to contribute to a transformation of economy and society towards sustainability, through the reduction of potential negative impacts, but also through increase of potential positive impacts.

Keywords: sustainability assessment, positive impacts, SDGs, sLCA, eLCA,

1. Introduction

A range of frameworks, methodologies and tools for assessing sustainability impacts, including product related assessments (e.g. Singh et al. 2012) have already been developed. Beside the need for further research on certain aspects, in particular for the economic and social dimensions and also some impact categories (ICs) like water or biodiversity, tools like environmental Life Cycle Assessment (eLCA)³ covering impacts on the environment are well established. However, a holistic framework for product sustainability assessment is not yet established.

Objective is to develop a scientific and feasible framework method for an integrated evaluation of the contribution of products and services to sustainable development, covering the whole life-cycle.

Thus, the present paper focuses on the development of the framework method as well as on its application on food products. It is part of a research project with a wider scope of products and services including further industry branches like consumer goods and housing (Beckmann et al. in preparation). The research project covers several sub targets:

1. Aim is that the framework method not only allows the assessment of potential negative impacts but also to take potential positive impacts regarding the socio-ethical, ecological and economic dimensions of sustainability into account. Negative impacts are defined as effects that should be decreased regarding sustainable development, e.g. greenhouse gas emissions. Accordingly positive impacts are defined as effects that should be increased, e.g. carbon sequestration. Hypothesis is that negative impacts should not be mixed up with positive impacts but should be assessed and accounted for separately to allow the identification of key parameters for transformation to sustainable development.

³ DIN EN ISO 14040: 2006; DIN EN ISO 14044:2006

2. Another target is that results shall be evaluated regarding their contribution to sustainable development. Thus, an evaluation scale to assess the potential contributions to sustainable development has to be set within the methodology.
3. A further goal is that the framework method shall be applicable with acceptable efforts: as simple as possible while the most precise necessary. This means that a challenge in the development of the framework method is the balancing between precision, validity and effort.
4. Furthermore, due to the fact that the results shall be easy to communicate to business partners or consumers, the possibility of further aggregation of evaluation results is examined within the development process - which may result in a single characteristic or reference number.

The methodology section describes the steps and scientific legitimation leading up in the development of the methodological framework. In the results section first results of the development of the framework methodology and its application on food products are presented, while in the last chapter the presented methodology and results are discussed and first conclusions are drawn.

2. Methodology

First a literature review of existing product assessment methods in the context of sustainability has been carried out to identify which yet existing methods are suitable for the framework method to be developed.

Second, a literature review regarding systems of value for sustainable development and sustainability evaluation methods has been conducted. Potential sources for such a normative system of values are science-based and policy based (Acosta-Alba & Van der Werf 2011). In the literature review these were extended with systems of value by economic actors and their institutions.

Third, approaches to simplify complexity were investigated. This includes the selection of material impact categories (IC), for which the two main approaches have been used: Top-Down and Bottom-up (Spohn 2004). Socio-ethical categories were identified through Delphi studies including a rating system. These socio-ethical ICs include overlapping economic topics which are further specified through a literature review of indicator schemes and are rated through stakeholder consultations in workshops. The selection of material ecological ICs has been based on scientific studies like the concept of Planetary Boundaries (Rockstroem et al. 2009) and the results of the Product Environmental Footprint (PEF) which is currently developed including multiple pilot studies (European Commission 2016).

Fourth, a further literature review on aggregation methods is planned but not yet conducted.

Based on the results to date of the described first three steps the preliminary framework method presented in section 3.4 was developed.

The development of the framework method is accompanied by regular stakeholder consultations in workshops within the different project phases, as well as expert consultations. For further adaptations case studies with business partners from the food industry are conducted.

3. Results

Following the objectives and the methods applied, the results are structured in first the literature review of existing methods for product sustainability assessment, second the results for the research on value systems as well as evaluation methods, followed by the results on reducing complexity. These results are brought together in chapter 3.4 with respect to the preliminary framework methodology.

3.1 Literature review of existing methods

Environmental Life Cycle Assessment (eLCA) is an established method for the assessment of a product's potential environmental impacts (ISO 14040series). Also for the still developing assessment of potential social impacts, social Life Cycle Assessment (sLCA) guidelines developed by UNEP SETAC (2009) are existent. The set of product assessment methods is completed by Life Cycle Costing (LCC), an assessment method for costs along a product's life cycle (Swarr et al. 2011).

Furthermore, several single IC methods have been developed in recent years like the Product Carbon Footprint (ISO/TS 14067:2013) or the Product Water Footprint (ISO 14046:2014). Also several other institutions like the World Resources Institute (2004) with the Greenhouse Gas Protocol (GHG 2011) or the Water Footprint Network (WFn 2016; Hoekstra et al. 2011) have designed methods to assess environmental impacts within single ICs or for single impact indicators.

In particular the first mentioned methods eLCA, sLCA and also LCC build up on a yet established and accepted approach to assess impacts regarding sustainability dimensions along a product's life cycle. Thus, Klöpffer (2008) understands the combined use of all three methods as Life Cycle Sustainability Assessment (LCSA). Also further methods developed to assess sustainability of products like the Product Sustainability Assessment (PROSA) of the German Institute for Applied Ecology (Grießhammer et al. 2007) or the AgBalance method developed by BASF (Frank 2014; BASF 2016) are based on eLCA, sLCA and LCC but can include also further assessments like e.g. a benefit assessment (PROSA) to get a more holistic picture with respect to sustainable development.

The existing methods all cover more (eLCA) or less (sLCA) established sets of ICs and indicators. In particular regarding sLCA the adequate ICs and indicators have to be chosen out of a pool of possible categories and indicators. But also within eLCA impact assessment methods differ within different assessment methods which results also in different indicators used for similar ICs, e.g. ecotoxicity potential is calculated very differently in ReCiPe and in Usetox resulting in 1,4 DCB-equivalents in Recipe Midpoint (Goedkoop et al. 2012) and in Comparative Toxic Units (CTU) in Usetox (Fantke et al. 2015).

Furthermore, in further research within the study it will also be analysed if – from a sustainability perspective – all relevant impacts are covered within the impact category sets used in existing methods. Hypothesis is that this is not always the case.

However, the analysed methods allow separately accounting for potential positive and negative impacts. Thus, the framework of LCA-methods can be used as basis for the development of the holistic sustainability evaluation method.

3.2 Literature review of value scales and evaluation methods

LCA methods do not include an evaluation step. Nevertheless several methods have yet been developed not only to assess potential impacts but also to assess the impacts from different ICs (e.g. Schmitz/Paulini 1999) and also to evaluate impacts between different sustainability dimensions, e.g. PROSA, Eco-efficiency assessment of product systems (ISO 14045:2012), AgBalance. However, to evaluate a product's potential contribution to a more sustainable development a normative value system is needed as evaluation scale. Thus, such an evaluation scale has been looked for to be used within the framework method.

In the literature review a variety of normative value systems have been reviewed, including for example Green Growth (OECDa 2016) and the Better Life Index (OECDb 2016) by the Organisation

for Economic Co-operation and Development (OECD), as well as the work of the Enquete Commissions of the German Parliament (Deutscher Bundestag 1998, 2013), as well as concepts from NGOs like the Living Planet Index, used by the WWF (2014), or Vision 2050 by the World Business Council for Sustainable Development (WBCSD 2016).

The review showed the Sustainable Development Goals (SDGs) by the United Nations (UN 2015) as promising for the purpose as evaluation scale, which was confirmed in stakeholder consultations. The SDGs integrate a wide range of sustainability issues, cover all three dimensions of sustainability, and also give a vision and concrete targets. Furthermore, shaped by the UN and accepted by governments worldwide, they are globally agreed upon without regional limitations and bias. Thus, the SDGs will form the normative value system to be used within the framework methodology.

The consecutive question is how to evaluate the contribution of a product to more sustainable development. For this purpose a 'distance to target' method seems to be most suitable (Acosta-Alba & Van der Werf 2011). Thus, several yet existing methods, e.g. EcoGrade (Bunke et al. 2002, Möller et al. 2005), Ecological Scarcity (Frischknecht et al. 2009, Grinberg et al. 2013), have been analysed with the aim to identify suitable components.

All methods analysed have in common that they measure the distance to a quantitative target, e.g. the distance to the target to reduce greenhouse gas emissions. However, they propose slightly differing calculation methods to calculate the respective distance.

Regarding the development of the framework method the calculation of the distance to target has to show if the respective effect reduces the distance to target, e.g. carbon sequestration reduces the distance to target with respect to stop climate change, or if the effect augments the distance, i.e. further greenhouse gas emissions increase the distance to target with respect to stop climate change. The calculation method to be used within the framework method is still under development.

3.3 Reducing complexity

Reducing complexity is tackling the potential trade-off between precision and practicability. Aim of the framework method under development is to reduce efforts but to allow valid results. Several standards dealing with sustainability try to reduce complexity, e.g. the Global Reporting Initiative (GRI) introduced in their version 4 a materiality analysis to 'make reports more relevant, more credible and more user-friendly' and in order to center sustainability reports on 'matters that are really critical in order to achieve the organization's goals and manage its impact on society' (GRI 2013). And also the PEF process is trying to identify most relevant ICs to be tackled per product group within pilot studies and expert consultations (European Commission 2016).

Aim of the present study is to keep the number of ICs as low as possible, but high enough to ensure conclusive results. First results regarding the identification of relevant ICs show that health and safety, employment relationships, information and transparency as well as issues like fair pricing, infrastructure, knowledge provision and product utility for the social dimension and the closely related economic dimension should be included. Within the environmental dimension the ICs biodiversity, land use, climate change, toxicity, eutrophication, radiation, acidification, ozone depletion, summer smog, water depletion, quality of soils, odour and noise pollution, thermal discharge and waste have to be considered. Aim of the next step is to investigate for this pool of ICs if it is possible to identify core ICs, which have to be analysed for every product, and product group-specific ICs, which have to be analysed for specific product groups.

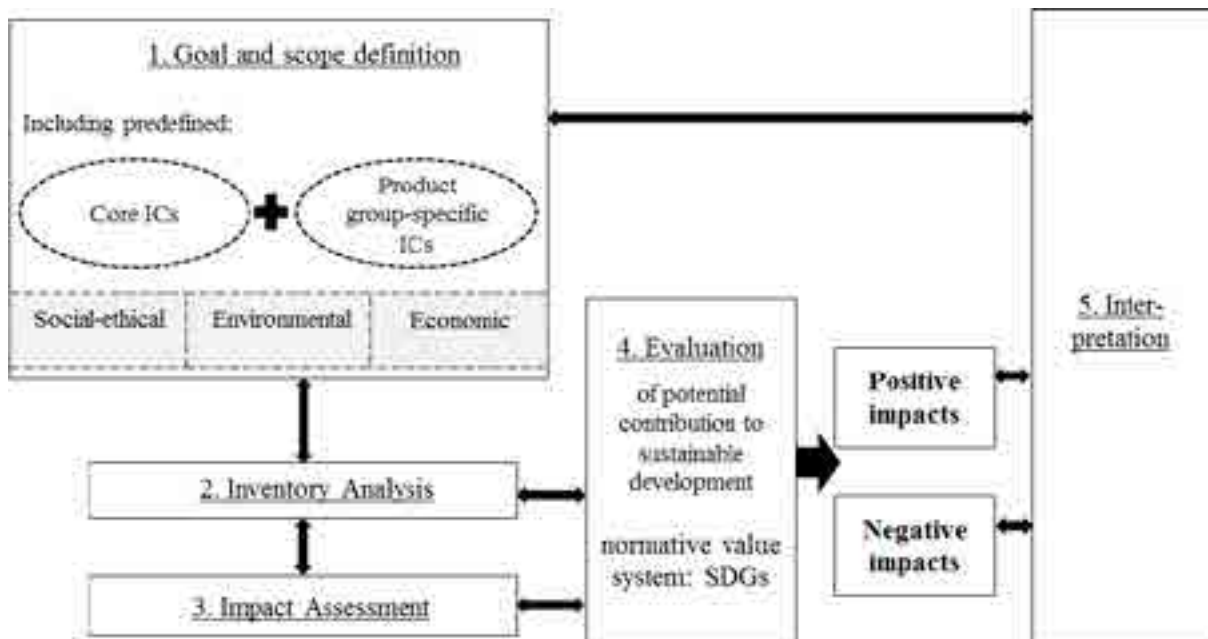
Regarding food the first selection of ICs showed that highly relevant ICs are water and land use, impacts on biodiversity, toxicity, greenhouse gas emissions, soil quality regarding the environmental dimension, and labour conditions, child and forced labour, health and safety, fair payment as well as

animal welfare regarding the socio-ethical and economic dimensions. If this first selection is significant will be further tested within food case studies, which will include an analysis for as many ICs possible (not for all ICs mentioned in the pool yet impact assessment methods are available) and also an analysis for selected ICs to enable comparison.

3.4 Preliminary framework method for food

The framework method is based on the established tools eLCA and sLCA, analysing the whole life-cycle of a product and relating to a clearly defined functional unit. How to tackle the economic dimension is still under investigation and will be included in the framework within the next month. The scheme is developed according to the LCA framework and was adapted: 1. Goal & scope definition, 2. Inventory Analysis, 3. Impact Assessment, 4. Evaluation, and 5. Interpretation. Figure 1 shows the preliminary framework method, including the adaptations.

Figure 2: Preliminary framework method for food (adapted according to ISO 14040:2006)



The following adaptations have been done:

1. Goal & scope definition: core ICs and product specific ICs are predefined within the method and are obligatory.
2. Evaluation: this step is introduced after the inventory analysis and impact assessment have been carried out. Results will be evaluated against the defined normative value system of the SDGs. In addition, to get a better understanding of options for optimisation, the results are depicted in a way that positive effects, e.g. sequestration, and negative effects, e.g. emission of greenhouse gases, are shown separately.

In addition, after the evaluation an aggregation step can be added. If and how this will be done is still under investigation.

4. Discussion & Outlook

The framework method makes use of accepted tools for the assessment of product sustainability. But due to adaptations made it opens the opportunity to identify also potential positive sustainability impacts. The most important new steps introduced in the framework are: predefinition of core and product specific ICs, evaluation with clear distinction between negative and positive effects, and the possibility of aggregation (& communication).

However, the presented framework is still preliminary and there is still some work left to be carried out in the next month. In particular the economic dimension has to be further investigated regarding assessment methods (will we include LCC or do we make use of a different approach?), but also the calculation method for the 'distance to target'-approach and the aggregation step are crucial for the validity of the framework method.

The SDGs are globally agreed upon and thus form a well-justified normative value system. Nevertheless it will need some further work to translate the 17 goals and their sub targets into a practicable set of quantitative goals.

Further research will also include pilot case studies in cooperation with industry partners. Within the case studies the framework will be applied and validity will be tested.

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161. Exposure to chemicals in food packaging as a sustainability trade-off in LCA

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ABSTRACT

Hazardous chemicals in packaging, including ‘eco-friendly’ and recycled food packaging, can migrate into food and expose humans. LCA has been fundamental to indicate more ‘eco-friendly’ packages, but currently LCA does not consider exposure to chemical migrants and methods have not yet been developed. In this study we question if exposure to chemicals in food packaging should be considered as a sustainable design consideration, i.e. if this human health risk is relevant in a life cycle context. To answer this question, we focus on developing methods to quantify exposure to chemicals in food packaging in a life cycle impact assessment (LCIA) framework. To put exposure during use in a life cycle context we perform a screening-level LCA of several life cycle stages of high impact polystyrene packaging (HIPS), with a functional unit of containing and delivering one kilogram of yogurt for consumption. For screening, we include exposure via environmental emissions from the production of the raw material HIPS, thermoforming into packaging, 14 day refrigeration by consumers, and disposal via incineration. The purpose of this screening is not to obtain a detailed and accurate LCA of HIPS but to provide life cycle context to compare the magnitude of characterized exposure to chemicals in packaging, in order to elucidate if this exposure pathway is important. We detail estimates of life cycle exposure to one known hazardous chemical in polystyrene packaging (styrene) that has data available on concentrations in yogurt packaged in HIPS and life cycle inventory releases. We also extend this analysis, given data limitations, to include exposure to three other chemicals in HIPS packaging through food. Given that data on concentrations of food packaging chemicals in food are often missing, we also explore methods to model the product intake fraction (PiF) as the fraction of chemical mass taken in through food packaging versus its initial mass in the food packaging. Results demonstrated that in the given cases consumer exposure to chemicals in packaging through consuming packaged food can be greater than population-level exposure mediated by the life cycle releases of such chemicals, even when only considering one or several chemicals in packaging that expose consumers. Occupational exposure was not considered in this study, but could be a focus of future work. Thus, this initial exploration indicates that exposure to chemicals in food packaging can be an essential consideration for burden shifting and quantifying design trade-offs in a life cycle context.

Keywords: life cycle impact assessment, food contact materials, high impact polystyrene, human health, product intake fraction

1. Introduction

Food packaging claiming to be more environmentally sustainable has the potential to increase human exposure to toxic chemicals through packaged food. For example, recent tests that compared with conventional food packaging have detected higher levels of toxic fluorinated chemicals (Blum et al. 2015) in ‘eco-friendly’ food packaging (Yuan et al. 2015) and phthalates (Serrano et al. 2014) in recycled food packaging (Gärtner et al. 2009). Very few assessment frameworks evaluating ‘sustainable’ product design, for instance for packaging, consider human exposure to chemicals when *using* a product alongside a product’s environmental impacts. To date, established methodologies are restricted to hazard identification. For example, products that are Ecolabel licensed (www.ecolabel.eu) or Cradle to Cradle Certified^{CM} (C2C) must verify they do not contain a list of hazardous chemicals of concern, e.g. according to the C2C Material Health Criteria, in order to increase safety beyond current

regulatory requirements. Using a qualitative hazard-identification based method offers several advantages, namely quantification of exposure is not needed. However, such an approach does not offer the possibility for exposure minimization strategies for the thousands of chemicals that are (or are not) on an assessment's self-made 'banned' list, but legally allowable in food contact materials. Furthermore, such a hazard-based approach does not allow for considering exposure to chemicals in packaging as a system or design impact trade-off.

Life cycle assessment (LCA) has been extensively applied to food and beverage packaging systems (Flanigan et al. 2013; Grönman et al. 2013) and results have for example supported transitioning from glass to plastic food contact materials to decrease environmental impacts (Hunt and Franklin 1996). Specifically, LCAs on packaging for baby food, injectable medicines, and beverages have indicated plastic having less potential for human toxicity and climate change *mediated by environmental emissions* mostly due to the lower weight and melting temperature of a unit of plastic versus glass (Humbert et al. 2009; Belboom et al. 2011; Amienyo et al. 2012; Gérand and Roux 2014; Dhaliwal et al. 2014). LCA of plastic waste also generally supports recycling to reduce environmental impacts and resource consumptions (Laurent et al. 2014). Human exposure research, however, indicates that plastic and recycled food packaging can be a substantial contributor to dietary intake of potential toxic chemicals (Geueke et al. 2014; Lee et al. 2014). LCA-compatible quantitative methods that consider human exposure and toxicity characterization of chemicals in food contact materials are not yet developed to inform 'sustainable' product design that considers human exposure through product use.

A criticism of including exposure to chemicals in food packaging within sustainability assessments is that such exposures should be 'safe' according to regulatory compliance, and therefore no further consideration is needed. Compliance with regulatory thresholds, however, does not offer means of identifying areas of **exposure minimization**, and furthermore does not offer consideration of **exposure in a life cycle context** e.g. where package A and package B can both comply with regulations, however one package may lead to lesser exposure than the other via environmental emissions throughout its life cycle or through use. In addition, guidance on applying regulatory safety standards are lacking for most food contact materials, such as food packaging, like paper and board where no EU-wide migration exist and therefore it is unclear if and how chemical safety of these materials is ensured, especially when it comes to use of recycled materials. A recent analysis of chemicals listed for food contact materials shows that there are at least 175 chemicals of concern legally used in food contact materials, with evidence of migration into food for some of these substances (Geueke et al. 2014; Geueke and Muncke, submitted).

In this study we explore science-based methods to inform decision making about packaging design that considers exposure to chemicals in food from packaging as a possible sustainability trade-off. Trade-offs can occur when decreased life cycle impacts based on other environmental indicators (e.g. decrease greenhouse gas emissions via transport by replacing glass with lighter weight plastic) incidentally increase potential for exposure through package use. On the other-hand trade-offs can also occur if efforts are made to decrease exposure (e.g. use of only virgin material) that can lead to increased life cycle impacts. For the first time, we present LCA-compatible methods for assessing exposure to chemicals in food packaging and we test if such exposure **is important in a life cycle context**.

2. Methods

We developed methods to analyze the relevancy of exposure to chemicals in food packaging in a life cycle context in terms of exposure magnitude and to eventually quantify exposure as a potential design trade-off. We selected high impact polystyrene (HIPS) as a packaging material to focus the analysis for data procurement. A reference flow of 8 HIPS cups of 125mL each, resulting in 0.017 of

HIPS materials used (Robertson 2012), to provide the function of containing and delivering 1kg of yogurt to consumers. Yogurt was chosen as the packaged food item because concentrations of chemicals migrating from HIPS were available through a recent food screening study by the United States Food and Drug Administration (US FDA) (Genualdi et al. 2014), and packaging dimensions were available (Robertson 2012).

To put exposure via package use (i.e. consuming packaged food) in a life cycle context, first, we characterized human toxicity potentials of the emissions resulting from various HIPS life cycle stages: HIPS material acquisition, thermoforming of HIPS into packaging, 14 day (in-home) refrigeration of the packaged food, and incineration. The life cycle inventory (LCI) from these stages were selected from ecoinvent v3.1 (Weidema et al. 2013) with default system allocation. Human exposure to life cycle emissions via environmental fate was estimated according to the ILCD methodology which relies on USEtox, where the impact (I , cases) is estimated as a function of the mass emitted (m_e , kg), the population-scale human intake fraction (iF kg taken in per kg emitted), and the EF (cases per kg taken in), $I = m_e \times iF \times EF$. The purpose of this exercise was not to perform a full LCA of HIPS food packaging, but to obtain basic screening to allow comparison to life cycle impacts according to default database values for unit processes. We are then able to identify potential human toxicity to compare for the first time exposure through consuming packaged food in a life cycle context.

The chemical inventory from life cycle stages were matched to chemicals known to occur in HIPS according to the US FDA (Genualdi et al. 2014) and the European Commission (Hoekstra et al. 2015) to identify chemicals where exposure could be tracked throughout the life cycle stages, including exposure via packaged food. Thereby, given the occurrence of a substance in HIPS and quantifiable mass transfer into food (exposing consumers), as well as occurrence as an HIPS life cycle emission and quantifiable fate and population-level exposure through the environment, we were able to explore in detail exposure magnitudes for a given chemical across the packaging life cycle. We also characterized this exposure using comparative toxicological units for humans (CTU) measured in potential disease cases, according to effect factors in USEtox, using a 1:1 route-to-route effect extrapolation for oral and inhalation exposure as suggested by Rosenbaum et al. (2011).

The novel contribution of this study is to provide a first demonstration and test of including exposure to chemicals in food packaging in a life cycle context. Thus, following the life cycle screening we characterize the mass of chemical within a package that migrates from the packaging into the food item thus exposing humans. Given weight per weight concentration (C , mg/kg) of a chemical in food, a mass of food (m_f , kg), the assumption that 100% of the food item is ingested, and effect factors (EFs , disease cases kg^{-1}) we estimated impact (I , disease cases) as $I = C \times m_f \times EF$. Specifically, for styrene we relied on empirical data on the concentrations in yogurt packaged in HIPS (Genualdi et al. 2014). When chemical concentrations were not known, we used the allowable amount migrating into food from packaging, according to EU authorities, and also quantified the resulting impacts for various percentages of this amount. When EFs were unavailable from USEtox, EFs were extrapolated from available No Observed Adverse Effect Levels (NOAELs) from animal experiments according to the methods used to derive the EFs in USEtox (Huijbregts et al. 2005). Thus we were able to include exposure to chemicals in packaging in an LCA context, and also compare to LCIA results for other life cycle stages. Finally, we screen all life cycle potential human toxicity impacts when including exposure through packaged food consumption for ‘hot spots’ to determine if exposure to chemicals in food packaging via use could potentially be a hot spot in LCAs of food packaging. Hot spots were identified as the largest contributors to HTP in the assessed life cycle processes (stages).

Chemical occurrence in packaging, transfer into food, and EFs are necessary components of our developed LCIA methodology to characterize chemicals in packaging. However, even when a

practitioner knows the chemical concentration occurring in a package, they may not know the mass transfer into various food items leading to exposure. Thereby to address such a data gap, we explore modeling exposure by various chemical-package-food combinations and aim to operationalize modeling approaches for LCIA. Chemical transfer from packaging into food was modeled as the product intake fraction metric (Jolliet et al. 2015). The product intake fraction (PiF) is analogous to the intake fraction (iF) (Bennett et al. 2002) used to relate environmental emissions to exposure in LCIA, but instead of intake per kilogram emitted (iF), PiF has units of chemical intake per kilogram of chemical initially in the product. In this way, exposure to chemicals in food packaging can be added to an LCIA characterization framework that typically only includes exposures to mass in the environment, where given the mass of a chemical in a package (m_p , kg), the PiF ($\text{kg}_{\text{intake}} \text{kg}_p$), the assumption that 100% of the food item is ingested, and effect factors (EFs , $\text{disease cases kg}^{-1}$) impact (I , disease cases) was estimated as $I = m_p \times \text{PiF} \times EF$. To model PiF we adapted a widely used regulatory model (Begley et al. 2005) for migration of chemicals from food packaging into food and applied realistic (instead of worst-case) partition and diffusion coefficients. We then explored this model for one chemical for various food and plastic combinations.

3. Results

Only one chemical, styrene (CAS 100-42-5), was identified to occur in HIPS packaging and in life cycle inventory. With over >6,500 substances known to be used in food contact materials such as food packaging (Neltner et al. 2013; Oldring et al. 2014) only 18 chemicals total were identified to definitively occur in HIPS (although likely many more occur in reality, and the exact composition will vary from package to package). To our knowledge no publically available database matches chemicals with specific packaging types, but only to e.g. ‘polymers’ in general. Out of these chemicals only the styrene monomer had effect factors (EF) available in USEtox. Furthermore, out of the 17 other chemicals only 7 were readily able to be matched to a CAS number. No Observable Adverse Effect Levels (NOAELs) were only available for 3 chemicals with CAS numbers from toxicity studies (personal communication with RIVM) which were required to estimate EF for characterization (CAS numbers 61167-58-6, 7128-64-5, 36443-68-2).

Screening life cycle exposure to styrene alone (Figure 1), the material acquisition stage was a hot spot when exposure through packaging use (i.e. through consuming food) was not considered. However, when considered, exposure to styrene through packaged food consumption (due to consuming 1 kilogram of packaged yogurt) was greater than the exposure of the entire population due to life cycle emissions of styrene related to producing packaging for 1 kilogram of yogurt. When further extending this analysis (Figure 2) to include characterization of exposure to styrene as well as the three other chemicals known to migrate into food from HIPS packaging (where concentration in yogurt and other foods was *unknown*), we found a similar result, that even when considering consumer exposure to only four chemicals migrating from HIPS into 1kg of food, at *levels at or below regulatory safety thresholds*, the human toxicity potential exceeded aggregated exposure to these chemicals from the other packaging life cycle stages. Specifically, when $\geq 1\%$ of the allowable amount to migrate from packaging into food was considered, exposure through 1kg packaged yogurt constituted more than 30% of the entire life cycle human toxicity potential when considering the chemicals included in this analysis. We did not consider food waste which would decrease exposure through consumption and increase environmental impacts relative to 1kg yogurt *consumed*.

When data were completely lacking for concentrations of chemicals in packaged food we explored preliminary migration modeling. Through modeling various food-package combinations we

corroborated previous findings that the type of food (and its fat content) as well as the type of package (and its diffusivity properties) drastically influenced results.

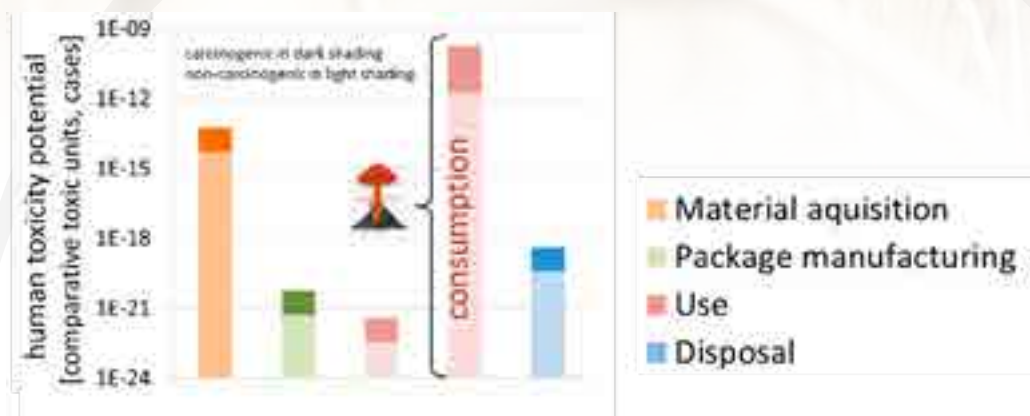


Figure 1: Human toxicity potential for life cycle exposure to styrene in food packaging (volcano is the symbol for the 'hotspot').

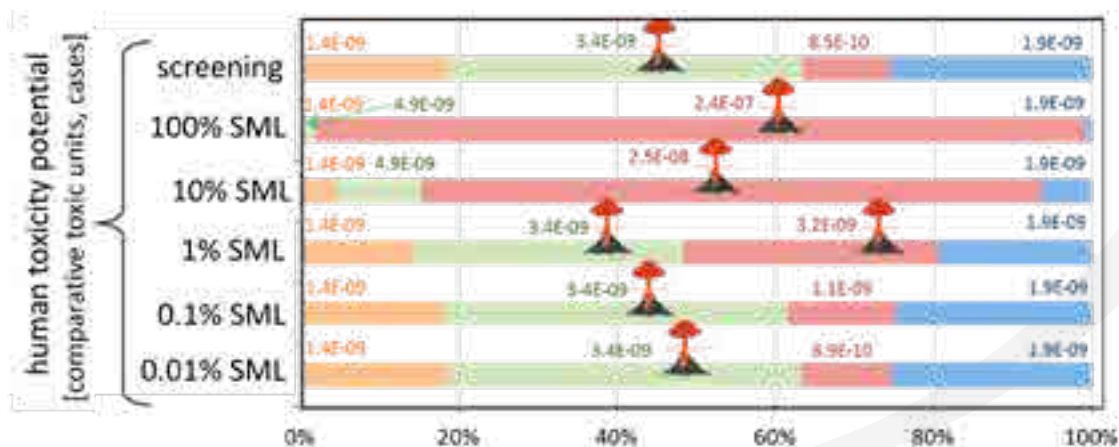


Figure 2 (same legend as Figure 1, with volcano symbols for hot spots): Use stage exposure via consuming packaged food to 5 chemicals was estimated and characterized as a function of the allowable amount to migrate from packaging into food according to regulatory specific migration limits (SML), where there is no SML established for styrene, so realistic exposure was used in all cases, but styrene contributed negligibly to use stage HTP via consumption of food.

4. Discussion

Consideration of exposure to chemicals in **food packaging** is currently missing from Cradle to Cradle and Ecolabel certification/licensing and Life Cycle Assessment (LCA), although chemical exposure via food packaging is a potential human health risk. In order to characterize chemicals in food packaging in LCIA, there are substantial obstacles. First, many chemicals occur in plastic as non-intentionally added substances (NIAS) which may not be assessed or known (Hoppe et al. 2016). Furthermore, even when substances are known to occur, data availability on chemical occurrence (e.g. frequency of occurrence, and type of packaging) in packaging is often protected by confidentiality agreements. These issues pose substantial barriers to developing an inventory of frequently occurring chemicals and their concentrations in packaging, however for a specific LCA study a practitioner may be able to gain knowledge from the assessment commissioner. Secondly, a main concern about chemicals in food packaging is the possibility of endocrine disruption as a mode of action for disease.

At this stage, it is unlikely regulatory animal tests at relatively high levels of exposure (e.g. mg/kg/day), which form the bases of both LCIA effect factors and allowable amounts in food specified by specific migration limits and other regulatory levels, cover low-level effects of endocrine disruption (which can result in carcinogenic effects and/or reproductive effects and/or other biological effects) and this contentious topic is under debate by regulators both in Europe and the United States (Muncke et al. 2014).

If the methods we develop are used in an LCA comparing virgin and recycled packaging materials, it is likely that recycled materials will demonstrate higher human toxicity potential through packaged food consumption (Biedermann and Grob 2013; Lee et al. 2014). With increasing effort towards developing a **circular economy**, especially for plastics, chemicals in materials that can be reused for food packaging is a central issue to ensuring the viability of material streams (World Economic Forum and Ellen MacArthur Foundation 2016). The methods developed in this study provide **first steps towards quantitative consideration of chemicals migrating into food from packaging in sustainability assessments**. However, it will be important that the interpretation of such possible results does not discourage recycling and resource use efficiency, but instead encourages systemic *improvement* of recycling systems for food packaging because of decreased impacts associated with recycled materials (Laurent et al. 2014). Furthermore, modeling demonstrated that the food-package combination is an important consideration. Because the food-package combination can lead to large variations in exposure to chemicals through packaging use, glass may be a more desirable package (leading to substantially less human toxicity potential through packaged food) for certain food products that have high potential for migration such as fatty foods and alcoholic beverages or products that are sterilized or pasteurized in bottles (i.e. fruit juices).

When empirical data are unavailable to estimate migration of known substances from a package into a food, modeling is a useful approach however it comes with substantial uncertainties tied to the required estimation of the diffusion coefficient and partition coefficient for the chemical from packaging into food (Begley et al. 2005). The model applied in this study is mostly used to estimate migration from packaging to liquid food items, and compiled data are not available to corroborate this model for a physicochemical space or across package-food combinations.

5. Conclusions

In this study we provide for the first time characterization of exposure to chemicals migrating from packaging into food in a life cycle context to test if this exposure pathway is important to consider in LCA. Data limitations (i.e., occurrence of chemicals in specific packaging types, their concentrations in packaging and/or packaged food, and effect factors) were a main obstacle to this exploration. Nevertheless, results demonstrate that even when characterizing only 4 chemicals migrating from packaging into food, at levels well below regulatory compliance, potential human toxicity due to food consumption was far greater than the potential human toxicity estimated for each considered life cycle stage following ILCD methodology. This implies that the use stage of food packaging (i.e. consuming the packaged food), at least for yogurt packed in HIPS, but likely for other packaging materials and food combinations, can be a human toxicity hot spot. Such a finding implies that exposure to chemicals in food packaging via food is the most important aspect to minimize potential toxicity throughout the life cycle. The results also imply that when designing eco-friendly food packaging exposure through use is an important consideration as a potential trade-off.

6. References

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170. One size does not fit all - toward farm-specific GHG mitigation options in agriculture through evaluating variability at multiple levels

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ABSTRACT

In order to create meaningful and effective mitigation options, we need to accurately evaluate the variability of GHG emissions within farming systems and their determinants (natural/biophysical context, production factors, production techniques, technologies, behavioral). The main objectives of this study are (1) to evaluate variability and amount of GHG emissions from a large number of dairy farms at multiple levels (farm, product, area) (2) to link GHG emissions to different farm- and area-specific indicators. We combined in-depth data and site-specific models for 500 dairy farms for the years 2011-2013 in Bavaria, Germany. An initial evaluation of 96 farms for the year 2013 shows a high variability of GHG emissions per kg FPCM (fat and protein corrected milk): 0.82-1.82 kg CO_{2-eq}/kg FPCM. GHG emissions from on-farm feed production contribute strongly to total GHG emissions and show a high variability. This variability can be explained mainly by the variation of (1) feed use efficiency (2) the balance between nitrogen input and nitrogen demand of plant production and (3) site-specific N₂O emissions. Evaluation of variability at farm and area level provide additional insight in potential GHG mitigation options. However, reduction in GHG emissions at farm or area level might go along with a shift in farm outputs.

Results show that evaluation of GHG variability at different levels and the link to site- and farming-specific indicators help identify GHG mitigation options for agricultural production systems. This approach is a first step towards robust and environmentally sound farming systems for specific locations.

Keywords: Variability, GHG mitigation options, dairy farms, functional units

1. Introduction

In order to create meaningful and effective mitigation options, we need to accurately evaluate the variability of greenhouse gas (GHG) emissions within farming systems and their determinants (natural/biophysical context, production factors, production techniques, technologies, behavioral). However, GHG modelling and life cycle assessment (LCA) approaches are often limited by a lack of in-depth and site-specific data for a high number of farms. Most studies that compare GHG emissions of e.g. different dairy cow production systems are based on “typical dairy farming systems” (O’Brien et al, 2012, Flysjö et al., 2012). Just a few studies are based on actual farm data. Yet even these studies are often based on relatively low number of farms (Zehetmeier et al., 2014) or a lack of in-depth and site- or farm-specific input variables (Thomassen and de Boer, 2005).

Studies evaluating GHG mitigation options of farming systems often focus on the product level as the functional unit (FU) (Salou et al., 2016). Product-based FUs enable a meaningful benchmark between different farms. This benchmark helps to identify the determinants of different GHG emissions. However, it is difficult to allocate of GHG emissions to different farm outputs. This is especially the case for dairy farms with milk as the main product and beef as a co-product. Several studies have addressed the implication of this issue (de Vries et al., 2015, Zehetmeier et al., 2014). Many studies derived from economic disciplines use the farm level as a FU to investigate the impact of carbon taxes or policy regulations (Flugge et al., 2005) on GHG emissions, whole farm income and

shifts in agricultural output at farm level (Thamo et al., 2013). The farm level also provides insight in GHG emission flows and sources of all production processes and the contribution of each production branch on total farm GHG emissions. Area based approaches can help identify site-specific GHG mitigation options and assess trade-offs between numerous functions of cultivated land as food production, preservation of biodiversity, carbon storage, or bioenergy production (Salou et al., 2016). Thus, the evaluation of variability at different levels (farm level, product level, area level) can help identify additional mitigation options and trade-offs between GHG mitigation, profitability and agricultural production.

The main objectives of this study are (1) to evaluate variability and amount of GHG emissions from a large number of dairy farms at multiple levels (farm, product, area) (2) to link GHG emissions to different farm- and site-specific indicators.

2. Methods

As a case study we evaluated mixed (cash crop and dairy) and specialized dairy farms from Bavaria, Germany.

Data sources

We linked different data sources already available on farm such as data from the farm accounting network (FADN), milk recording, and the integrated administration and control system. Additional farm management data such as the amount and type of mineral and organic fertilizer application, amount and type of on-farm and bought in feed were taken from farms participating in discussion groups (Dorfner et al., 2013). The database of discussion group farms contains over 500 farms for the years 2011-2013 in Bavaria, Germany. In this study we conducted an initial evaluation of 96 farms for the year 2013.

GHG modelling

Modelling of GHG emissions was conducted based on an LCA approach. The global warming potential of the different GHG during a time horizon of 100 years was calculated based on IPCC (2013). We accounted for emissions from production of all farm inputs (mineral fertilizer, purchased feed, purchased animals) except emissions from the production of buildings and machinery. Upstream emissions of purchased feed were calculated based on the FeedPrint model (Vellinga et al., 2012). Emissions for all other inputs were taken from the Ecoinvent database (Ecoinvent, 2013). Emissions from enteric fermentation, manure storage and management were calculated based on the German national inventory model GAS-EM (Haenel et al., 2014). Emission modelling from on-farm feed production and cash crop production was based on German national inventory data (Haenel et al., 2014). To assess N₂O emissions from nitrogen input into the soil, however, we used a novel and site-specific approach. We linked the bio-physical model of Dechow and Freibauer (2011) with data from the integrated administration and control system for each farm. Thus, N₂O emissions from nitrogen input into the soil were based on field specific N₂O emission factors. For a detailed description of the bio-physical model compare Dechow and Freibauer (2011).

Evaluation levels and functional units

We explored variability of GHG emissions at different evaluation levels (farm, product, area). The corresponding FUs were: (1) One dairy farm for a one-year period in 2013 for the farm level. (2) One kg of fat and protein corrected milk (FPCM) for the product level. The definition of a production

system was based on one dairy cow and associated replacement heifers. The number of replacement heifers was calculated based on replacement rate of the dairy farm. We did not allocate GHG emissions between milk and co-products. Instead we calculated the potential beef output per kg FPCM for each dairy cow production system to assess trade-offs between milk and beef output. Potential beef output includes beef from culled dairy cows and beef from fattening of calves in bull and heifer fattening systems not needed for replacement. To benchmark dairy cow production systems with different beef output per kg FPCM we added GHG emissions from suckler cow beef production systems to dairy systems with lower beef output per kg FPCM. Values and data to calculate potential beef output for dual purpose and specialized milk breed dairy cows were taken from Zehetmeier et al. (2014). (3) One ha of on-farm land was defined as the FU for the area based evaluation level.

A summary of descriptive statistics for dairy farm characteristics is provided in Table 1. The sample of farms shows a high variation in production traits such as milk yield (4461 – 10264 kg FPCM/cow) and replacement rate. As we linked the GHG model to a site-specific N₂O model, the N₂O emission factor for nitrogen input into the soils varies between fields within investigated farms. Table 1 shows the average on-farm N₂O emission factor and its variation between evaluated dairy farms.

Table 1: Farm characteristics for analysed dairy farms (N=96)

Item	Unit	Mean	Sd	CV	Min	Max
Number of dairy cows per farm	-	79	30	37.73	13	172
Arable land	ha/farm	56	34	60.60	12	207
Permanent grassland	ha/farm	33	15	44.66	4.59	83
Area related to dairy enterprise feed production in relation to total farm area	%	66.27	18.78	28.34	27.07	100
Milk yield	kg FPCM per cow	7898	875	11.08	4461	10264
Replacement rate	%	23.86	6.14	25.75	11.00	45.00
Age of first calving	months	28	1	05.01	24	32
Calving interval	days	385	17	04.35	359	438
MJ NEL intake per dairy cow in relation to MJ NEL requirement ¹⁾		1.04	0.08	07.47	0.92	1.29
Dairy cow breed:						
Fleckvieh		90				
Holstein-Friesian		6				
N ₂ O emission factor ²⁾		0.0103	0.0042	40.45	0.0052	0.0200
Weighted nitrogen balance ³⁾		77	69	89.09	-91	294
Profit ⁴⁾	cent/kg FPCM	10	6	56.39	0.32	46.56
	€/farm	101731	64536	63.44	-21410	313878

¹⁾ NEL = Net Energy Lactation, ²⁾ field specific emission factor based on the model of Dechow and Freibauer (2011), ³⁾ Nitrogen input per ha minus nitrogen demand by plant (average per farm, weighted by crop area), ⁴⁾ Farm profits include all economic returns minus variable and fixed costs (not included: imputed labour and land cost and direct farm payments), FPCM = fat and protein corrected milk yield, sd = standard deviation, CV = coefficient of variation

Statistical analysis

Descriptive statistics and regression analysis were assessed at the farm and area level using the Rcrane statistics software (Rcrane, 2016). At the product level a dominance analysis was undertaken to explain the impact of single predictor variables on the variation of GHG emissions per kg FPCM. In the case of dominance analysis “one predictor is more important than another if it is selected over another in all possible subset models where only one predictor of the pair is to be entered” (Azen and Budescu, 2003). Dominance analysis provides a meaningful decomposition of the total predicted variance in the criterion variable. This is also true in the case of multicollinearity of predictor variables. Dominance analysis was implemented in this study using the statistical programme R (Equation 1).

$$Equation (1) LMG(x_k) = \frac{1}{p} * \sum_{i=0}^{p-1} \left(\sum_{\substack{S \subseteq \{x_1, \dots, x_p\} \setminus \{x_k\} \\ n(S)=i}} \frac{seqR^2(\{x_k\} | S)}{\binom{p-1}{i}} \right)$$

where $LMG(x_k)$ equals the average over model sizes i of average improvements in R^2 when adding regressor x_k to a model of size i without x_k , $seqR^2(\{x_k\} | S)$ equals = additional R^2 when adding x_k to a model with the regressors in set S (Groemping, 2006)

3. Results

Farm Level

Figure 1 shows a trend of increasing total GHG emissions with increasing milk output per farm. However, variability between farms points at potential mitigation options. Some farms show relatively high GHG emissions from on-farm feed production (up to 44% of their total GHG emissions). These farms are often characterized by a high N_2O emission factor from nitrogen input into the soil. The cultivation of forage plants with a low nitrogen demand could be an efficient GHG mitigation option on these farms. The contribution of single GHG emission sources on total farm emissions provides insight in most important emission sources and hot spots (Figure 1). These insights can be discussed with single farmers and farm advisers and thus be helpful to identify farm-specific GHG mitigation options. For example the contribution of CH_4 emissions from manure storage varies between 6 to 17% of total farm GHG emissions within the group of evaluated farms. High GHG emissions of this source can be addressed with technical mitigation options such as the implementation of anaerobic digesters.

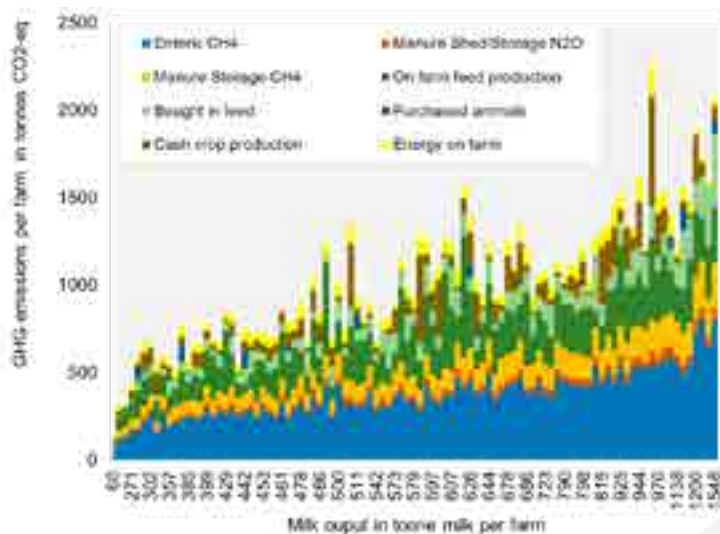


Figure 1: Total GHG emissions per farm ordered by milk output per farm

Linear regression analysis showed a significant correlation ($p < 0.001$, $R^2 = 0.3213$) between total GHG emissions per farm and farm profit (Figure 2). Total GHG emission also show an increasing trend with increasing size of cultivated farm area. There is a number of farms with similar profit (e.g. 90 thousand € per farm) and similar cultivated farm area size but a relatively high variation of GHG emissions (600 – 1000 tonnes CO₂-eq). A more detailed analysis of these farms can point at low cost GHG mitigation options.

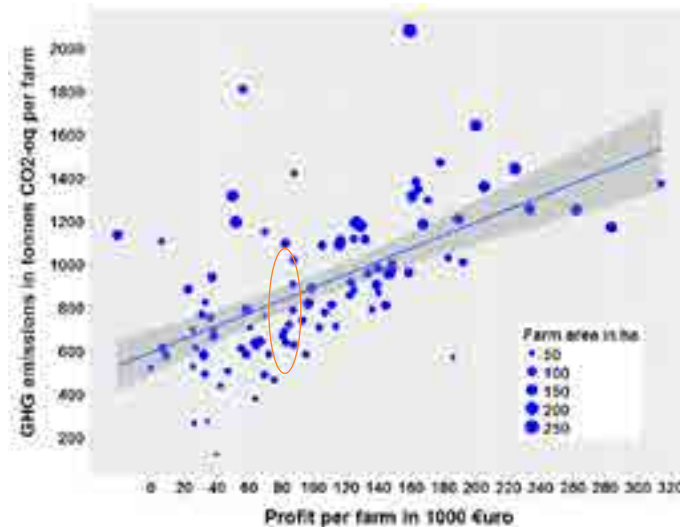


Figure 2: GHG emissions per farm in relation to profit per farm

Product Level

The product level allows a benchmark between the dairy cow branch of the evaluated dairy farms. Figure 3 shows the boxplot of the most important GHG emission sources per kg FPCM. Total GHG emissions vary between 0.82 and 1.82 kg CO₂-eq/kg FPCM. The variation is primarily due to the variation of GHG emissions from feed production on the farm. The GHG emissions from manure storage show a relatively low variation. This can be explained by the lack of farm-specific data on the type of manure storage system. Thus, the most common slurry storage system (slurry tank with natural crust) was assumed as the default system (Haenel et al., 2014). We used dominance analysis to identify those variables that contribute most to the variability of total GHG emissions per kg FPCM. The five variables included in the dominance analysis explained 70% of total variation of GHG emissions per kg FPCM. Thereof 29% of variation could be explained by differences in feed intake efficiency of dairy cows. We calculated the ratio between energy intake in MJ net energy lactation (NEL) to energy requirement in MJ NEL as an indicator for feed intake efficiency. Compared with other variables listed in Table 1 the feed intake efficiency has a relatively low coefficient of variation. However, as this indicator has a direct impact on CH₄ emissions from enteric fermentation and the amount of feed to be produced its total contribution to GHG emission variability is relatively high. Another efficiency indicator: nitrogen demand by plants minus nitrogen input per ha, has a high impact (25%) on explained variation of GHG emissions per kg FPCM. The site-specific N₂O emission factor determines 16% of explained variation in GHG emissions. Thus, the amount of nitrogen application on plots with a relatively high emission factor can have an important impact on total GHG emissions.

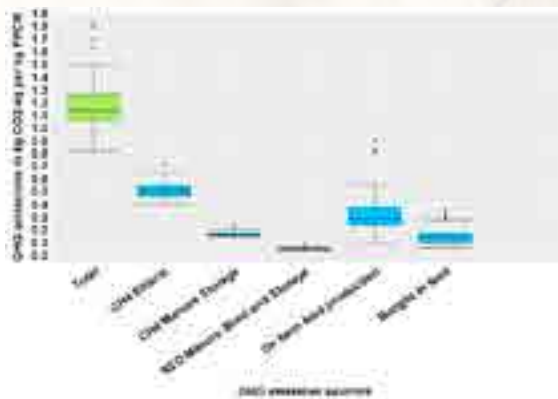


Figure 3: Boxplots of GHG emission sources per kg FPCM

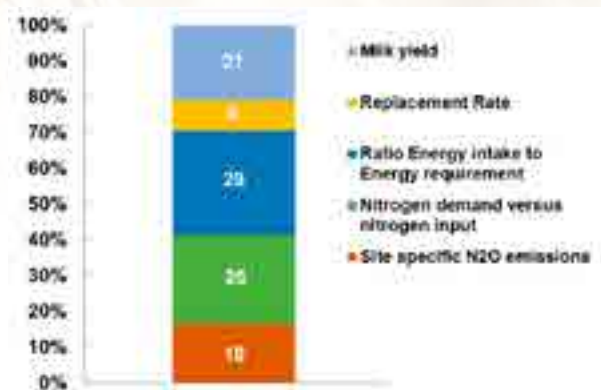


Figure 4: Decomposition of explained variance ($R^2 = 70\%$) in GHG emissions per kg FPCM, compare equation 1

The variation in milk yield explains 21% of the explained variation in GHG emissions per kg FPCM. Several studies point at the negative impact of increasing milk yield on the amount of beef output from dairy cow production systems. Thus, we explored also the variation of GHG emissions per kg FPCM while keeping beef output per kg FPCM constant (Figure 5). The farm with the highest potential beef output per kg FPCM serves as benchmark (88 kg beef per kg FPCM). The difference in potential beef output of the other farms is filled by beef from suckler cow production systems. Figure 5 shows that total GHG emissions do not decrease with increasing milk yield when keeping beef output constant.

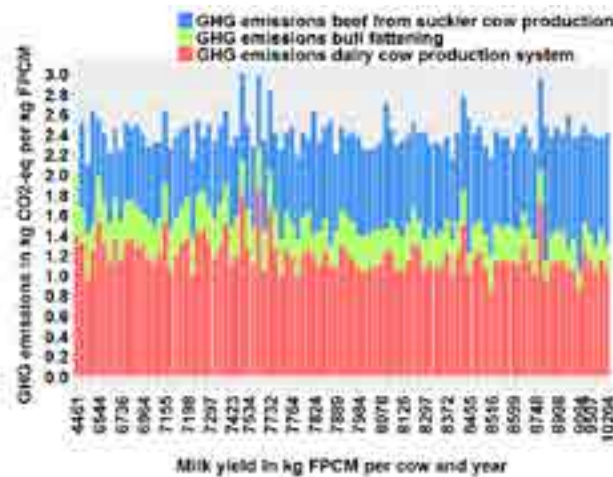


Figure 5: GHG emissions per kg FPCM assuming a constant amount of beef output per kg FPCM

Area level

At the area level we analysed the relationship between GHG emissions, livestock density and profitability. A simple linear regression analysis showed that livestock density expressed in livestock units (LU) per ha cultivated farm area is a main determinant of GHG emissions per ha (Figure 6). Variation in GHG emissions per ha increases with higher LUs per ha. The regression analysis between profit per ha farm area and LU per ha showed a significant relationship but a lower R^2 (0.33).

Combining the relationships of Figure 6 and Figure 7 provides insight in the reduction of profit and GHG emissions when decreasing LU per ha. Within the group of evaluated farms, a decrease in LU per ha results in GHG abatement costs of 130 €/t CO₂-eq (decrease in profit divided by decrease in GHG emissions).

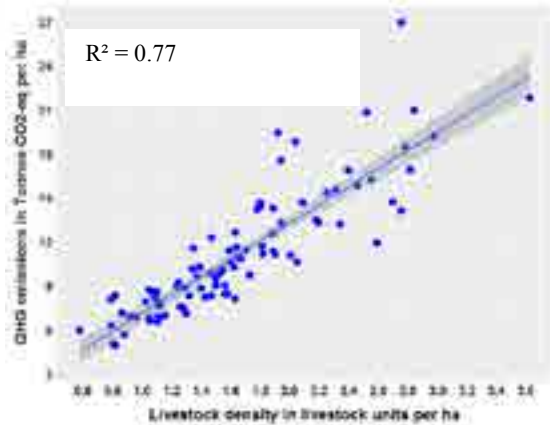


Figure 6: GHG emissions per ha cultivated farm area in relation to livestock density per ha cultivated farm area

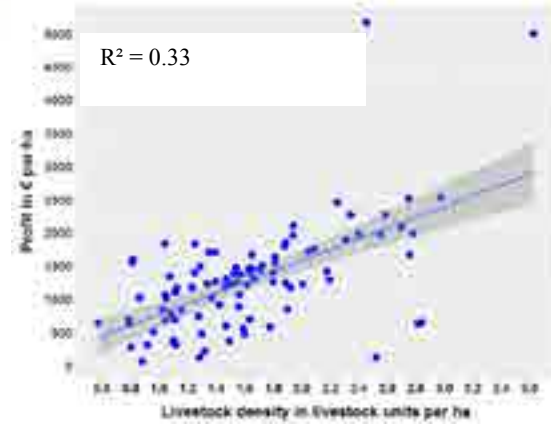


Figure 7: Profit per ha cultivated farm area in relation to livestock density per ha cultivated farm area

4. Discussion

Variability of GHG emissions at product level

Our analysis of a group of 96 dairy farms showed that feed and nitrogen management efficiency are the main indicators explaining variability of GHG emissions per kg of FPCM and are thus the main levers to reduce GHG emissions. Other studies exploring variability of GHG emissions per kg milk identify milk yield as the most important variable. Christie et al. (2012) showed that milk yield per cow explained 70% of the variance in GHG emissions/kg of milk of 41 Australian dairy farms. In the study of Zehetmeier et al. (2014) milk yield contributed 55% to the explained variation of GHG emissions per kg FPCM within a group of 27 dual purpose dairy farms. The lower impact of milk yield in our study highlights that a higher availability of detailed farm data for a higher number of farms can reveal additional and even more important GHG mitigation options in dairy cow production systems. Although a large amount of detailed data for actual farms was available to our study, we would like to emphasise that particularly data on feed intake and nitrogen management have a high epistemic uncertainty. Data on feed intake and crop specific amounts of nitrogen application were derived from book recording and data collection from farm advisers. Future research that helps to improve standardized data collection or measurement methods can improve benchmark of GHG analysis. A limitation of our study is the lack of variation in technical GHG mitigation options between farms in manure management. Studies exploring GHG mitigation costs (Moran et al., 2011) show that technical GHG mitigation options such as application of anaerobic digesters are often more cost intensive than farm management options such as improved feed efficiency and nutrient management. Studies on the relative contribution of farm management in comparison to technical mitigation options for a high number of actual farms would provide important insight in cost effective GHG mitigation options for different farms.

Farm and area based evaluation of GHG emissions

Analysis at farm level provide insight in whole farm emission sources and hot spots. CH₄ emissions from enteric fermentation had the highest contribution to total GHG emissions for most farms.

However, the contribution of other sources such as CH₄ emissions from manure management or GHG emissions from on-farm feed production varied highly between farms. The analysis of individual hot spots is especially important for bi-directional knowledge transfer between farmers and advisers. Another advantage of farm level analysis is discussed in the study of Marton et al. (2016). The authors conclude that only a farm approach was able to explore the benefits of mixed farming systems. Marton et al. (2016) used different approaches to ensure comparability between farms when exploring environmental impact at farm level. Our study gives only a first rough insight in whole farm emissions and relations to product output and farm size. Future research might explore suitable methods to enable a benchmark of GHG emissions using the farm level as FU.

Area based FUs are less frequent in GHG mitigation literature compared to product based FUs (Salou et al., 2016) but are recommended for future research as land is one of the most limiting factors in agricultural systems and sustainable land occupation is, besides food production, a main function of agriculture (Salou et al., 2016). Flugge et al. (2005) explored the substitute from livestock to crop production for two different farming systems. The authors emphasized that the opportunity costs of substituting from livestock to crop production are the main critical factor for GHG abatement costs. In our study we calculated average GHG abatement costs of 130 €/t CO₂eq when reducing livestock density per ha farm land. This value is highly sensitive to prices such as milk price, grain prices, land prices, cattle prices and their relations. Abatement costs of reducing livestock density per ha is also higher for grass based systems in comparison to dairy systems based on arable land due to lack of production opportunities (Flugge et al., 2005). The reduction in livestock density per ha especially in areas with a high concentration of livestock density can improve additional environmental issues such as ammonia emissions or high nitrate contents in water (Salou et al., 2016). Thus, abatement costs can be allocated to different environmental issues. Similar to the product based FU the consideration of co-products needs to be explored for the farm and area level in future studies to ensure comparability between agricultural systems with multiple outputs. In our study we only conducted a multi-criteria analysis at the product level. Multi-criteria analysis at the farm and area level will provide further insight in determinants of GHG emissions of these levels.

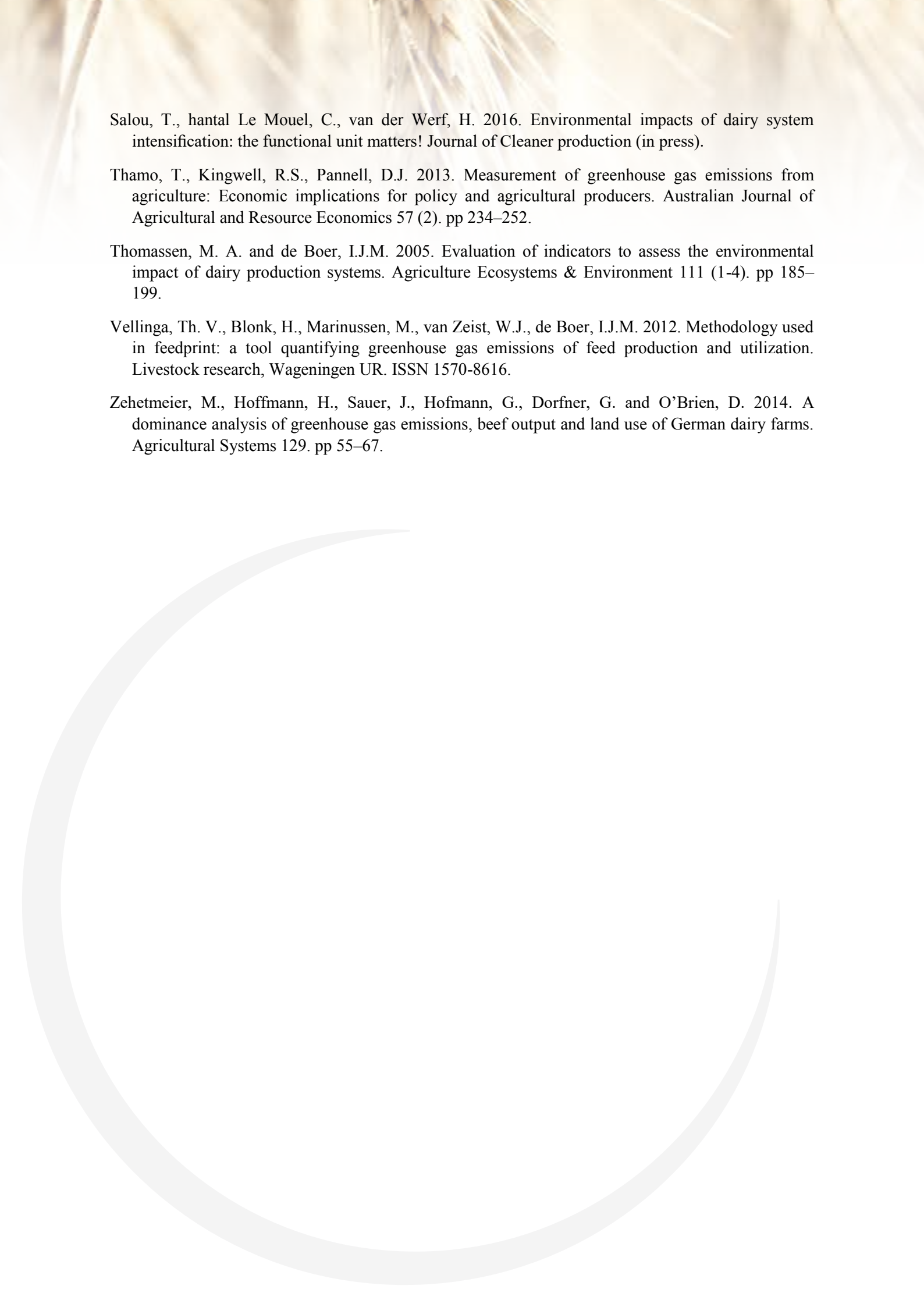
5. Conclusions

We analyzed variability of GHG emissions from 96 mixed and specialized dairy farms at different levels (farm level, product level, area level). The availability of in-depth and site-specific on-farm data showed that feed and nitrogen use efficiency play the most important role explaining variability of GHG emissions at product level. Improvement of these indicators is usually associated with an improvement in profitability and a reduction in GHG emissions. Thus, specific approaches as on-farm advisory service might be needed to realize these mitigation options. Site-specific N₂O emission factors had a high contribution in explaining variability of GHG emissions. The single farmer has only limited impact on this factor as it is mainly affected by climate and soil related indicators. However, special attention needs to be paid on the amount of nitrogen application in areas with high emission factors.

The evaluation of GHG emissions at farm and area level provided insight in trade-offs between profitability, agriculture production and GHG emissions. A detailed analysis of farms that minimize these trade-offs can help to explore additional GHG mitigation options compared to product based analysis. We conclude that analyzing variability of GHG emissions at multiple levels can help to provide a more balanced and farm-/site-specific assessment of GHG mitigation options.

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156. Understanding regional and country-specific differences in environmental impacts of production systems using a globally consistent modelling approach

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ABSTRACT

This study aims at analyzing regional differences between life cycle assessment data using a globally consistent bottom-up mass-flow model. The model allows to assess environmental impacts of both livestock and crop production systems in all countries which are reporting to FAOSTAT. Flexible methodical assumptions such as different functional units, allocation methods or cut-off criteria, can be applied consistently across all livestock and crop activities as well as countries. The study identifies different main driving factors for environmental resource efficiency, depending on the impact category. The model presented in this paper can be used as a global life cycle database for a large variety of crops and livestock activities in all countries of the world. However, the data quality provided for some variables, especially in smaller and developing countries is weak. Therefore, our results cannot currently substitute the calculation of specific LCAs but rather serve as a basis for further detailed calculations. Due to the consistent global approach for calculating mass-flows product-specific impacts can be modelled in a harmonized way, which ensures that non-linearities due to different soil conditions, land availability and food-feed competitions can be taken into account. This allows to go beyond a purely attributional LCA perspective and even to assess production systems for future scenarios, e.g. with respect to human diets.

Keywords: mass-flow model, global scenarios, human diets, FAOSTAT, database

1. Introduction

Life cycle assessment has become the dominant approach for assessing environmental efficiency both in scientific and in business context. However, assessing impacts of agricultural production systems requires a large amount of inventory data which are usually not available. Existing global databases often compile datasets from various studies using different assumptions and methods. This makes LCA results hardly comparable across studies. Particularly due to the rising demand for LCA data in a business-related context, those figures which have been modelled with different assumptions are often taken as a basis for comparisons of production systems and ultimately for decision-making without sufficient reflection on the underlying assumptions. Current initiatives that aim at harmonizing methods and assumptions internationally are still in their infancy (Del Borghi, 2013; Ingwersen und Stevenson, 2012; Ledgard *et al.*, 2014).

Considering the large number of combinations of geographical origins and production systems, which can result in different environmental performances of products, a comprehensive coverage of products based on individual detailed studies seems to be unrealistic. Therefore, the question arises whether simplifications and deductions from existing global data could result in meaningful results. For crop products, a statistical approach extrapolating from existing life cycle inventories has been developed by Roches *et al.* (2010). Another approach for solving this problem is to develop a comprehensive food system model (Schader *et al.*, 2015). This study aims at exploring whether such a food system model approach is able to deliver meaningful results analyzing regional differences of products using a globally consistent bottom-up approach.

2. Methods

Model overview

The model which we used in this study has been developed based on Schader *et al.* (2015) and allows to assess environmental impacts of both livestock and crop production systems in all countries which report to FAOSTAT (FAOSTAT, 2013a). The description which follows is taken from to large parts Schader *et al.* (2015), in order to ease the understanding of the model for readers of this paper. The model is a mass-flow model which covers physical flows in the food system and processing stages of the primary products and human nutrition as reported in the FAO Food Balance Sheets and Commodity Trees (FAOSTAT, 2013b). Further specification are specifically made for grasslands, e.g. via calculating protein and energy production based on the actual net primary production (NPP_{act}) data by Erb *et al.* (2007). The limited disaggregation of livestock activities in FAOSTAT is amended by introducing herd structure models based on a cross-entropy approach based on Leon *et al.* (1999). Flows between crop and livestock production activities (Figure 1) are linked via feedstuffs and organic fertilizers. Additional data covers characteristics of inputs, processes, outputs and losses, such as nutrient, energy, protein contents (Schader *et al.*, 2015).

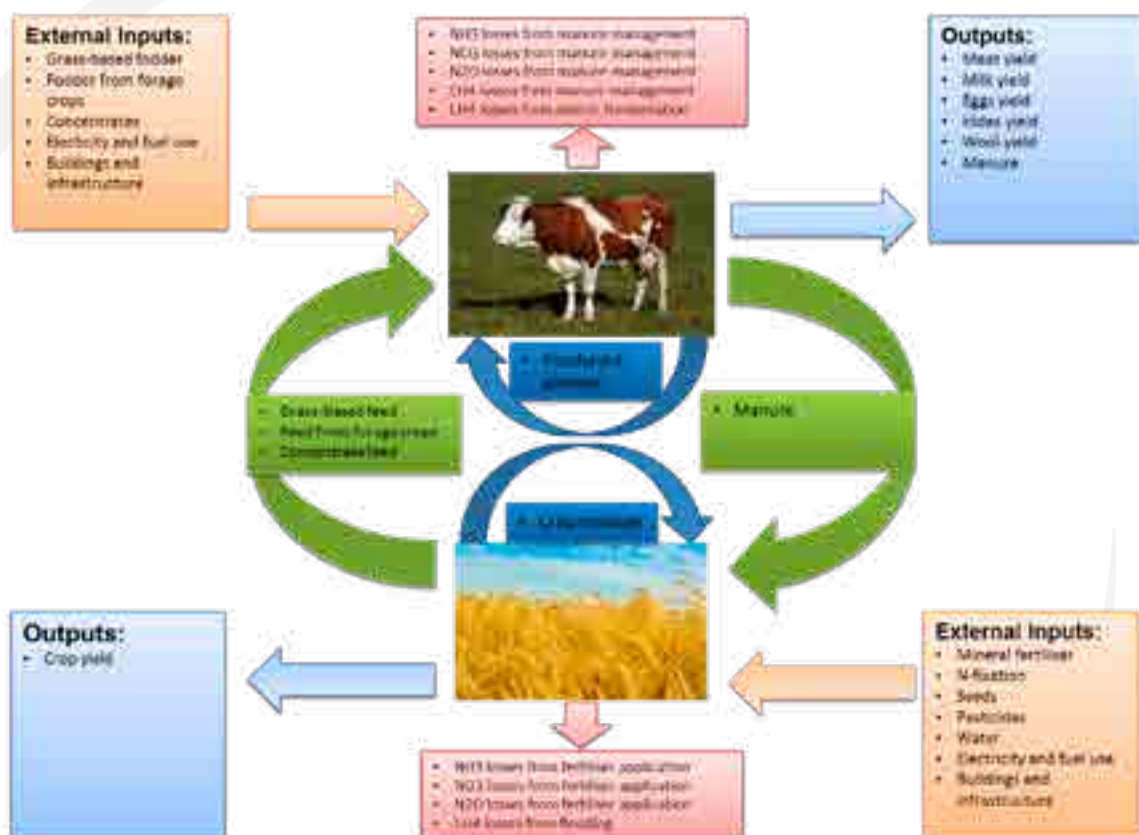


Figure 1: Overview of physical flows in the model which have been used for specifying the life cycle inventories for livestock and crop activities (Schader *et al.*, 2015).

For each activity, we defined inputs and outputs, i.e. all physical flows related to individual activities. Inputs for livestock activities include four categories of livestock feeds: a) fodder crops grown on arable land, i.e. land being cropped or fallow, b) concentrate feed derived from human-edible food (e.g. grains, pulses) grown on arable land, c) grassland-based fodder and d) fodder from agricultural/agri-industrial by-products. While a) and b) are in competition with production of human-edible food, c) and d) are not. The term grasslands is used synonymously with the term grazing land. Further inputs for livestock activities are energy input for buildings, in-stall processes and fences. Outputs of animal production activities include human-edible and human-inedible products, manure excretion, nutrient losses and GHG emissions due to enteric fermentation and manure management (CH₄, N₂O, NO₃ and NH₃). Country-specific data for amounts of concentrate feed and by-products used are derived from FAOSTAT food balance sheets. Inputs for plant production activities included arable or grassland areas, mineral fertilizers, manure, crop residues, symbiotic nitrogen fixation, herbicides, fungicides, insecticides and management practices. Outputs from plant production activities include crop yield quantities, crop residues and nitrogen losses during fertilizer application. Based on these data, we calculated livestock feed and fertilizer supply/demand balances at national, regional and global level. The main model outputs are food availability (Equation 1).

$$FA_{i,m} = \sum_{j,k} AL_{i,j,k} * OUT_{i,j,k,l=yields,s=mass} * NCHC_{i,j,k,m} * UF_{i,j,k,n=food} \quad \forall i,m \quad (1)$$

Where *i* is the index of geographic units, *j* is the index of activities, *k* is the index of farming systems, *l* is the index of inputs and outputs, *m* is the index of nutrients for human consumption, *n* is the index of utilization types (food, feed, seed, waste, other) and *s* is the index of units of inputs and outputs. FA is the food availability expressed in kcal or g protein, AL is the activity level [ha/year for land use activities, number of animals/year for livestock activities], OUT is the output [kg/ha or kg/animal], NCHC is the nutrient contents for human consumption [%] and UF is the utilization factor [%].

Modelling environmental impacts

Environmental impacts are aggregated across all geographic units, activities and farming systems (Equation 2). Activity levels (AL_{*i,j,k*}) are multiplied by inputs (IN_{*i,j,k,l,s,o*}) and the impact factors of the inputs (IF_{*i,j,k,l,s,o*}).

$$EI_{i,o} = \sum_{j,k} AL_{i,j,k} * (IN_{i,j,k,l,s,o} + OUT_{i,j,k,l,s,o}) * IF_{i,j,k,l,s,o} \quad \forall i,o \quad (2)$$

Where, EI is an environmental impact, *o* is the index of environmental impacts, IN = inputs [kg or ha] and IF = impact factors [environmental impact / kg of input or output/emission]. An overview of the environmental indicators used in this study and their units are given in Table 1. Further methodological details on the main indicators and more detailed results on the other indicators are provided in (Schader *et al.*, 2015; SI, Section 1.3.10).

Land occupation

This indicator measures how much land is necessary for agricultural production each year. Because arable land is much scarcer and more valuable than permanent grasslands for food production, we

differentiate between land occupation of arable land and grassland. For Equation 2, the inputs (IN) that are taken into account are grassland and arable land. For all arable crops and grasslands the impact factor (IF) is defined as one. This indicator combines values for areas harvested with values for cropping intensities that indicate how often, on average, a hectare is harvested per year. On average, cropping intensity is less than one; therefore, land occupation is larger than the values for areas harvested [6, 7].

N-surplus

NO₃ losses to soil, and NH₃ and N₂O losses to the atmosphere occur as a result of N use in agricultural systems. Consequently, sensitive terrestrial and aquatic ecosystems are adversely affected.

N-surplus is defined as the difference between the N content of outputs (e.g. yields) and inputs (e.g. fertilizer quantities) for each country and activity. Changes in cropping areas, animal numbers (manure), production quantities, mineral fertilizer use and N-fixation thus potentially lead to changes in N-surplus. Based on Equation 2, the amount of N is calculated by multiplying the mass of an input (IN) or output (OUT) by its N content. Relevant inputs for calculating the N-surplus are: mineral N fertilizers, N-fixation, organic fertilizer, crop residues and seeds. Relevant outputs are yields and crop residues. IF is defined as the N-content of the inputs, while all outputs are defined as negative values. As a basis for calculating GHG emissions, N-losses during fertilizer application are separated according to the type of fertilizer (mineral, manure, crop residues) and the substance emitted (NH₃, NO₃, N₂O). Model factors are specified according to IPCC 2006 Guidelines (Tier 1). Model calculations for the total N-balance in the base year are in line with literature values reported for different sources and the overall balance [1, 31, 32]. We did not include estimates of atmospheric nitrogen deposition in the N-surplus calculations.

Greenhouse gas emissions

GHG emissions of the agricultural sector have been estimated by several projects at regional [28] or global level [33-36]. Estimations of global GHG emissions of the agricultural sector are between 4.2 and 5.2 Gt CO₂-eq [20] and this constitutes approximately 10-12% of total global emissions.

GHG emissions were modelled according to the Global Warming Potential (GWP) “IPCC 2006 100a” Tier 1 methodology [37]. For enteric fermentation modelling, we used the Tier 2 methodology in order to capture the impacts of different feeding regimes on GHG emissions. Additionally, the GWP due to the production of inputs from non-agricultural sectors (mineral fertilizers and pesticides) was included in calculations according to LCA studies [38, 39], the ecoinvent 2.0 database and [40]. To calculate the GHG emissions from processes and buildings, the CED-values for different processes were taken from ecoinvent 2.0 and transformed into GWP values with process-specific conversion factors derived from ecoinvent 2.0. Emissions from deforestation and from organic soils under agricultural use were taken directly from [41]. According to Equation 2, all relevant inputs (e.g. fertilizers) and processes (e.g. enteric fermentation) were specified in physical quantities. The respective CO₂-eq values of CO₂, CH₄ (25) and N₂O (298) were used as IF, as suggested in the IPCC 2006 Guidelines. Restricting the analysis to the common emission categories, total GHG emissions calculated for the base year in our model are similar to [16, 41]. These references only differ substantially in terms of enteric fermentation calculations; the results of our model are similar to [41].

Annual deforestation potential

Because agricultural land is scarce and natural grasslands are generally not well-suited for cultivation (water or temperature limited), increasing the amount of land needed for agricultural production increases pressure on grasslands and forests [42]. Conversion of grassland to cropland may also indirectly lead to increased deforestation, due to displacement effects that result in the conversion of forests to meadows and pastures [43, 44]. With limited data available, we have assumed that additional cropland generally increases pressure on forests and may lead to increased deforestation. Following [45], we have attributed 80% of deforestation to agriculture. Following [7], we have forecast constant grassland areas.

The deforestation potential of agricultural land expansion was estimated from the average annual growth in agricultural area and the average annual deforestation rates in each country from 2005-2009 (taken from FAOSTAT). Deforestation rates in the scenarios were calculated by multiplying the change in land areas in each scenario by the ratio of deforestation areas over agricultural land area expansion, scaled by a factor of 0.8 to account for the 80% of deforestation attributed to agriculture.

In cases where no change in agricultural land area was reported for the years 2005-2009, deforestation values were calculated using the *total* agricultural area (instead of the change in agricultural area) as a proxy for the pressure of agriculture on forests. In these cases, deforestation rates were calculated by multiplying the total agricultural land area by the ratio of deforestation areas over total agricultural land area, scaled by the factor 0.8. The indicators for deforestation were applied only in cases of positive deforestation rates. Deforestation was set to zero in countries where total forest area increased.

Other indicators

Here, we provide short descriptions only, further details can be found in (Schader *et al.*, 2015; SI, Section 1.3.9). *P-surplus* is calculated analogously to the N-surplus. All P-flows are expressed as P₂O₅. No differentiation between types of P-losses is made. Therefore, the balance (inputs – outputs) calculated expresses a “loss potential”, acknowledging that large quantities of P are fixed in soils. The total P-balance in the base year as calculated in our model is in line with literature values reported in [31]. *Non-renewable energy use* is calculated according to the life cycle impact assessment methodology, “cumulative energy demand” (CED) [40]. Only the non-renewable energy categories (fossil and nuclear energy) are used and renewable energy components are disregarded. Inventory data for each activity were taken from the ecoinvent 2.0 database and [41-44]. *Water use* was derived based on AQUASTAT [46] data for irrigation use per ton of irrigated production and data on irrigated areas for various crops and crop categories covered in [12]. As there is no consistent dataset on *pesticide use* covering different countries, we developed an impact assessment model for assessing pesticide use incorporating three factors: pesticide use intensity per crop and farming system, pesticide legislation in a country, and access to pesticides by farmers in a country. *Soil erosion potentials* were derived based on an assessment of soil erosion susceptibility per crop and soil erosion rates per country (literature review and expert judgements, details in (Schader *et al.*, 2015).

Analyzing differences between countries

For exploring the ability of the model to function as a “Tier 1” life cycle assessment database, we systematically differentiate between a) inputs, outputs and losses which represent a specific life cycle inventory for each activity in each country and b) impact factors of these components of the life cycle inventory. This life cycle assessment perspective is currently linked to agricultural activities and their inputs, while processing, distribution and retailing are not considered. Furthermore, quantities for some inputs, no global dataset of sufficient quality is available. Thus, eco-toxicity and human toxicity,

which are substantially driven by pesticide input, cannot be covered with usual life cycle impact assessment methods in our approach. Also energy and fuel use could only be roughly extrapolated based on ecoinvent 2.0 inventories (Nemecek und Kägi, 2007).

For exploring the ability of the model to differentiate environmental impacts of agricultural products of different origin, we used products which are produced and are modelled with a country-specific inventory in a large number of countries as examples. Apples (n=92), beans (n=96), tomatoes (n=165), soybeans (n=91), onions (n=50), maize (n=158) and wheat (n=124) have been selected for this purpose. This explicitly excludes especially many small islands and some larger developing countries for which the quality of data reported to FAOSTAT is inconsistent according to plausibility checks with our mass flow model. Calculations for this study include the impact categories Cumulative Energy Demand [MJ CED], Global Warming Potential (GWP) [CO₂-eq], Land Occupation [hectares], N-Surplus [kg N], P-Surplus [kg P₂O₅] and Water Use [Liters of Water] for the functional units, “kg of production”, “area of production”, “number of animals”, “kcal energy for human nutrition” and “g protein for human nutrition”. Allocation procedures that have been used were: a) no allocation, b) mass allocation, c) economic allocation (based on unit values reported in TRADESTAT, averaged between 2005 and 2009).

The data quality and the relevance of it for inventory data was analyzed in order to estimate the degree of uncertainty. Data quality was judged as “poor” if country-specific data is only available for very few countries, as “medium” if country-specific data can be modelled but substantial differences from the (unknown) real country averages are likely for some countries, and as “good”, if the data is country specifically available and mostly plausible. The relevance of the inventory flow was judged as “high”, if it is a flow which can have a substantial impact on an activity, as “medium” if the flow contributes to the total impact usually less than 15%, and as “low” if the contribution of that flow is usually lower than 5% of total impact. Naturally, these judgements can only give a rough indication as the variability of the importance of the inventory flows can be strongly different, depending on the region and on the crop.

3. Results

Table 1 provides an overview of inventory flows and processes considered in the model, their relevance for different impact categories and the judgement of data quality. It shows land occupation can be modelled with highest precision as there are few relevant inventory flows and all of them are contained within FAOSTAT with an at least reasonable data quality. It needs to be noted that especially for grasslands and for livestock products (not displayed) there is a higher degree of uncertainty due to uncertainties in feed intake and feed origin. The higher degree of uncertainty for livestock products applies to all impact categories described below.

Also N and P surplus can be modelled well. Greatest uncertainties are related to the nitrogen being fixed by legumes and the patterns of N deposition that need to be assumed. As these are irrelevant for P, we assume a higher uncertainty related to the N-surpluses reported. Another factor for uncertainty is the allocation of the fertilizers between the crops. In our model, the fertilizer is distributed according to nutrient demand. It is likely that in reality there is a tendency of farmers overfertilizing financially attractive case crops, while underfertilizing others. In our model, water demand for crops is taken directly from AQUASTAT figures, which also suffer from some gaps and uncertainties but are available at least at country level. We assume similar uncertainties as for N and P nutrient surpluses.

Table 1: Overview of inventory data for crop production processes in the model. The table shows the relevance of each flow for each impact category and its data quality

Activity type	Flow type	Flow/Process	Sub-Flow / Sub-Process	Unit for Flow	EnergyUse [CED]	GWP [CO2-eq]	LandOccupation [hectares]	N Surplus [kg N]	P Surplus [kg P2O5]	WaterUse [Liters of Water]	Comment
Plant	Input	Buildings	Depots	ha							Extrap. from Ecoinvent 2.0
Plant	Input	CultivationOrganicSoils	CultivationOrganicSoils	ha							
Plant	Input	Deforestation	Deforestation	ha							
Plant	Input	Irrigation	Irrigation	l							
Plant	Input	LandArable	LandArable	ha							
Plant	Input	LandGrass	LandGrass	ha							
Plant	Input	Machinery	Care	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Machinery	Drying	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Machinery	Fertilisation	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Machinery	Harvesting	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Machinery	Other	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Machinery	PPTSpraying	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Machinery	Seeding	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Machinery	Tillage	ha							Extrap. from Ecoinvent 2.0
Plant	Input	MineralFertilisers	K2O_MineralFertiliserInput	kgK2O							
Plant	Input	MineralFertilisers	N_MineralFertiliserInput	kgN							
Plant	Input	MineralFertilisers	P2O5_MineralFertiliserInput	kgP2O5							
Plant	Input	N-deposition	N-deposition	ha							
Plant	Input	N-fixation	N-fixation	ha							
Plant	Input	OrganicFertilisers	N_ManureInput	kgN							
Plant	Input	OrganicFertilisers	N_ResiduesInput	kgN							
Plant	Input	OrganicFertilisers	P_ManureInput	kgN							
Plant	Input	OrganicFertilisers	P_ResiduesInput	kgN							
Plant	Input	PlantProtection	Fungizides	ha							Extrap. from Ecoinvent 2.0
Plant	Input	PlantProtection	Herbizides	ha							Extrap. from Ecoinvent 2.0
Plant	Input	PlantProtection	Insecticides	ha							Extrap. from Ecoinvent 2.0
Plant	Input	PlantProtection	OtherPPT	ha							Extrap. from Ecoinvent 2.0
Plant	Input	Seeds	Seeds	kgSeeds							
Plant	Loss	FloodedRice	CH4FloodedRice	ha							
Plant	Loss	MineralFertiliserFieldEmissions	N2O_MineralFertiliserApplicatio	kgN2O							
Plant	Loss	MineralFertiliserFieldEmissions	NH4_MineralFertiliserApplicatio	kgNH4							
Plant	Loss	MineralFertiliserFieldEmissions	NO3_MineralFertiliserApplicatio	kgNO3							
Plant	Loss	OrganicFertiliserFieldEmissions	N2O_ManureApplication	kgN2O							
Plant	Loss	OrganicFertiliserFieldEmissions	N2O_Residues	kgN2O							
Plant	Loss	OrganicFertiliserFieldEmissions	NH4_ManureApplication	kgNH4							
Plant	Loss	OrganicFertiliserFieldEmissions	NH4_Residues	kgNH4							
Plant	Loss	OrganicFertiliserFieldEmissions	NO3_ManureApplication	kgNO3							
Plant	Loss	OrganicFertiliserFieldEmissions	NO3_Residues	kgNO3							
Plant	Loss	ResidueManagement	N2OResidueManagement	kgN2O							
Plant	Loss	ResidueManagement	NH4ResidueManagement	kgNH4							
Plant	Loss	ResidueManagement	NO3ResidueManagement	kgNO3							
Plant	Output	Yield	Yield	kgYield							

Relevance
for for
impact
category

high
medium
low

good
medium
poor

Data
quality

For Global Warming Potential (GWP), there are substantial uncertainties related to our calculations, as especially the N₂O emissions are modelled based on IPCC Tier 1 factors which is known to have a high variability due to site-specific factors that cannot be considered in our model. Nevertheless, as the same assumption is used for all crops and the assumption is usually also made in specific LCA studies, this assumption does not impair the comparability of results stronger than in most other LCA. Similarly for livestock production enteric fermentation and manure management, as the most important drivers for GWP can be modelled consistently, while strong uncertainties are related to feed consumptions of animals. Due to covering trade flows in our model, feed production and also potential land use changes associated to it can be assessed more consistently than with individual attributional approaches.

Also fertilizers, especially nitrogen fertilizers suffer from the lack of crop-specific fertilizer levels in most countries. Also most of the energy-use-related CO₂-emissions are quite uncertain, as crop and country-specific data for modelling energy use is hardly available. Therefore these need to be extrapolated from existing datasets which was in our case mainlyecoinvent 2.0. While energy use does not play a substantial role for GWP (with the exception of mineral nitrogen fertilizer production), it is crucial for Cumulative Energy Demand. Hence, we consider Cumulative Energy Demand being the impact category with the highest level of uncertainty in our approach.

It should be noted that for LCAs calculated with a product-related functional unit, yields are one of the main influencing factors if it comes to differences between regions and farming systems. We consider the yield data that is available on country level as being of rather good quality.

Exemplarily, results for Global Warming Potential (GWP) of apples, beans, tomatoes, soybeans, onions, maize and wheat using the functional unit “kg of fresh product”, with no allocation being applied are presented in this paper. We chose GWP for this as it is an impact category with high relevance, mainly influencing factors and an uncertainty according to the analysis above.

Figure 2 shows that the lowest GWP is associated with apples, tomatoes and onions which is not surprising due to the higher moisture content of these fruits and vegetables compared to grains and pulses. In general the calculations are within the range of plausible results.

Only very few outliers were identified for all exemplary crops for different crops. In terms of variability, soybeans show the highest variability followed by wheat and beans. Outlier countries are mostly different ones.

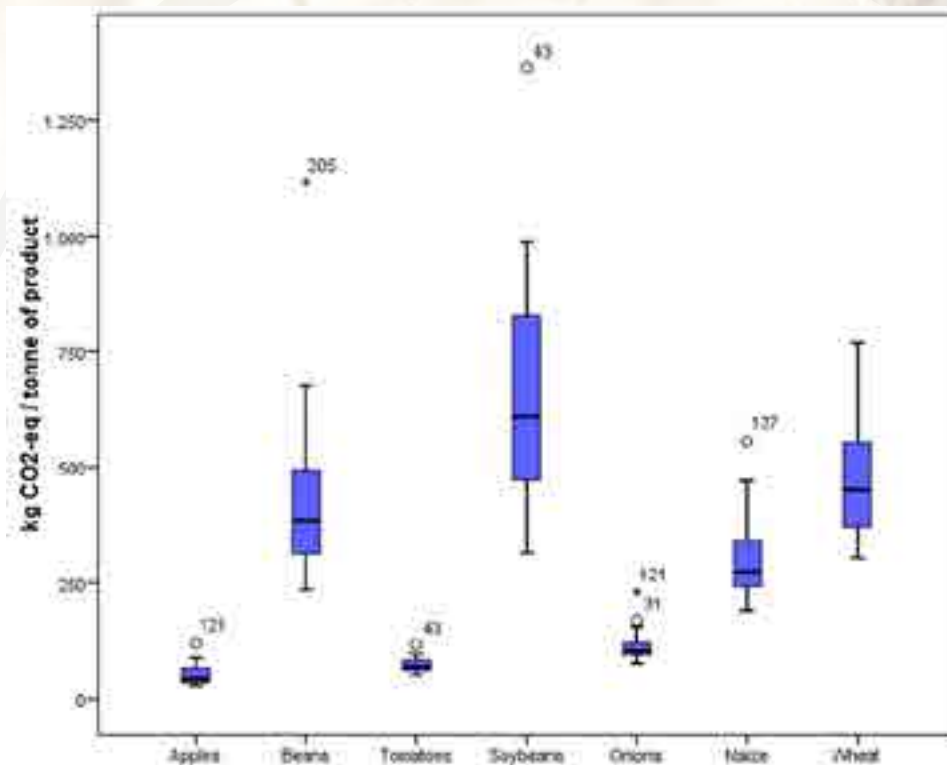


Figure 2: Overview of global warming potential of selected key commodities from different countries

4. Discussion and Conclusions

The model presented in this paper can be used as a global life cycle database for a large variety of crops and livestock activities in all countries of the world. However, the data quality provided for some inventory flows, especially for smaller countries and developing countries is partly weak. Comparing our results with results from country specific LCA studies, our approach seems to rather overestimate GHG emissions per kg of product. An important factor is that the yields reported in FAOSTAT are often lower than those used in the LCA studies we compared them with (Abeliotis *et al.*, 2013; Gan *et al.*, 2014; Knudsen, 2011; Mouron *et al.*, 2006; Theurl *et al.*, 2014). Tomatoes are an exception because inventories of greenhouse heating and infrastructure are not yet implemented in our model. For livestock production, at global level our figures are in line with the recent assessments from FAO (Gerber *et al.*, 2013; Tubiello *et al.*, 2013). For enteric fermentation, we are more in line with Gerber *et al.*, which is about a third higher than Tubiello *et al.* likely mainly due to a Tier 2 approach being used in Gerber *et al.* compared to Tier 1 in Tubiello *et al.* Therefore, our results cannot currently substitute the calculation of specific LCAs but rather serve as a basis for further detailed calculations. Due to the consistent global approach for calculating mass-flows product-specific impacts can be modelled in a harmonized way. After checking and, if needed, plausibilizing inventory data, our results should yield results that can serve as a consistent source of LCA data.

Furthermore, non-linearities due to different soil conditions, land availability and food-feed competitions can be taken into account and the model can be used for analyzing the food system both from the product-related perspective as well as from a global food systems perspective, which can lead

to quite different results as has been shown by Schader *et al.* (2014). This allows to go beyond a purely attributional LCA perspective and even to assess production systems for future scenarios, e.g. with respect to human diets.

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162. Introducing ethical rules in the Life Cycle Inventory phase with local partners in developing countries

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ABSTRACT

Since 2009, CIRAD (French Agricultural Research Centre for International Development) has been developing an ambitious Life Cycle Assessment (LCA) platform for analysing the life cycle of Mediterranean and tropical agricultural products. The LCA-CIRAD team involves around 15 LCA practitioners worldwide (France, Senegal, Thailand, Tunisia...). Implementing LCA in Mediterranean and tropical contexts raises first and foremost the challenge of data scarcity on foreground agricultural systems required for the Life Cycle Inventory (LCI) phase, and second the lack of LCI data sets for specific background processes. In those contexts, the practitioners must spend significant resources to collect large amounts of LCI data on-site for both primary and secondary data.

Creating high-quality shareable data sets was a key ambition for the new LCA platform right from the start. To this end, data required for the LCI phase are mostly collected on-site with local partners. The LCA-CIRAD team started to develop a specific set of ethical rules for the LCI phase with local partners in developing countries. This internal ethical charter is a part of the integrated quality management system (QMS) at practitioner level developed since 2012 (Biard et al. 2015).

The ethical charter is based on four main pillars: i) legal status of information shared between stakeholders in LCA, ii) protection of databases (for instance Directive 96/9/EC of the European Parliament), iii) CIRAD's values (sharing, research quality, openness/transparency, commitment to development) and iv) long term partnerships with co-management of datasets.

The LCA-CIRAD ethical internal charter follows an on-going improvement cycle, with regular feedback from practitioners, and allows building strong partnerships based on trust and transparency.

Keywords: ethics, partnership, LCA, LCI, good quality data, developing countries, legal system, values

1. Introduction

Life Cycle Assessment (LCA) is a promising tool for supporting the eco-friendly development of agri-food chains in developing countries. However, it remains poorly known and has been scarcely applied in most of these countries (e.g. within Africa). There is therefore a need for introducing and promoting the LCA conceptual framework and widening LCA applications to value chains in those regions. Agricultural value chains involve various stakeholders along the product life cycle. Depending on the life cycle stage, stakeholders may have various interests and accesses to information and data. LCA is a very data-intensive methodology, where most of the time spent by practitioners is used for the LCI phase. The identification of all the processes belonging to the foreground system leads to the consolidated list of primary data required for the study. The efforts needed for collecting

and validating specific primary data are particularly high for agri-food products of developing countries given the great diversity of products and the relatively low number of dedicated LCA applications.

LCI databases usually used for background processes, i.e. secondary data, also provide few specific datasets appropriate for developing countries, leading practitioners to use mostly European generic inventories as proxies for developing countries processes (industry, machinery, transport, chemicals production). These proxies introduce further uncertainty sources in the LCA results interpretation. Moreover, LCA results can be sensitive information that can influence the reputation of private and public stakeholders.

In this context, developing strong partnerships is one of the cornerstones of the LCA-CIRAD team working method. Its approach, founded on mutual trust between partners, aims to build up LCA win-win situations: partners in developing countries build their capacity in LCA methodology and are well informed about the implications of the study to which they collaborate, while CIRAD's scientists can benefit from the best existing data on agricultural systems in these contexts and deliver reliable LCA studies for all. This approach required taking into account ethical and legal considerations on the collection and use of LCI data with different partners.

The management of data acquired from partners outside of a practitioner's institution and used for LCA has not really been studied and discussed among the LCA community so far. Ethical implications related to the management and sharing of these data has also not been much explored. Ethical data management covers several aspects such as long term vision and institutional values shared by partners; data transformation throughout LCA from goal and scope until LCIA; regional and international intellectual property and legal status of data and database; data sharing between partners from the academic and private sectors; and dataset economic model.

The objectives of this paper are:

- ✔ To present and discuss the approach used to develop these ethical rules and provide some perspectives on the issues addressed.
- ✔ To present the set of ethical rules developed by the LCA-CIRAD research group to guide the collection and management of LCI data with partners.

2. Methods

The development of the LCA-CIRAD set of ethical rules started with a critical analysis of a range of CIRAD LCA projects over the past 8 years in relation with data management, partnership and potential ethical issues. This state of the art revealed a wide range of situations, leading to develop a specific work on i) data flow description, ii) legal status and iii) discussion about ethics (Figure 1). The LCA-CIRAD team decided to work together on this topic in small workshops under the supervision of the engineer in charge of the LCA-CIRAD Quality Management System (Biard et al. 2015).

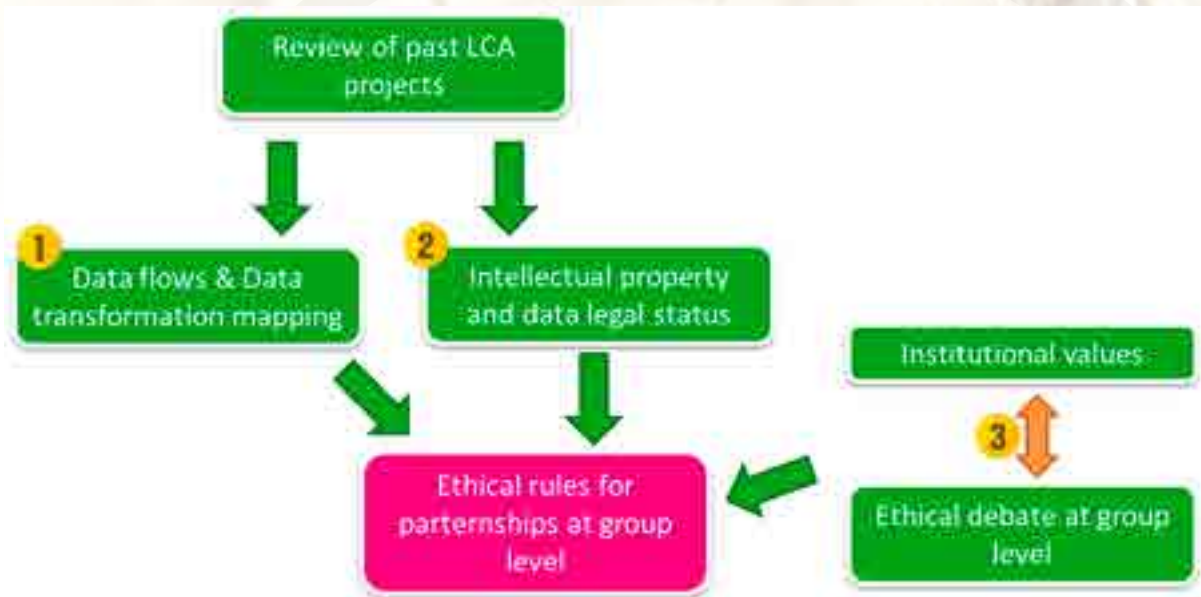


Figure 1: Ethical charter development approach at group-level.

2.1 Mapping the data flows

At the beginning of the process, some LCA-CIRAD team members expressed their uncertainty about the legal status of the data they had been collecting and using in their LCA projects. In this perspective, the first issue addressed was to map the data flows and data transformation in LCA-CIRAD's projects, to define data flows and key actors, and related potential legal issues (Figure 2).

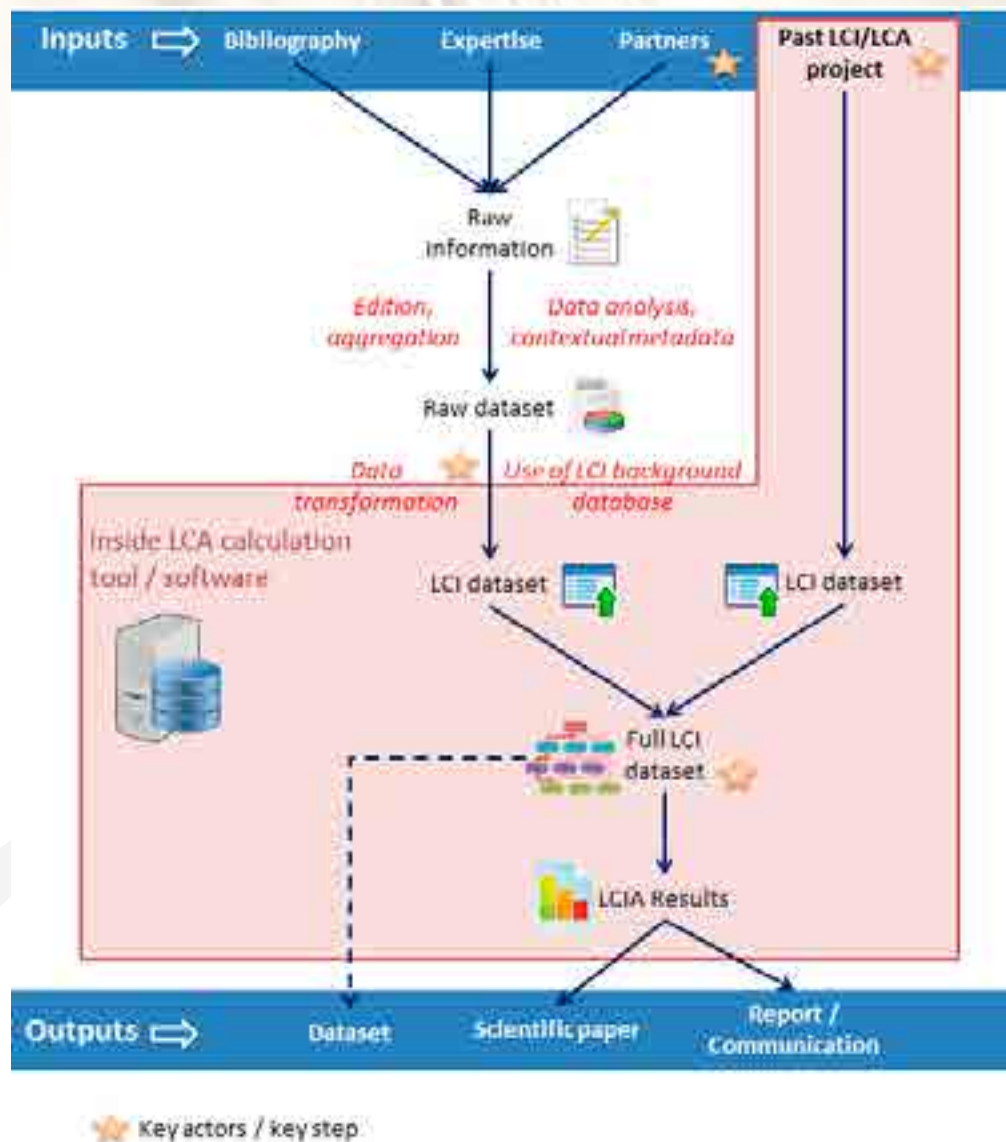


Figure 2: Data flow and data transformation mapping, based on LCA projects experienced.

The mapping reveals that from raw data sources until LCI datasets integrated in LCA software, the data flows could be summarized as a continuous process of aggregation of information and data, adding metadata. This transformation progressively leads to consistent LCI datasets which store new consolidated information in a standardized compact format, according to the specific procedures on dataset and metadata quality as formalized in the LCA-CIRAD Quality Management System (Biard et al. 2015).

Full LCI dataset and LCIA results are the two coproducts of a LCA study. The mapping highlights the data life cycle inside a LCA study; LCI datasets are both final products of the work (research outputs) and potential raw material (research input) for subsequent LCA studies. This mapping based on LCA-CIRAD team members' experience was used as a baseline for legal status analysis, presented in the following section.

2.2 Dealing with legal status

The second step was to describe the legal status of data at each key step of the data flow. To do so, it was necessary to first identify the property rights and their holders, and to transpose the legal framework in the data flow. This step focused mainly on database Intellectual Property (IP) in compliance with the European Union law.

In the European Union law (the European Parliament and the Council of the European Union 1996), a database is a “collection of independent works, data or other materials arranged in a systematic or methodical way and individually accessible by electronic or other means.” The legal system is divided in two parts: copyright and Sui Generis⁴ rights, whose characteristics are shown in Table 1.

Table 1: simplified characteristics of the European Union database legal system

	Copyright	Sui Generis Right
Scope	Database structure	Database content
Protection criteria	Originality, creativity	Investment
Right holder	Creator (natural person)	Investor (natural or legal person)

The European Union database legal system claims that LCI datasets - as database content - are produced by the investor, in our case Cirad as a legal person. This default situation could be applied generally in any LCA project team where the LCA practitioner is an employee. The wages of these LCA practitioners are legally considered as substantial investments; therefore the employer or the institution normally gets the legal producer status. On the contrary, new LCI data are factual information and are not considered, by definition, as copyrighted material⁵.

The LCA-CIRAD team members received the necessary training on this specific topic, because the legal system could be seen as counterintuitive: data is the first fruit of the work of researchers, who could consider that they own the data. But even if they “technically” produce some data, they are not producers in the legal sense.

2.3 Debating ethical rules

The third step was then to debate and formulate ethical rules at group-level, based on both i) CIRAD institutional values and ii) fruitful discussions on ethical values and partnership’s experience of LCA-CIRAD team members.

With the legal analysis in mind, LCA-CIRAD team members clearly formulated a need for consultation on ethical rules and advices in agreement with CIRAD institutional ethical and strategic position. Firstly, the LCA-CIRAD team should base its ethical rules on CIRAD proclaimed institutional values. In addition to the French National Charter for Research Integrity (2015) and the 8th opinion of the Joint Consultative Ethics Committee (Cirad Inra 2015), CIRAD declared some

⁴ In law and especially IP, the term “Sui Generis” is used to identify a legal situation that exists independently of other preexisting categorizations because of its singularity. That is the case for database content protection formalized in Directive No. 96/9/EC of 11 March 1996 and called Sui Generis Right.

⁵ Only a specific contract about information exchanged before LCI leads to a legal status for this information, which becomes copyrightable material.

values⁶ for its operations, determining its partnership working method: “i) each and every one of its operations fits into projects conducted jointly with partners working for development in developing countries ii) a constant concern if we are to make an effective contribution to tomorrow's agriculture: generating, sharing and passing on knowledge.” Secondly, the LCA-Cirad team decided to debate in little workshops and non-exhaustively the following ethical issues in addition to the previous legal analysis:

- Which baseline scenario for data ownership should be promoted?
- Do all the data produced need to be shared?
- How to improve partners' capacity to exploit the data by themselves?
- What could be the most relevant and useful level of dataset aggregation?
- How to define the economic value of a dataset?
- How to protect the interests of data producers or data sources?
- How to define what are sensitive data, and how to manage them?
- How to apply the Sui Generis Right in Cirad's hierarchy and how should the decision chain look like?
- How to take into account partnership specificities of Cirad?

The fruitful discussions on ethics promoted by LCA-CIRAD team members led to the following shared values and principles. The members observed that i) producing high quality data is a major ethical requirement for the group, ii) dataset owning and sharing issues requires formal discussions and knowledge transfer with partners and iii) given the growing numbers and the extreme diversity of CIRAD partnership situations leading to multiple choices⁷, a decision process on data management on a case by case basis is inevitable.

3. Results

The ethical charter developed by the LCA-CIRAD team begins with a summary of European database legal framework, simplified and adapted to the specific context of data flows and transformation in LCA as presented in this paper. For specific issues out of the scope of the basic legal framework, the case should be discussed with CIRAD's legal department.

The LCA-CIRAD team decided to go further than the legal framework, putting more emphasis on trust and partnerships in their set of ethical rules, acknowledging the fact that strong partnerships are particularly important in the context of LCI data collection and sharing. The LCA-CIRAD team established the following main rules based on consensus:

1. The quality of the relationship with the partners and scientific development are the two pillars;
2. No data dissemination must be done without taking into account the impact this could have on the interests or reputation of the partners and their relationship with LCA-CIRAD or CIRAD as a whole;
3. Datasets dissemination for direct commercial valuation to strict dataset-buyers is not a strategic priority for CIRAD;

Additionally, the LCA-CIRAD team should contribute to LCA capacity building in developing countries, so that partners become experts and work independently on LCA projects, rather than act

⁶ <http://www.cirad.fr/en/who-are-we/our-values>

⁷ Especially in terms of status, geographical context and resources.

only as simple data providers. This entails building medium-term or long-term partnerships offering LCA trainings at novice and expert levels and also specific trainings on LCA database Quality Management System (QMS).

The charter goes more into the details of implementation, beginning with interactions with partners. If full LCI dataset or LCIA results dissemination is required in a specific project, those conditions should be well explained to partners. Partners' validation of the conditions should be written as much as possible in the collaboration agreement. Moreover, external demands for LCI datasets or LCIA results arise after the end of the project, the impact on the relationship with the partners must be taken into account, in addition to the contractual clauses concerning the data dissemination.

Regarding the decision process about data dissemination, the decision belongs to the scientific team leader, seen as the most convenient decision-maker to exercise the Sui Generis Right. The scientific lab leader is encouraged to base his decisions on the advices of practitioners who have been working on the data. It is also at this level that data dissemination could be decided after an embargo period, i.e. scientific publication and with agreement of the partners.

4. Discussion

The ethical charter developed by the LCA-CIRAD team presents some limitations, which are under discussion between the members. The following paragraphs summarize the key remaining issues and potential improvements:

- **Geographic and legal scope:** the approach was based on the European Union database legal system, with copyright and Sui Generis Right. LCA-CIRAD team members decided to apply the highest level of data protection from that E.U legal system to work with partners in developing countries where data management laws may be less advanced⁸. The next step would be to examine in depth other database legal systems, which are not based on the same principles.
- **Technical scope:** the approach was mainly focused on dataset, avoiding software, patents and personal information data issues. Those topics need to be covered, because some data used in a LCA study could contain personal information or patented processes.
- **License development:** the ethical rules do not cover explicitly the type of licenses that could be used for dataset dissemination. Further work on Open Database License (ODBL) or Creative Commons 4.0 could be useful to clarify that point.

5. Conclusions

This paper presented and discussed the set of ethical rules developed by the LCA-CIRAD research group, emphasizing how this ethical framework can guide the data collection and management within a LCA study.

The LCA-CIRAD ethical internal charter meets the need formulated by the team, and the content of the charter (set of ethical rules, key values and decision process) established by the members is relevant for Cirad's partnership situations. The operational implementation of the charter allows initial discussions with partners on LCA issues, data collection rules and process, dataset dissemination and

⁸ As far as we know and non-exhaustively: Brazilian law does not offer sui generis database rights.

reuse. Those discussions are a prerequisite for establishing mutual trust between partners. As detailed in the discussion, the ethical charter may not yet cover exhaustively the wide range of international partnerships situations, but its on-going improvement is one of the goals of the LCA-CIRAD team.

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ORAL PAPERS

SOIL, CARBON AND PESTICIDES

5. Soil, Carbon and Pesticides

150. Assessing the environmental impacts of agricultural production on soil in a global Life Cycle Impact Assessment method: A framework

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ABSTRACT

A decrease in soil quality due to human activities, also known as soil degradation is associated with negative, in some cases even irreversible effects for ecosystem processes such as biotic productivity. Soil degradation encompasses effects like erosion, organic matter decline, salinization, compaction, landslides, contamination, sealing and soil biodiversity decline. In life cycle assessment there are no comprehensive operational methods for the impact assessment of soil degradation yet. With this paper we propose a new framework for the impact assessment of soil degradation in agricultural production on regional as well as on global scale. It encompasses four aspects on soil degradation trying to avoid overlapping effects from different impacts. The impacts are quantified in terms of “long-term yield loss” and are aggregated to estimate the overall impact on the biotic production potential. In one example we show the characterization factors for soil compaction in integrated potato production. However, effort remains to make the framework operable also for other impact pathways than compaction.

Keywords: LCIA, soil degradation, framework, yield loss, impact pathway

1. Introduction

Sustainable management of soils is a key issue of modern times. Growing populations compete for food, fodder, fuel and fabrics and thus for soil that is essential for the production of the different assets. Soils have manifold functions besides biomass production: Soils build the physical environment for humans, they harbor biodiversity living belowground (Pulleman et al. 2012), they are the source of raw materials, they store carbon and finally they store, filter and transform nutrients, substances and water (McBratney et al. 2011). A decrease in soil quality due to human activities, also known as soil degradation, is associated to negative long-term and in some cases irreversible effects on soil functioning (Lal 2009). These effects make the soil less fit for specific purposes such as crop production (Bindraban et al. 2012). The likelihood is high that degraded land will be compensated by gaining land through deforestation that causes additional negative impacts on the environment (Gomiero 2016). As 25 % of the global agricultural land is said to already be highly degraded (FAO 2011), it is urgent to stop the negative impacts on soils and to preserve its functioning.

Soil degradation is a combination of different negative impacts on soil quality. In Europe the most important processes leading to soil degradation are said to be erosion, organic matter decline, salinization, compaction, landslides, contamination, sealing and soil biodiversity decline. The costs of these impacts are estimated to be up to €38 billion yearly in the EU25. These estimates are rough due week quantitative and qualitative data (Montanarella 2007).

Erosion removes the nutrient rich and organic matter dense upper layer of the soil by the force of wind or water power. The amount of material lost exceeds the amount of new built soil from pedogenesis. The global average erosion rate vary from 0.001-2 t soil/ha*yr in flat areas and 1-5 t soil/ha*yr in mountainous regions (Pimentel 2006). This results in lower capability to fulfill functions as e.g. water runoff, water holding capacity or soil fertility.

Soil fertility is also degraded due to organic matter decline. This is the reduction of the share of organic matter in a soil. Reasons for that are erosion, drainage, cultivation practices and else. The organic matter decline thus reduces storage and availability of nutrients and it has a negative effect for instance on the soil structure.

Erosion and salinization are perhaps the most extensive degradation processes (DeLong et al. 2015). Soil salinization is the accumulation of soluble salts, mainly from Na, Mg and Ca due to poor irrigation technology, inappropriate drainage and the use of saline irrigation waters (Montanarella 2007). It mostly occurs in arid and semi-arid agricultural regions (Año-Vidal et al. 2012).

Soil compaction generally describes the compression and shearing of soil pore structure. The outcome is reduced soil aeration, drainage capability, root penetration etc. It is induced by heavy machinery load or trampling on wet soils. The economic impact of soil compaction is estimated to be of the same magnitude as the impacts described above.

Landslides are mass movements of soils at slopes. Combinations of different conditions, as for example clayed subsoils, intensive land use through tourism and heavy rainfalls, can trigger landslides (Montanarella 2007).

In many production processes substances are used, either direct (as pesticides) or indirect (for example as waste disposal). They can contaminate soils and harm agricultural production and groundwater.

All these soil degradation processes also decline soil biodiversity, which comprises at least one quarter to one third of all living organisms of the planet (Breure et al. 2012). It is essential for the metabolic capacity of the ecosystem and soil formation (Montanarella 2007).

Additionally to the European key threats, desertification should be mentioned too. The UN Convention on Combating Desertification defined desertification as “land degradation in arid, semi-arid and dry sub-humid lands resulting from various factors including climatic variation and human activities”.

One method to identify the impact of production processes on soil degradation is the method of Life Cycle Assessment (LCA). In LCA there are only a few indicators addressing soil quality or soil degradation (Garrigues et al. 2012), though there is widespread recognition that more comprehensive indicators are needed (Milà i Canals et al. 2007a; Nemecek et al. 2016). The barriers which have prevented such development include the complexity of soils and the lack of models for computer based simulations in regional assessment (Mutel et al. 2012).

Below we discuss existing approaches that try to quantify and assess soil quality, soil degradation and soil functioning. Methods assessing land use considering biodiversity as e.g. (Chaudhary et al. 2015), ecosystem services and functions (Koellner et al. 2013), soil contamination, acidification and eutrophication are not discussed, because they are covered in other assessment methods.

Assessment methods for overall soil quality. Several existing approaches address soil organic matter (SOM). The most detailed approach is presented by (Brandão et al. 2011; Milà i Canals et al. 2007b), where SOM is a sole indicator of soil quality. It is used as a proxy for soil quality, but it omits important drivers of soil quality loss like compaction and salinization (Hauschild et al. 2012) and sealing. The assessment requires SOM measurements for the inventory, but calculations of SOM content from models or values from literature could be used as well (Hauschild et al. 2012). It can be applied for agriculture and forestry only (Garrigues et al. 2012). The method SALCA-SQ (Oberholzer et al. 2012) assesses SOM too and adds eight other soil quality indicators affected by the agricultural management. It is said to be the method with the highest level of description of soil quality, accordingly the data requirement is high and it is calibrated for Swiss farms (Garrigues et al. 2012). The level of SOM is also suggested to be addressed by Cowell et al. (2000) and Achten et al. (2009). (Cowell and Clift 2000) discuss allocation problems, occurring irregularly during one crop rotation, as well as changes in soil mass, nutrients, weeds and weed seeds, pathogens, the level of SOM, salts, the soil's pH and the form of the topsoil. All these factors are suggested to be considered. (Achten et al. 2009) propose cation exchange capacity (CEC) and base saturation (BS) of the topsoil to quantify soil fertility and SOM of the topsoil and soil compaction (e.g. infiltration rate is used as a soil compaction indicator) to assess soil structure. Both are indicators for ecosystem structural and functional quality. The impact indicator scores are the relative impacts compared to the values in a system with potential natural vegetation.

Assessment methods for single soil degradation processes. The potential desertification impact of any human activity is included in an assessment method developed by (Núñez et al. 2010). It considers variables such as the aridity index, water erosion, aquifer overexploitation and fire risk. The characterization factors (CF) for erosion are derived from the world map of the Global Assessment of Human induced Soil Degradation GLASOD. In a second study, a globally applicable, spatially differentiated LCIA method for assessing soil erosion was developed. The importance of regionalized assessment (e.g. site-dependent soil properties) was shown in a case study (Núñez et al. 2012). (Feitz and Lundie 2002) propose a preliminary soil salinization impact model for the assessment of potential land degradation. The model is based on the relationship between the sodium adsorption ratio (SAR)

and the electrolyte concentration (EC), which addresses soil permeability hazard and extent of soil dispersion, potential dispersion and flocculation. Its application is limited to soil salinization from irrigation practices. The model has to be adapted to particular sites, e.g. the electrolyte threshold curve. (Leske and Buckley 2004) developed a salinity impact category, which addresses the total salinity potential for different compartments (atmosphere, surface water, natural surfaces and agricultural surfaces) relevant for South African conditions. (Payen et al. 2016) presents a new framework for salinization that includes the studies above.

Assessment methods for selected soil functions. The LANCA®-tool has been made operable for different mining and agricultural processes in selected countries. It quantifies the effects on four soil regulating services: mechanical filtration, physicochemical filtration, biotic production and groundwater replenishment (Beck et al. 2010). The model needs site-specific input data for several time steps, e.g. soil texture, declination, summer precipitation, type of land use, skeletal content, humus content, surface type for calculating erosion resistance etc. If specific data is not available the tool provides data on country-level. Differentiations between farming management practices are not possible (Beck et al. 2010). (Saad et al. 2011) used the LANCA®-tool to calculate CFs for different spatial levels. The results highlighted the importance of using spatially differentiated characterization factors for the assessment of soil quality.

Research gaps. The aforementioned methods address soil degradation due to agricultural processes without distinguishing different management practices and production standards. Furthermore, they do not consider all relevant aspects of soil degradation since they assess only single soil degradative processes. Some of the methods are limited to the assessment in specific countries. Moreover, most of the methods presented above are difficult to apply because of the excessive data requirements. Here, we will present a new framework for the impact assessment of soil degradation in agricultural production, applicable on regional as well as global scale (Figure 1). The framework includes the main drivers and impact pathways of compaction, organic matter decline, erosion, desertification, salinization and sealing. The impacts are quantified in terms of “long-term yield loss” and aggregated across the various impact pathways to estimate the overall impact on soil degradation.

2. The framework

Some of the methods presented above are difficult to apply because of the excessive data requirements. We therefore set up a multi-level system, in which the LCA practitioner enters data on location, production standard, the kind of crop and the “use” of constructed area. This information allows for the query in a background database containing relevant information on e.g. soil texture, weather data, elevation, slope and land use including machinery use, its specification and else. The information acquired from the data base query is consequently used to calculate regionalized characterization factors (CFs). The spatial differentiation is relevant when studying territories with heterogeneity in environmental characteristics (Nitschelm et al. 2016) as it is the case for soils. Local weather data is relevant for soil degradation as well. Today’s weather data is available globally and regionalized (e.g. www.meteonorm.com) and including it into the model is a necessary next step improving the quality of LCIA. In the background database we also provide standard datasets about agricultural practices in different production systems and for different crops. These datasets can be adapted when more accurate data is available.

The nine main soil threats we consider in our framework are erosion, organic matter decline, salinization, compaction, landslides, contamination, sealing, soil biodiversity decline and desertification. They are related directly or indirectly up to different degrees. In order to avoid double counting of impacts it is reasonable to carefully make a selection of relevant impacts. Soil organic matter (SOM) was considered to be the most appropriate indicator for soil quality in LCA and CFs were calculated for eight land use types on the climate region level (Brandão and I Canals 2013). To the same conclusion came (Milà i Canals and de Baan 2015) when they described the state of the indicators. But (Milà i Canals et al. 2007b) stated that not all aspects of soil quality are represented by SOM. Erosion, compaction, build-up of toxic substances, acidification and salinization are not directly assessed by using SOM as an indicator. We therefore suggest using soil organic matter as a proxy for erosion, soil organic matter decline and desertification. Additionally, we suggest considering soil compaction, salinization and sealing in order to have an accurate set of impacts for

the assessment of soil degradation. The remaining threats are landslides and soil biodiversity decline. Landslides are indeed important threats to the soil but are not in the focus when assessing agricultural processes (except for land use changes, such as deforestation and land abandonment (Montanarella 2007)). Soil biodiversity decline could be integrated in biodiversity impact assessment methods. However it is also represented in the assessment of SOM, that is crucial for soil biodiversity (Montanarella 2007).

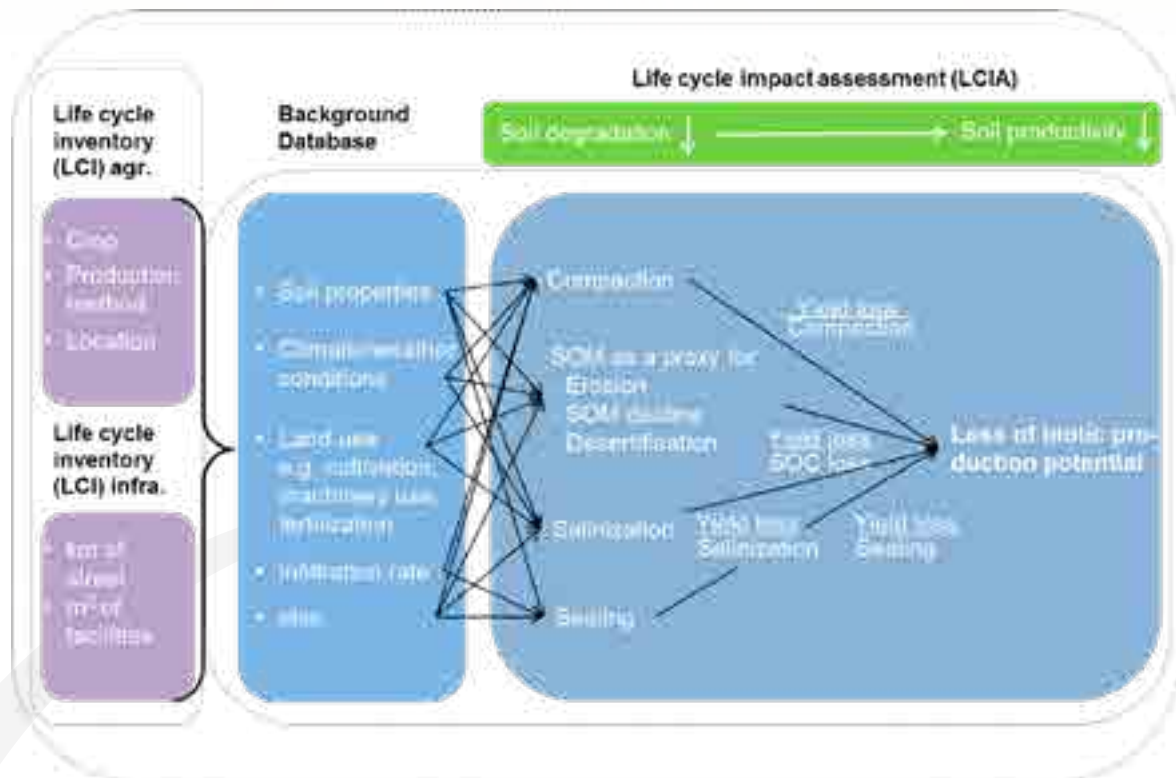


Figure 1: Impact pathway of soil degradation processes on soil productivity.

The framework we suggest includes the main drivers and impact pathways of the four selected aspects of soil degradation: Compaction, soil organic matter decline, salinization and sealing. Impacts are then quantified in terms of “long-term yield loss” and aggregated across the various impact pathways to estimate the overall loss of biotic production potential through soil degradation (Figure 1).

The application of the framework is illustrated for the impact of soil compaction (Figure 1). The model of (Arvidsson and Håkansson 1991) was adapted to assess yield losses through soil compaction in a regionalized manner, with global coverage. The background database comprises crop production data (with around 150 crops and production methods – organic and integrated production standard), regionalized soil texture data (ISRIC - World Soil Information 2013), soil moisture data and machine specifications for all machines used in crop productions. A publication about the development of a new soil compaction method, based on the model of (Arvidsson and Håkansson 1991), with a set of background data and readily applicable CFs is in preparation.

Characterization factors for the assessment of soil organic matter decline have been developed and tested by various researchers, for example (Goglio et al. 2015; Mattila et al. 2012; Morais et al. 2016). The IPCC provides relative carbon stock change factors for soil (IPCC 2006). These factors are available for different land-use types (e.g. long- and short-term cultivated cropland or permanent grassland) as well as land-use management types (e.g. different tillage and fertilization practices). Furthermore, they provide estimations of the initial carbon stock of the natural vegetation in different climate regions. Brandão and Milà i Canals (2013) used the SOC values and change rates to develop a LCIA method (with CF) for the biotic production potential. For our goal we need an extension of the method by Brandão and Milà i Canals (2013) to relate crop specific yield and SOC change ($\Delta C \text{ year}^{-1} \text{ m}^{-2}$). There are two ways to do so: One is to estimate the yield loss via nutrient stock change. The available nitrogen (N) mineralized from SOM (NH_4^+ and NO_3^-) can be taken up by plants, but will

also get lost partly via leaching, volatilization or denitrification, which should be considered. Bontkes and Keulen (2003) suggested that 25% of the mineralized N are lost via volatilization and denitrification. Estimations of leaching are more difficult to make as it largely depends on the actual rainfall amount. More accurate estimations might be possible using the method SALCA-NO₃ (Richner et al. 2014). As crop yields do not solely depend on the N supplied by the soil, the N supplied by the organic or synthetic fertilizer has to be taken into account. Finally, yield can be predicted using nitrogen-yield response curves. Nitrogen-yield response curves were firstly suggested by Eilhard Alfred Mitscherlich (Harmsen 2000). Mueller et al. (2012) used Mitscherlich-Baule nitrogen-yield response curves to estimate global maximum attainable yields for different crops considering fertilizer application, irrigation and climate. Alternatively, crop yields and SOC content in response to fertilizer management could be modelled using crop growth models. Those models were already used in other LCA studies (Adler et al. 2007; Kim et al. 2009; Veltman et al. 2014). With crop growth models such as Daycent (Del Grosso et al. 2008) or CropSyst (Stöckle et al. 2003) that take climatic and soil conditions into account, the yield of specific crops could be modeled for different fertilizer scenarios. Furthermore, the effect of a certain management scenario can be evaluated over many years and taking crop rotations into account as well as restoration time.

(Payen et al. 2016) evaluated the existing life cycle impact methods addressing salinization. She proposed a three-stage approach for the setup of a relevant and complete model to assess salinization impacts in LCA. It will focus on anthropogenic salinization and considers salinization associated with land use change, irrigation, brine disposal and overuse of a water body (e.g. through seawater intrusion). However, this approach is still on a conceptual level and not yet operational. For soil degradation we would select the impacts associated with land use change, irrigation and brine disposal. That leads to the proposed midpoint indicator “soil fertility and structure decline”. The normalized CFs could be used in the relationship of soil salinity and energy harvested by photosynthesis as (described in (Munns and Gilliam 2015)). The energy harvested in turn can serve as an indicator for yield loss. The average crop specific salt tolerance (Katerji et al. 2000) has to be considered by implementing another factor reflecting the crop differences. Effects of salinity have been studied in various field experiments for different crops (e.g. (Katerji et al. 2003; Kim et al. 2016)). These results could be used to verify the results.

For the impact of sealing we propose a very rough estimate. Up to date we are not aware of existing LCIA methods considering sealing aspects. But we are aware of the importance to include sealing impacts into LCIA of agricultural products. Our suggestion is to use the runoff curve number as a proxy for the sealing intensity of roads, buildings and other infrastructure. The runoff curve number is dependent on the intensity of the sealing (Maurer et al. 2012). The amount of area “used” in a production of a product is multiplied with the runoff curve number given in construction guidelines for the rainwater runoff (e.g. (Petschek 2015)). The result will afterwards be multiplied with the yield of the according crop, in order to get a proxy for the yield loss through sealing.

As described above, we propose to consistently address four soil degradation processes and express them in the same unit, to make their soil degradation effect comparable. However, since the effects of compaction, SOM loss, salinization and sealing are not linearly additive, we propose to use a similar approach as followed by the response addition concept for the assessment of chemical mixtures.

3. Example: soil compaction

To illustrate our method, in the following we present a set of CFs (expressed in % yield loss) for compaction applicable for potato production (Figure 2): It is calculated under the assumption that potatoes are grown everywhere and on wet soils. It is therefore not a realistic picture but it shows the possible extremes.

4. Discussion and Conclusion

Many attempts have been made in the last few years to include soil degradation impacts in LCIA but no one was able to cover the whole spectrum of soil degradation. Our attempt outlines a framework that aims to achieve this goal. Challenges include finding the right balance between detail

and completeness. The question of reference state and uncertainty should also be investigated. The implementation of our method is illustrated for the impact pathway of soil compaction (publication in prep.). In the future, we aim to include the other aforementioned impact pathways in our method in a consistent manner and to integrate the whole method for soil degradation in existing LCIA methods. The applicability also depends on the flexibility of LCA software to use regionalized impact assessment methods. Special attention has to be given in avoiding double counting, when the method is used together with future other methods.

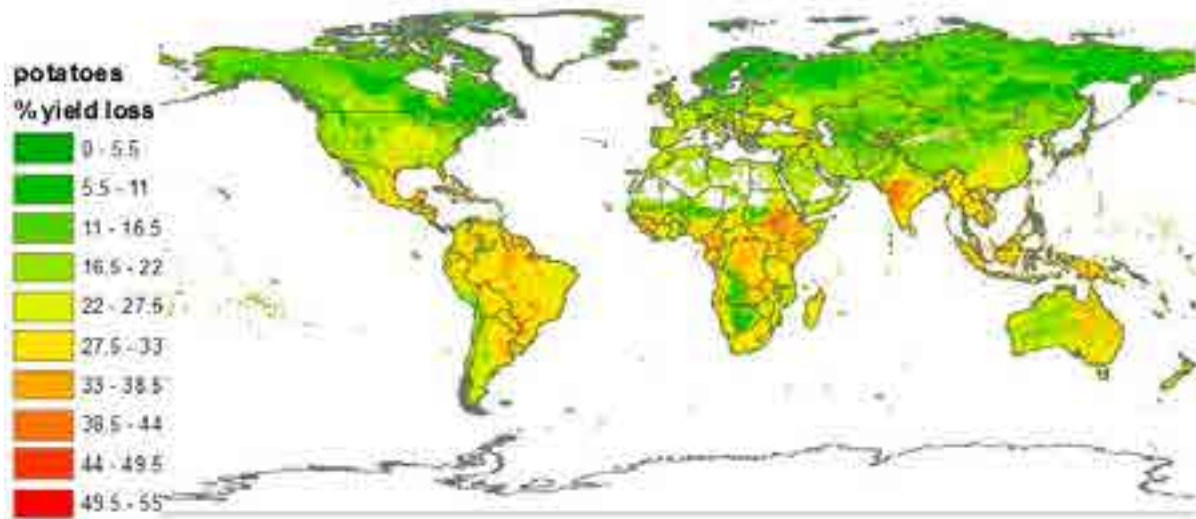


Figure 2: Yield loss of potatoes due to soil compaction. Results show a worst case scenario with high soil humidity and high production intensity in integrated production. Differences in yield losses are driven by varying soil texture.

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144. The importance of a life cycle approach for valuing carbon sequestration

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ABSTRACT

Carbon sequestration (C-seq) in grassland has been proposed as a strategy to reduce the net contribution of livestock to climate change. Carbon stored in soils, however, can easily be re-emitted to the atmosphere if soil conditions change. The aim of this study was to evaluate the importance of C-seq for reducing the impact of dairy production on climate change over time, based on life cycle assessment. Annual emissions of carbon dioxide, methane and nitrous oxide (cradle-to-farm gate) were analysed for two types of dairy systems in the Netherlands: a grass-based system (high C-seq potential) and a maize-based system (low C-seq potential). Soil carbon fluxes were modelled to quantify the uptake and release of carbon dioxide in agricultural soils. The climate impact per ton of fat-and-protein corrected milk (FCPM) for each system was modelled over time, based on the radiative forcing and atmospheric lifetime of annual emissions and soil carbon fluxes. Systems were compared for a situation in which soil carbon was re-emitted after 20 years, or stored for an indefinite period of time. Results show that C-seq favours the grass-based system in the short-term, until the point at which soil carbon is re-emitted (i.e., 20 years), or reaches equilibrium (after 70 years). Results demonstrate the importance of including both annual emissions and soil carbon fluxes by means of a life cycle approach, and to consider the climate impact over time, when valuing the potential benefit of C-seq.

Keywords: livestock, greenhouse gas emissions, climate change, mitigation, dairy production

1. Introduction

The livestock sector is a significant contributor to anthropogenic climate change, estimated to being responsible for about 14.5% of global anthropogenic greenhouse gas (GHG) emissions (Gerber et al., 2013). Most of these emissions are non-carbon dioxide (CO₂) greenhouse gases; methane (CH₄) and nitrous oxide (N₂O) accounts for 44% and 29% of emissions, respectively, with CO₂ accounting for the remaining 27%. The cattle sector (beef and dairy) is the main contributor to these emissions, with a share of 65% of total emissions (Gerber et al., 2013), based on a life cycle approach that includes on-farm as well as off-farm emissions (i.e., emissions related to the production and transport of farm-inputs and to the transport and processing of milk).

One strategy frequently proposed for mitigating the contribution of livestock to climate change is carbon sequestration (C-seq) in grassland soils. Worldwide, permanent pastures are suggested to potentially store 0.01-0.3 Gt C yr⁻¹ (Lal, 2004), which equals up to 4% of global GHG emissions caused by the livestock sector (Gerber et al., 2013). Estimates about the maximum C-seq potential of grasslands, however, vary widely (Henderson et al., 2015). Some studies even suggest that grazing systems can have a negative GHG balance, acting as a net sink rather than a source to the atmosphere (de Figueiredo et al., 2015).

To assess C-seq as a strategy to reduce the climate impact of livestock systems, it is vital to understand how livestock systems influence the carbon cycle. Two important sources of GHG emissions from livestock are enteric fermentation from ruminants and manure decomposition. Both sources contribute to the emission of CH₄ through decomposition of organic matter in oxygen deprived conditions. In addition to CH₄, livestock systems contribute to the emission of CO₂ through animal respiration, through burning and microbial decay of biomass (e.g. deforestation), and through fossil fuel use to provide energy for on-farm and off-farm activities. Livestock systems can also contribute to the uptake of atmospheric CO₂ through photosynthesis during crop and grass cultivation. Most of this CO₂ is re-emitted to the atmosphere on a short timeframe as ingested and respired by livestock, thus not entering into a long-term soil carbon sink. Depending on e.g. management, soil type and climate some of the CO₂ taken up by plants and grasses can be converted into stable carbon (in soil organic matter) after microbial decomposition of plant roots and residues and stored in the soil

for a long time-period, here referred to as soil C-seq. Finally, grassland / crop fertilization and manure management are important sources of N_2O .

As noted above, CO_2 uptake by crops consumed by the animal are assumed to be balanced by CO_2 emissions from animal respiration. Both aspects are considered to be part of the short-term carbon cycle and are usually not included in life cycle GHG calculations (Munoz et al., 2013). Changes in carbon stocks in livestock are generally neglected as well. Other emissions, including emissions of CH_4 and N_2O , as well as CO_2 emissions from the use of fossil fuels are usually included. The impact of biogenic CH_4 is somewhat lower than that of fossil CH_4 because it doesn't include the impact of CO_2 resulting from the atmospheric decay of CH_4 (Myhre et al., 2013). Emissions of CO_2 and CH_4 from the use of fossil fuels are advised to be fully accounted as they result from human interference with the long-term carbon cycle, which naturally includes the formation and oxidative weathering of fossil fuels over millions of years (Berner, 2003).

Soil C-seq is characterised by two important aspects. First, soil carbon stocks stabilize over time as soil C sinks become saturated. Second, soil C-seq is a reversible process, so any sequestered carbon can easily be re-emitted if soil management change. Accounting for C-seq in GHG calculations, therefore, can be difficult. Studies generally sum CO_2 sequestration and annual emissions, lowering the impact of livestock systems (e.g., Henderson et al., 2015). This simplified summing up, however, makes long term impacts to be hidden by temporary storage solutions (Jorgensen and Hauschild, 2013). For example, summing up C-seq and fossil fuel combustion, implies that storing 1 Gt C in agricultural soils neutralises the emission of 1 Gt C from combustion of fossil fuels. At the moment the soil organic carbon is re-emitted, however, the real output of the system would still be equal to the impact of the combustion of fossil fuels (1 Gt C), only delayed by the number of years the carbon has been stored.

An important challenge when accounting for soil C-seq, therefore, is to recognize the differences in time scale over which the carbon fluxes take place. Within this context it is equally important to recognize differences in impacts on radiative forcing and in atmospheric lifetime between GHGs. On a weight basis, the impact of CH_4 on radiative forcing is about 120 times the impact of CO_2 (at current atmospheric concentration levels), while N_2O is about 210 times as strong as CO_2 (Myhre et al., 2013). However, CO_2 has a much longer atmospheric lifetime (with 20-30% of emissions remaining in the atmosphere for more than thousands of years), than that of CH_4 (12 years) and N_2O (120 years). Greenhouse gases are generally summed based on their global warming potential (GWP) over a 100-years' time horizon. The GWP of a GHG is defined as the time-integrated radiative forcing of an emission pulse of that gas relative to that of an equal mass of CO_2 (Persson et al., 2015). As a result of differences in atmospheric lifetimes, emissions that are equal in terms of their GWPs can have different climatic consequences, which vary over time (Smith and Wigley, 2000). As stressed by Pierrehumbert and Eshel (2015), this can have important consequences when assessing the impact of C-seq. Sequestration of a very long-lived greenhouse gas (CO_2) is, in that case, exchanged against a very short-lived greenhouse gas with a higher radiative forcing (CH_4).

This study aims to assess the value of C-seq in climate change mitigation of livestock systems, by accounting for time-varying uptake and release of GHGs. Dairy production is used as a case study to examine the impact of C-seq over time, while accounting for other GHG emissions based on a life cycle approach. To account for emission (storage) timing, this study uses cumulative radiative forcing functions as an alternative to the time averaged 100-year GWP.

2. Methods

Annual emissions and soil carbon fluxes were analysed for two types of dairy systems in the Netherlands: a maize-based system and a grass-based system. Farm data were based on national statistics (FADN, 2014) and represent the average of 27 maize-based and 13 grass-based farms over

three years (2011, 2012, 2013). General characteristics of the two farm types are included in Table 1. All farms were located on sandy soils.

Table 1. General characteristics of the maize-based farm and the grass-based farm.

	Maize-based	Grass-based
Total land area [ha]	70.1	35.0
Percentage grassland [%]	54.2	99.2
Percentage maizeland [%]	32.2	0.8
FPCM ¹ [kg farm ⁻¹ yr ⁻¹]	1,140,140	472,981
FPCM ¹ [kg cow ⁻¹ yr ⁻¹]	9,082	8,047
FPCM ¹ [kg ha ⁻¹ yr ⁻¹]	18,361	13,449
Dairy cows [n]	123.8	57.1
Young stock [n]	80.5	34.2

¹FPCM = fat-and-protein-corrected milk

Life cycle assessment was used to evaluate GHG emissions (including CO₂, CH₄, and N₂O) of both types of farms from cradle-to-farm gate. Processes included were the extraction of raw materials to produce farm inputs, the manufacturing and distribution of these inputs, and all processes on the dairy farm. Emissions were calculated per ton FPCM, i.e. the milk is corrected to a fat percentage of 4.0% and a protein content of 3.3%. Economic allocation was used to allocate emissions to the different outputs of the dairy farm (i.e. milk and meat). In both systems, 89% of all emissions was allocated to the milk, while 11% was allocated to the meat. Also at chain level economic allocation was applied for emissions related to purchased feed products in case of multiple output systems. A detailed description of the emission calculations can be found in Dolman et al. (2014).

Soil carbon fluxes were modelled over time based on the Introductory Carbon Balance Model of Kätterer and Andrén (1999), validated by Vellinga et al (2004) for the Dutch situation. Soil organic carbon stocks increase relatively quickly in young pasture, and continues to increase at a lower rate in older pastures. Soil organic carbon levels on the farms that were included in this study were unknown. Generally, grasslands on Dutch dairy farms are over 50 years old and close to saturation (Vellinga and Hoving, 2011). In this study, however, we aimed to evaluate the maximum potential of C-seq in climate change mitigation. We, therefore, assumed that on-farm grassland was newly seeded at year one, and that no rotation was applied. This scenario reflects the most promising scenario in terms of C-seq. Based on model simulations, soil carbon stocks in the top soil (0-20 cm) were estimated to be at the level of 39.0 ton carbon ha⁻¹ at year one, and to increase up to a maximum level of 83.5 ton carbon ha⁻¹ over a period of 70 years. After 70 years, soil carbon stocks were assumed to be stabilized (Vellinga et al., 2004).

Systems were compared for a situation in which soil carbon is re-emitted after 20 years because grassland is changed back into arable land (t_{20}), or stored for an indefinite period of time (t_{ind}). Model simulations showed that soil carbon stocks build up to about 67.8 ton carbon ha⁻¹ over a period of 20 years, and will revert to a level of 39.0 t carbon ha⁻¹ over a stabilization period of 70 years when grassland is changed back into arable land. Figure 1 shows changes in soil carbon levels over time for situation t_{20} and t_{ind} . In the first year, soil C-seq equals 2514 kg C ha⁻¹, or 9218 kg CO₂ ha⁻¹. Changes in soil organic carbon stocks of land outside the farm were not considered.

Following Persson et al. (2015), the climate impact per ton of FCPM per system was modelled over time using the same expressions for relating emissions of GHGs to changes in atmospheric concentrations and resulting radiative forcing as the IPCC uses for calculating GWPs (Myhre et al. 2013). The model simulates the radiative forcing trajectory related to the emission of a certain amount of CO₂, CH₄ or N₂O. It uses the same set of assumptions regarding the radiative efficiency and atmospheric lifetimes of the GHGs as used by IPCC AR5. The model was used to show the climate impact of the production of one ton of FPCM yr⁻¹ at all time scales up to 200 years. To put results into perspective, we first calculated the climate impact related to the production of one ton of FPCM by both systems in the first year of analysis based on a 100-year GWP (Myhre et al. 2013).

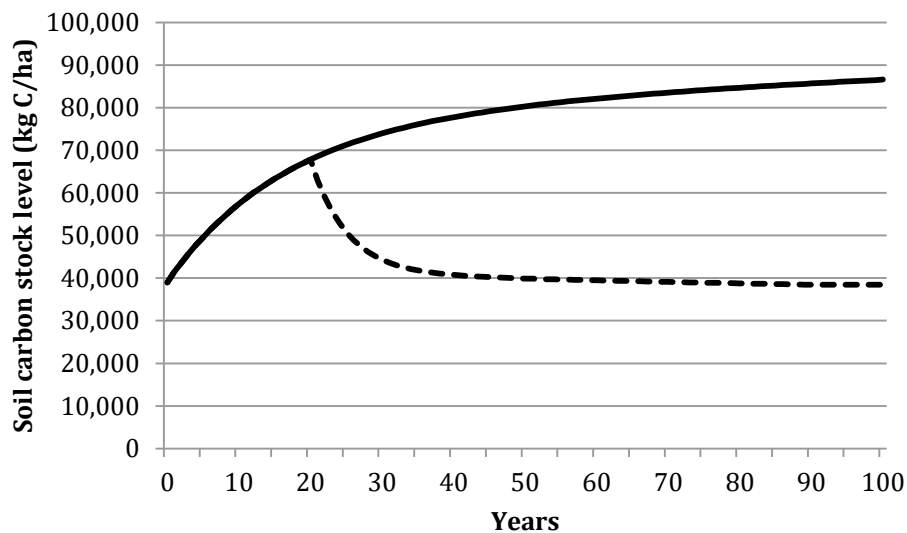


Figure 1. Changes in soil organic carbon stocks per hectare of permanent grassland (solid line) and of grassland that is changed into arable land after 20 years (dashed line). Simulations are based on the model of Katterer and Andrén (1999), adapted by Vellinga et al. (2004)

3. Results

Table 2 shows the emissions of GHGs and soil C-seq per ton of FPCM produced by the maize-based system and the grass-based system in the first year of analysis. The maize-based system results in somewhat lower CH₄ and N₂O emissions than the grass-based system. A higher proportion of maize in the diet of dairy cows generally reduces emissions of enteric CH₄ and of N₂O related to on-farm roughage production (van Middelaar et al., 2013). Based on a 100-year GWP, the maize based system (1078 kg CO₂eq t FPCM⁻¹) has a lower climate impact than the grass-based system (1126 kg CO₂eq t FPCM⁻¹) when excluding soil C-seq.

Due to a lower proportion of grassland, the maize-based system results in a lower C-seq potential per ton of FPCM in comparison to the grass-based system (Table 2). Based on a 100-year GWP, the maize based system (805 kg CO₂eq t FPCM⁻¹) has a lower climate impact than the grass-based system (528 kg CO₂eq t FPCM⁻¹) when including soil C-seq. Results are based on soil C-seq potential in the first year of analysis, which represents the maximum sequestration potential of 2514 kg C ha⁻¹, or 9218 kg CO₂ ha⁻¹.

Table 2. Emissions of greenhouse gases and soil carbon sequestration (C-seq) related to the production of one ton of fat-and-protein-corrected milk (FPCM) by a maize-based system and a grass-based system¹ in the first year of analysis.

	Maize-based	Grass-based
CO ₂ emissions [kg t FPCM ⁻¹]	360.06	359.38
CH ₄ emissions [kg t FPCM ⁻¹]	22.07	22.91
N ₂ O emissions [kg t FPCM ⁻¹]	0.38	0.52
Soil C-seq ² [kg CO ₂ t FPCM ⁻¹]	273	598

¹ Including emissions from all processes up to the farm-gate.

² Results are based on the C-seq potential of on-farm grassland in the first year of analysis, which equals 9218 kg CO₂ ha⁻¹ grassland.

Comparing the climate impact per ton of FPCM in terms of radiative forcing over time can provide insight into the importance of C-seq to reduce the climate impact of dairy production on the long term. Figure 2 shows the radiative forcing from a continuous flow of emissions related to the production of one ton FPCM yr⁻¹ from a maize-based system (upper graph) and a grass-based system (lower graph), including and excluding soil C-seq. Both graphs show a similar pattern. Soil C-seq

reduces the radiative forcing related to milk production, but this reduction is limited when soil carbon is re-emitted after 20 years (t_{20}). The reduction related to C-seq over an indefinite period of time (t_{ind}) is more pronounced in case of the grass-based system. In both systems, however, the impact of C-seq on radiative forcing is relatively low in comparison with the total radiative forcing of the systems.

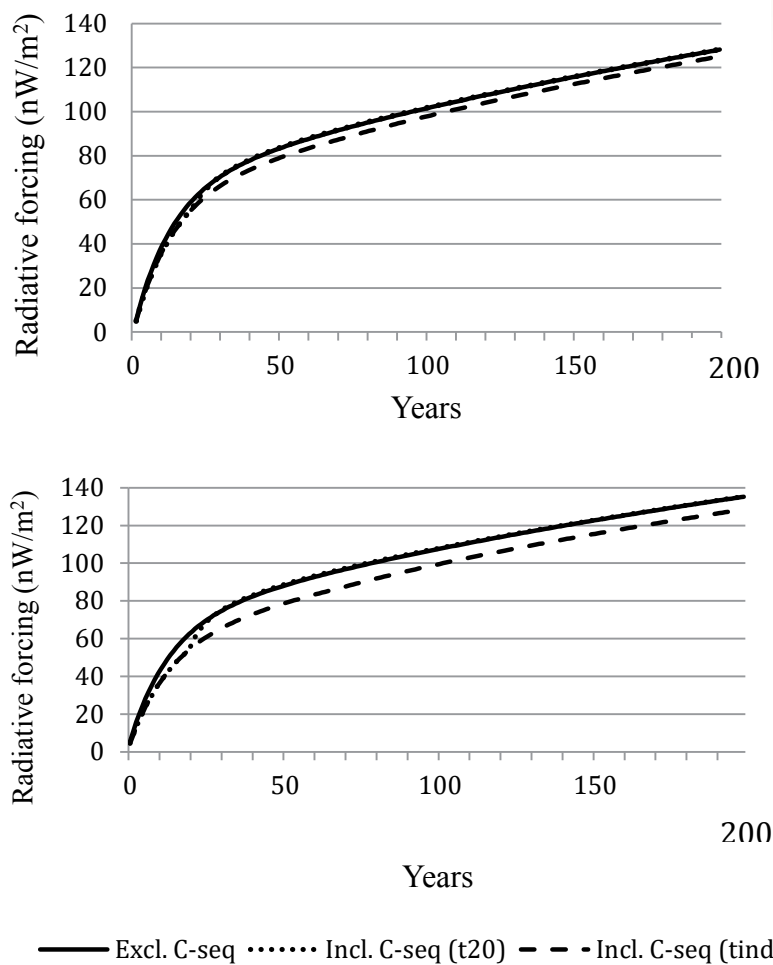


Figure 2. Radiative forcing from a continuous flow of emissions related to the production of one ton FPCM yr^{-1} from a maize-based system (upper graph) and a grass-based system (lower graph), including and excluding soil carbon sequestration. Radiative forcing is a measure of the radiative energy imbalance due to increased levels of greenhouse gases that causes the atmosphere, land and oceans to warm.

Differences between the two systems are presented in more detail in figure 3. Figure 3 shows the radiative forcing from a continuous flow of emissions related to the production of one ton FPCM yr^{-1} from a grass-based system relative to that from a maize-based system, excluding and including soil C-seq. Results show that C-seq in grassland soils temporarily favours the grass-based system, but only until soil carbon is re-emitted (t_{20}), or reaches equilibrium (t_{ind}).

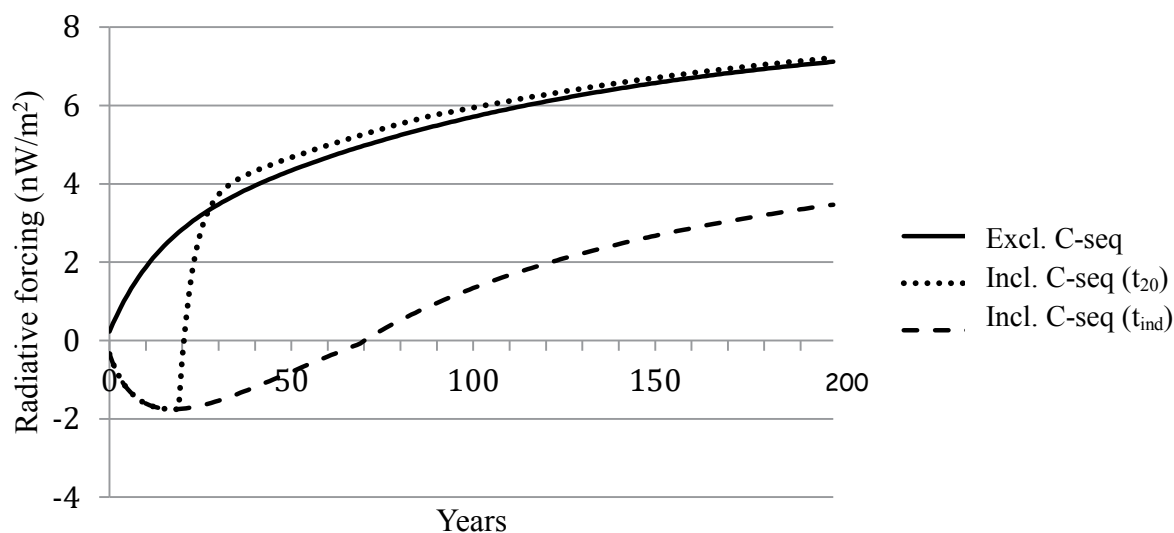


Figure 3. Difference in radiative forcing from a continuous flow of emission related to the production of one ton FPCM yr^{-1} from a grass-based system relative to that from a maize-based system, excluding and including soil carbon sequestration.

4. Discussion

Soil C-seq potential of grasslands depends on a number of factors including existing carbon stock levels. In the Netherland, most grassland soils are close to saturation (Vellinga et al., 2004). To evaluate the maximum potential of C-seq in climate change mitigation, this study modelled the most promising scenario in terms of C-seq. Taking into account current soil C-stock levels will result in lower sequestration levels and increase the climate impact of grass-based systems over maize-based systems. As soil carbon stocks become saturated when they reach equilibrium, the yearly uptake of CO_2 will level off, but the benefits in terms of climate impact remain until the carbon is re-emitted. In comparison with strategies that lower the annual emission of GHG, however, C-seq seems to be a temporary solution. Results showed that even if soil carbon in on-farm grasslands is stored indefinitely, the maize-based system results in a lower climate impact than the grass-based system after soil carbon stocks reach equilibrium.

Promoting soil C-seq will stimulate dairy farmers to increase their area of grassland over maize land. This will not only affect on-farm C-seq potential, but also other on-farm processes and production of farm inputs. Such changes could alter the level of GHG emissions as well as GHG composition related to dairy production. Different GHGs have a different atmospheric lifetime and degree of radiative forcing, and therefore, have a different climate impact over time. Moreover, soil carbon stocks stabilize over time as soil C sinks become saturated, or could be re-emitted if soil conditions change. To fully understand the importance of C-seq to reduce the climate impact of livestock systems, therefore, the impact of C-seq should be considered over time and from a life cycle perspective.

5. Conclusion

Comparing the climate impact of milk produced by a maize-based system and a grass-based system showed that soil carbon sequestration in grasslands favours the grass-based system until the point at which soil carbon is re-emitted, or reaches equilibrium. Results demonstrate the importance of including both annual emissions and soil carbon fluxes by means of a life cycle approach, and to consider the climate impact over time, when valuing the potential benefit of carbon sequestration.

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124. Soil organic carbon development in a cereal-dominated region – Impact of crop rotation diversification

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ABSTRACT

Soil organic carbon (SOC) losses from arable soil may lead to a decline in soil fertility and productivity and are a source of greenhouse gas emissions (GHG). The objective of this study was to investigate SOC changes in a current cereal-dominated crop rotation, typical for many agricultural regions in Europe, and for a crop rotation diversified with grass cultivation. SOC changes were modelled based on carbon input from crop residues and residues from grass digestion. Results show that the current crop rotation is losing SOC and emitting ca 140 kg/ha CO₂eq. The investigated diversification measure turned the soil into a carbon sink, both improving soil quality and potentially mitigating 3% of the GHG emissions of the Swedish agricultural sector if applied to the study region.

Keywords: grass crops, soil organic carbon modelling, crop residues, greenhouse gas, global warming potential

1. Introduction

Soil organic carbon (SOC) content in agricultural soils strongly influences soil fertility, nutrient holding capacity, risk for soil compaction and subsequently crop yields. The environmental impact of changes in SOC are often neglected in crop production assessment studies [1], although the impact on the greenhouse gas (GHG) balance can be significant [2]. Agricultural soils can act either as carbon sinks, or, if SOC is declining, as contributors to GHG emissions. SOC levels in Swedish agricultural soils are increasing on average [3]. However, SOC levels in the main intensively-cultivated agricultural production regions tend to decrease due to cereal-dominated crop rotations and low availability of organic fertilizers (e.g. manure) [4].

SOC content can be actively influenced by addition of carbon from e.g. intermediate crops, crop residues, manure or other organic fertilizers, or by changing crops in the crop rotation.

The aim of the present study was to investigate SOC development of a 6-year cereal-dominated crop rotation currently typical for the main agricultural regions in southern Sweden and for many agricultural regions in Europe. As an alternative, a diversified crop rotation with 2 years of grass crops was assessed. The grass was assumed to be harvested as biogas substrate and the digestate, i.e. the organic residues from the biogas process, was used as biofertilizer in the crop rotation. The impact of crop rotation and digestate application on SOC development was modelled and changes in SOC were recalculated as a GHG effect. In a parallel paper [5], a full LCA was carried out, where SOC changes were accounted for.

2. Methodology

Study region

The studied region was located in one of the most productive agricultural regions in Sweden (55°20'-56°28'N; 12°26'-14°21'E), where crop production is dominated by cereals. Livestock production is low and, subsequently, only insignificant amounts of manure are available as soil amendment. Therefore, effects on SOC of manure application were not accounted for. The initial SOC content for the investigated region ranged from 1.25 to 2.22% (mean: 1.59%) based on data from [6]. Clay content in the region ranged from 1.3 to 30.8% (mean 13.9%) [7].

Crop rotation scenarios

The effect of two different crop rotations on soil organic carbon (SOC) development was investigated: the first represented a currently typical cereal-dominated crop rotation ('current'), while in the other crop rotation, crops of year 5 and 6 were exchanged for grass production ('modified'),

Table 1. Based on data of current crop production in the study region [8], the area cultivated under the current crop rotation is 197,000 ha [4].

Table 1. Normal dry matter yields [kg/ha] including pre-crop effects for crops grown in the studied regions.

Crop rotation	Current	Yield	Modified	Yield
Year 1	Winter wheat	6854	Winter wheat	6054
Year 2	Sugar beets	12959	Sugar beets	12959
Year 3	Spring barley	5072	Spring barley	5072
Year 4	Winter wheat	5654	Winter wheat	5654
Year 5	Spring barley	4272	Grass, year I	9686
Year 6	Winter oilseed rape	3527	Grass, year II	6317

Crop production

In the modified crop rotation, grass is assumed to be cultivated as biogas feedstock with relatively short growth periods (3 harvests per year). Grass is undersown in the previous crop, i.e. winter wheat. The first production year is a full production year, where the grass is harvested three times, resulting in high biomass yields combined with high methane potentials. The second year is the break year, when the crop is ploughed up after the second harvest in order to allow for an autumn crop to be established. In the current crop rotation, all crops are assumed to be fertilized with mineral fertilizer, while in the modified crop rotation, mineral fertilizer is partly replaced by digestate from biogas production from grass. Besides plant nutrients, organic carbon was therefore added to the soil.

Crop yields

For grass crops, normal harvest level data for high-intensity production was not available, since official statistics include even low and medium-intensity production systems, as well as organic production systems [9]. Therefore expectable biomass yields have been estimated based on variety field experiments hosted by the Field Research Unit (FFE) at the Swedish University of Agricultural Sciences. Variety experiments for the above mentioned grass species were chosen, since in these experiments grass species are cultivated in a way to demonstrate the biomass yield potential and are fertilized with high rates of nitrogen, potassium and phosphorous, so that availability of plant nutrients will not limit plant growth.

Biomass DM yields were based on a grass crop mix of perennial rye-grass, meadow fescue and timothy-grass, which were assumed to represent 40, 30 and 30% of the biomass in the field. No biomass yield was attributed to legumes such as red and white clover, since the high nitrogen fertilization levels expected in intensively cultivation of grass-clover crops likely results in marginal shares of legumes [10].

For each grass species, variety experiment results available from FFE were extracted for the period of 2004-2014 for each harvest occasion for field experiments harvested 3 times per year and two, two and three consecutive years for the study region.

Absolute biomass yields were calculated for each grass crop species using the mean biomass yield across all varieties. These biomass yields were adjusted for each harvest occasion to represent harvest by standard field machinery (Table 2) using the following empirical relation [11]:

$$\text{Recovery coefficient [\%]} = 1.3828 \cdot \text{Biomass yield} \left[\frac{\text{t}}{\text{ha}} \right] + 64.603 \quad \text{Eq. 1}$$

Table 2. Calculated biomass dry matter yields [kg/ha] for grass crops in the modified crop rotation.

Cut	Year 1			Year 2			
	I	II	III	Total	I	II	Total
	4559	2826	2301	9686	4317	2000	6317

Crop yields for the food crops of the crop rotations presented above were taken from official statistical sources [9] in the form of standard biomass yields,

Table 1. Pre-crop effects for winter oilseed rape, grass, oat and sugar beets were taken into account for the final yields in the crop rotations [12]. In the sensitivity analyses described below, cereal and oilseed dry matter grain/seed yields were assumed to be positively correlated to the SOC content to an extent of 35 kg/t of SOC change based on Bauer *et al.* [13].

Crop residues

Crop yields as presented above were used to calculate the amount of crop residues (straw, stubble, roots and extra root biomass) added to the soil. A linear correlation between harvestable biomass (i.e. grains, seeds, beets, above-ground biomass) and remaining residues in the form of fixed mass ratios for the different plant parts [2] were used to calculate residue amounts, Table 3.

Table 3. Dry matter coefficients for crop residues relative to the harvested biomass dry matter yield as used for modelling soil organic carbon changes. Coefficients were recalculated from Nordic data [14-21].

Crop	Straw/grass		Stubble	Roots	Extra root	Crop residue input	
	field	recovered				Above-ground	Below-ground
Grass crops, establishing year	1.25	1.00	0.00	0.00	0.13	0.00	0.13
Grass crops, full production year	1.25	1.00	0.00	0.00	0.41	0.00	0.40
Grass crops, breaking year	1.25	1.00	0.25	1.48	0.96	0.25	0.75
Oats ^a	0.50	0.32	0.17	0.43	0.28	0.50	0.75
Spring barley	0.35	0.18	0.18	0.32	0.21	0.35	0.53
Spring oilseed rape ^a	0.90	0.58	0.31	0.31	0.20	0.90	0.52
Sugarbeet	0.30	0.27	0.03	0.01	0.01	0.30	0.02
White mustard ^a	0.67	0.00	0.67	0.51	0.33	0.67	0.83
Winter oilseed rape	0.92	0.78	0.14	0.21	0.14	0.92	0.35
Winter wheat	0.57	0.43	0.14	0.31	0.20	0.57	0.52 ^b

^a Used for model calibration based on the long-term field experiment data only. ^b Only wheat. In the modified crop rotation, the undersown grass root biomass is included and the value is 0.56-0.58.

Swedish studies support the above crop residue coefficients that result in high biomass inputs from root and extra root material, especially in grass crops. For root biomass yields of grass crops, Swedish studies fitting long-term soil carbon measurements to a soil carbon model suggest a constant dry matter contribution of 6 t/ha [22]. However, in this study, a proportional root dry matter biomass development was assumed with a ceiling value of 6 t/ha.

Carbon addition

The carbon addition from crop residues was calculated assuming a carbon content of 45% of DM in crop residues (Table 4). The amount of carbon added to the soil via digestate was calculated as the amount of carbon removed as methane and carbon dioxide in the biogas process subtracted from the initial amount of carbon in the grass biomass (Table 4), assuming a volatile solids (VS) content of 91% and a methane potential of 334 L/kg VS before storage losses of 9.5% of DM and 6.7% of methane potential [4].

SOC modelling

The ICB model used in this study is a two-pool model with individual mineralization rates [23]. Added organic material enters the young carbon pool, from where a fraction described by the

humification coefficient (h) continues relatively quickly (50% within less than one year) into the old carbon pool. The other fraction is mineralized and carbon is released to the atmosphere as carbon dioxide. The carbon in the old pool has a much lower mineralization rate (50% within approx. 100 years) compared to the young carbon pool and is therefore considered much more stable than that of the young carbon pool. The annual average SOC change was calculated as the average SOC change over the period of 40 years after the change in crop cultivation [2].

Table 4. Carbon input [kg/(ha·a)] for individual crops and scenarios. CR = crop rotation.

Yea	Cron	Scenario	Cron residues innut		
			Abovegrou C input	Belowgrou C input	Digestat C input
1	Winter wheat	current CR	1764	1603	0
		modified CR	1661	1509	0
		modified CR +	1661	1509	608
2	Sugar beet	current CR	1739	99	0
		modified CR	1739	99	0
		modified CR +	1739	99	0
3	Spring barley	current CR	788	1181	0
		modified CR	788	1181	0
		modified CR +	788	1181	0
4	Winter wheat	current CR	1456	1322	0
		modified CR	1456	1417	0
		modified CR +	1456	1417	591
5	Spring barley	current CR	661	991	0
		Grass	0	1755	0
		Grass	0	1755	975
6	Winter oilseed	current CR	1459	563	0
		Grass	901	2700	0
		Grass	901	2700	592

Humification coefficients

Humification coefficients (h) of 0.27, 0.35, 0.12 and 0.15 were used for grass digestate, root biomass and for aboveground crop residues of grass and other crops, respectively [19]. Poeplau et al. [24] showed that the soil clay content impacts the humification coefficient for aboveground crop residues:

$$h_{\text{aboveground}} = -0,044 + 0,0036 * \text{Clay content} [\%] \quad \text{Eq. 2}$$

For the base case, the humification coefficient corresponding to the minimum, mean and maximum clay content in the study region was calculated to be 0.00, 0.01 and 0.07, where negative outcome was zeroed.

Model calibration

The SOC model was calibrated against data from a long-term field experiment, located within the study region (Ekebo, 55.99° N 12.87° E, 17.8 % clay, 1.43 kg/dm³ bulk density [25,26]). Data on annual yields and SOC content determined regularly, was available for two different crop rotations with 16 different fertilization regimes for the years 1962-2014. Calibration was carried out by changing the mineralization rate of the old carbon pool in order to maximize the coefficient of determination (R²). Resulting mineralization rates with and without adjustment of h were 0.0095 and 0.0100 per annum, respectively.

GHG changes caused by SOC changes

GHG emissions were calculated based on SOC released as CO₂ and nitrous oxide (N₂O) emissions. N₂O emissions corresponded to 1% of N released by soil organic matter (SOM) mineralisation, assuming a C/N ratio of 10 [27]. Accordingly, 1 kg SOC change corresponded to 4.1 kg carbon dioxide equivalents (CO₂eq).

3. Results

At mean initial SOC and mean clay content, SOC content in the current crop rotation was decreasing by approx. 34 kg/(ha·a), while at maximum initial SOC content the corresponding value was ca 165 kg/(ha·a), Figure 1. At an initial SOC content of 1.4%, the SOC content remains stable under the current crop rotation and unchanged crop yields. Modifying the crop rotation led to a SOC-maintaining state at mean initial SOC of 2.1%, which increased to 2.6% when also the digestate was returned to the fields.

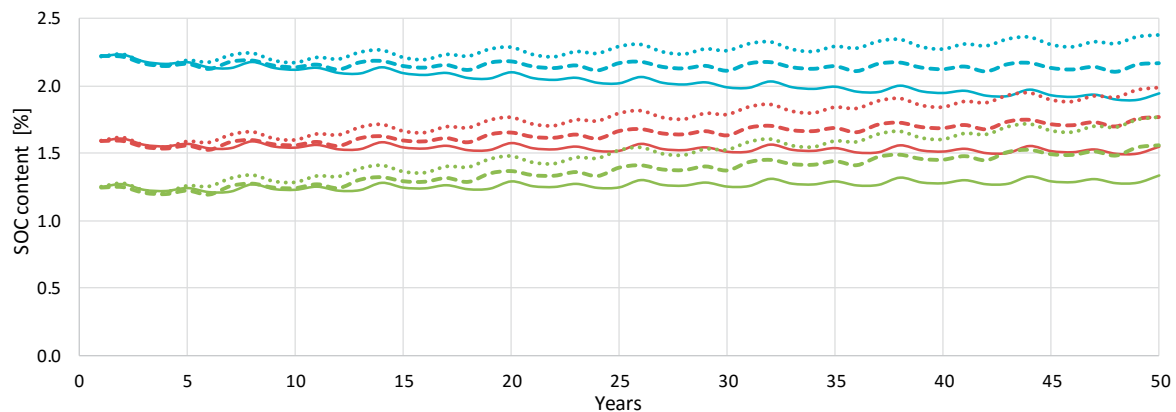


Figure 1. SOC development under different initial SOC contents (maximum: blue; mean; red; minimum: green, in the study region) for the current crop rotation (solid lines), the modified crop rotation with no digestate addition (dashed lines) and with digestate addition (dotted lines).

Soil clay content had a substantial impact on the SOC development (Figure 2). SOC development for the low and mean clay content were only marginally different, since both resulted in the use of nearly the same humification coefficient.

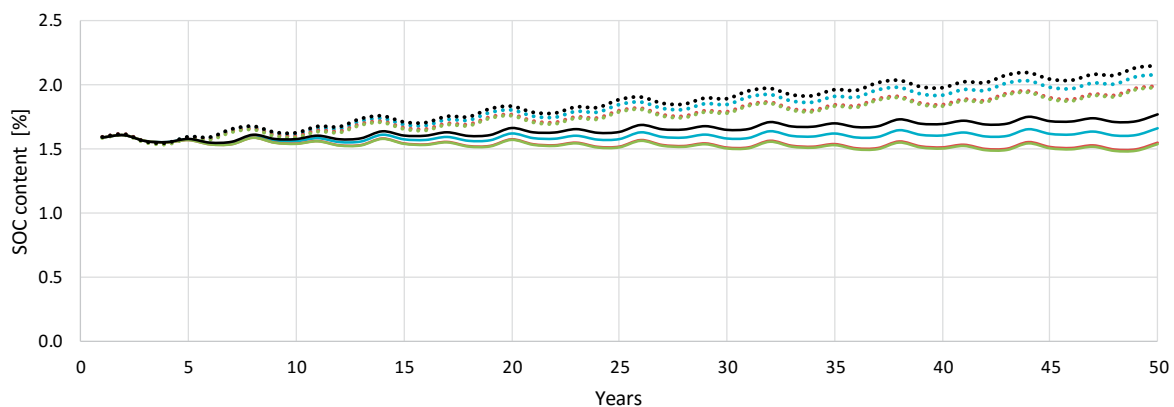


Figure 2. SOC development at mean initial SOC under different soil clay contents (high: blue; mean: red; low: green) influencing aboveground carbon input stabilization and results disregarding clay content (black) for the current crop rotation (solid lines) and the modified crop rotation, including digestate application (dotted lines).

Average annual GHG emissions based on SOC development differed considerably between crops and crop rotations (Figure 3). In the current crop rotation, 0.14 t/(ha·a) CO₂eq are emitted. In the modified crop rotation, grass contributed with a GHG mitigation of 2.6 t/ha for the two years of grass cultivation, where contribution in a full production year (year 5 of the crop rotation) was high, but even higher in a break year (year 6), when all root biomass was accounted for. Changing from the

current to the modified crop rotation resulted in an SOC increase of 140 kg/(ha·a) on average over the whole crop rotation, which corresponded to a CO₂ mitigation of 0.58 t/(ha·a). The additional GHG mitigation by sequestering carbon in the form of digestate was large, contributing a mitigation of another 0.46 t/(ha·a) as averaged over the crop rotation. Assuming yield sensitivity to changes in SOC, the net GHG flux was found to be amplified by 2.8, 3.9 and 5.8% on average for the whole crop rotation in the current crop rotation, the modified crop rotation without digestate and the modified crop rotation with digestate application.

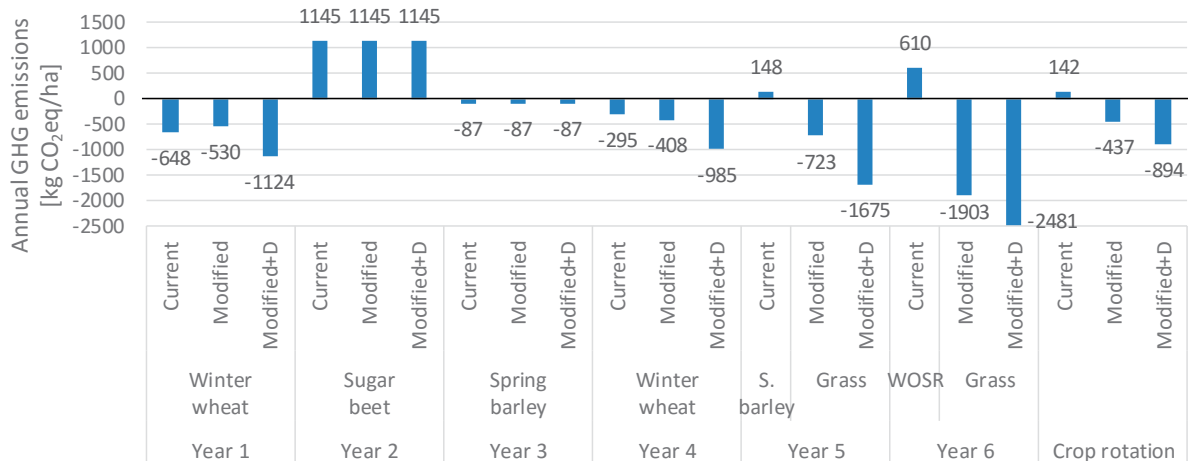


Figure 3. Annual GHG emissions [kg CO₂eq/ha] at mean initial SOC content for individual crops and the whole crop rotation according to scenario. Negative emissions are carbon sequestered in the soil. “+D” = including digestate application.

4. Discussion

SOC conservation

Loss of soil organic carbon (SOC) is one of the main processes threatening soil fertility throughout the EU [28,29]. A negative trend of SOC content in agricultural soils has been reported for many European countries [e.g. 30,31,32], but not for Sweden [3], where the increasing grass cultivation area was identified as one of the major causes to this development. As the results have shown, cereal-dominated crop rotations are prone to loose SOC even at high crop yields. In the main agricultural regions in Sweden where such crop rotations are typical, availability of organic fertilizers is usually marginal and grass is often not cultivated due to lack of applications as feed. A biogas plant would give grass a market value as feedstock for production of renewable vehicle fuel and additionally deliver nutrient-rich organic fertilizer as soil amendment.

Results of the present study confirm the positive impact on SOC development when grass crops were included in the crop rotation. Contrary, the negative SOC development of land under cereal-dominated crop rotations underlines the importance of measures to stabilize or even increase SOC stocks in agricultural soils in order to maintain soil productivity. Accounting for the clay impact led to more conservative estimations of SOC development. However, a verification of this effect was not attempted in this study.

GHG mitigation

In the evaluated region with a cereal-dominated crop rotation, SOC losses are twice as high as for average mineral soils in Sweden [33], and cause an emission of GHGs corresponding to the combustion of 45 L/(ha·a) of fossil diesel. Earlier studies have shown that the combined CO₂ mitigation effect from grass production and utilization of the resulting digestate as biofertilizer corresponds to a large share of the overall GHG mitigation effect [2]. If implemented throughout the study region, the modified crop rotation would have a GHG mitigation potential from grass cultivation and digestate use of approx. 0.2 Mt/a, which corresponds to 3% of the total Swedish GHG emission from the agricultural sector in 2013 [33]. In a similar vehicle fuel production system using grass as feedstock, replacement of mineral fertilizer with digestate and replacement of fossil fuels

with biogas vehicle fuel led to additional substantial effects on the GHG balance [2,34]. On the other hand, GHG emissions from indirect land use change caused by producing the replaced food crop elsewhere need to be accounted for. Assessment of these effects was outside the scope of this study, but are presented in a parallel paper [5], showing GHG mitigation effects even when applying a full LCA.

5. Conclusions

From a soil quality perspective, a negative SOC development is not sustainable in the long run, threatening food security and turning arable land into a source of GHG emissions. Cultivating grass two out of six years, using the grass for biogas production and recirculating the produced biofertilizer in the crop rotation proved to be a sufficient measure to stop and even reverse this trend by turning arable land into a carbon sink. Besides soil quality improvements, this measure gave a large benefit when analysed from a climate perspective.

6. Acknowledgements

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142. Arable land as carbon sink – a regional case study on greenhouse gas emission impact of diversifying cereal based crop rotations

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ABSTRACT

The objective of the study was to investigate the development of soil organic carbon (SOC), with focus on greenhouse gas emission impacts, for conventional cereal based crop rotations and for modified crop rotations that was diversified by including grass. The case explored was stockless farming on high-yielding arable land with unvaried cereal based crop rotations, with a high risk of SOC losses. The demand for grass as cattle feed was low in the region, and the grass produced in the modified crop rotations was assumed to be used for the production of biogas, which in turn was used to replace fossil fuels. The region with the highest wheat yields in Sweden was chosen for the study. The current cereal based crop rotation was shown to give a release of SOC corresponding to 140 kg CO₂-eq ha⁻¹ a⁻¹. The diversification of the crop rotations gave a strong impact on SOC-build up, where the arable land was turned into a carbon sink, and carbon sequestration corresponded to avoided emissions of 900 kg CO₂-eq ha⁻¹ a⁻¹ as average for the six year crop rotation. Diversifying unvaried crop rotations has many benefits, and here it was shown how the introduction of grass could be a tool to convert mineral soils to carbon sinks. The SOC loss in the present crop rotation is not sustainable from a soil quality perspective. For the proposed modified crop rotation, the SOC impact is large and relevant to include in broader sustainability assessments of arable land use.

Keywords: LCA, grass, cultivation systems

1. Introduction

It is increasingly being stressed that for efficient climate mitigation, we must both look at replacing fossil fuels and at possibilities for carbon sequestration (e.g SEPA (2015)). Agriculture is a sector that has a capacity for soil carbon sequestration, and where, especially in developing countries, this has been presented as a tool for increasing productivity (Lal, 2010). Due to increasing specialization, intensification and reduced use of bio-fertilizer, we have organic matter losses from arable land also in the highly productive regions (SEPA, 2015), and 45% of the soils in the EU have low or very low (0-2%), and declining, soil organic carbon (SOC) content (Jones *et al.*, 2012). A region located in one of the most productive agricultural regions in Sweden (55°20' -56°28'N; 12°26' -14°21' E), where crop production is dominated by cereals, was selected for the present study. The currently dominating six-year crop rotation is shown in Figure 1. In this region, 61% of the land area is arable land, and 56% of this, or 197 000 hectare (ha), was on average 2010-2014 under the current cereal dominated crop rotation (Olsson, 2015). Livestock production is low and, subsequently, only insignificant amounts of manure are available as soil amendment.

The SOC content for the investigated current crop rotation was shown to decline (Prade, 2016). From a soil quality perspective, this development is not sustainable in the long run, and also turns arable land into a source of CO₂ emissions, which is why soil carbon stocks are included in the national greenhouse gas inventory reports within EU since 2013 (SEPA, 2015). Stopping and even reversing this trend by transforming arable land into a carbon sink requires powerful measures. A recent investigation, however, showed that in contrary to the development in the neighbouring countries, the SOC content is increasing in many Swedish counties (Poeplau *et al.*, 2015). This was shown to be due to an increase in the share of grass cultivation on agricultural land, and the main driver for this development was the increasing horse population (Poeplau *et al.*, 2015). In addition, large areas of long term set-aside land are also covered with grass, contributing to the SOC build-up.

The approach investigated here was to maximize carbon input in the highly productive crop rotation by cultivating grass two out of six years (Figure 1). This would allow the majority of the crop production to continue. Integration of grass-clover in crop rotations of annual commodity crops has

previously been shown to enhance the delivery of soil ecosystem services (Albizua *et al.*, 2015). The grass was used as an energy crop for biogas production since there is little demand for grass for animal feed in the region, and this choice was also made to exemplify the potential conflict as energy crops replace food / feed crops on arable land. The biogas produced was used as fuel replacing diesel, and the produced by-product, which retains all the nutrients of the grass, was used as biofertilizer in cultivation, where it replaced mineral fertilizer.

The current and modified scenarios were evaluated from a climate perspective, taking SOC changes into account. The results presented here are part of a larger study, covering several regions and other environmental impacts, as well as cost (Björnsson *et al.*, 2016). The data selected for the present paper focus on how a regional perspective and LCA methodology is applied to include SOC in the evaluation of assessing arable land use sustainability from a greenhouse gas perspective. As has been suggested by Hildingsson and Johansson (2016), policy measures that are designed to include several sustainability concerns need to take local contexts into account, and respond to new knowledge. Local conditions and spatial perspectives are, however, just as SOC changes, neglected in many crop production LCAs (Brandao *et al.*, 2011).

The aim of the study was to present facts on SOC and greenhouse gas emissions for arable land in a highly productive region, and to evaluate a possible future scenario aiming at improving carbon sequestration. The point to be made was that these aspects are relevant to include in assessments of sustainability in the use of arable land.

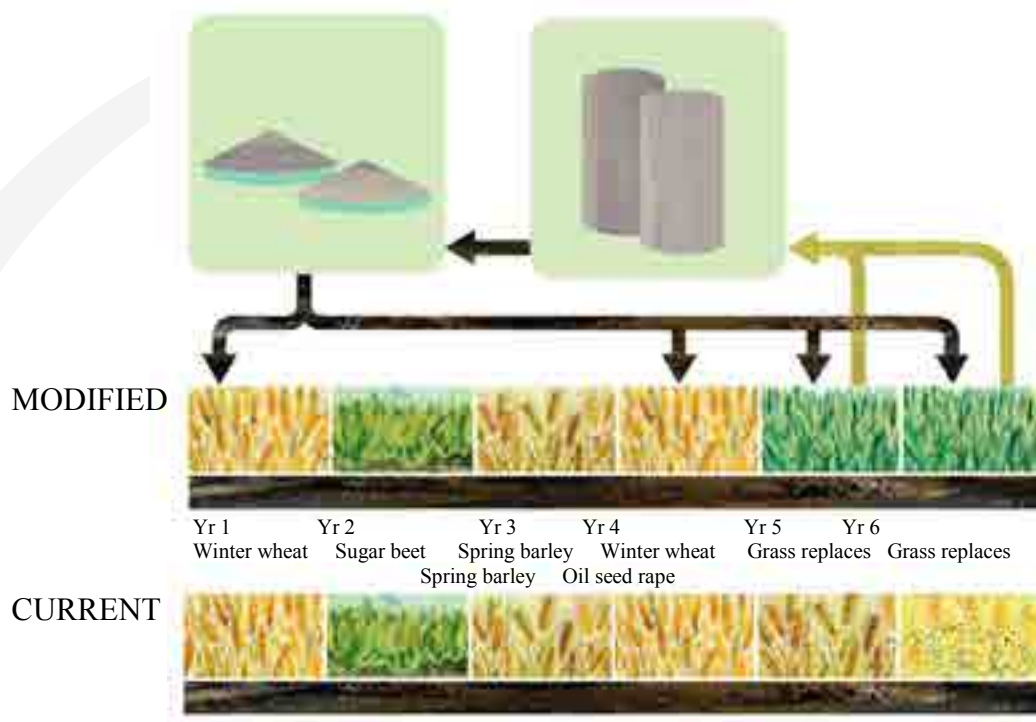


Figure 1: The current six-year crop rotation (lower) and the modified crop rotation (upper), where grass is introduced year 5 and 6. The grass is used as feedstock for biogas production, and the by-product is recirculated as biofertilizer for winter wheat and grass. Illustration: Anna Persson.

2. Methods

Selection of the Swedish region with highest wheat yields and share of cereals in current typical crop rotations was based on inventory and processing of national statistics from the Swedish Board of Agriculture (SJV), Statistics Sweden (SCB) and the Swedish Energy Agency (SEA). A range of approaches and method applications were used, and detailed method descriptions can be found in

Björnsson *et al.* (2016). The ICB model, which is used for SOC inventory in Sweden (Andrén and Kätterer, 1997), was used to model SOC development, and details on the SOC modelling is presented by Prade (2016).

Assessment of greenhouse gas emissions was based on life cycle assessment methodology (ISO, 2006) with a functional unit of an average hectare of arable land in the crop rotation. The impact was given as global warming potential (GWP) in CO₂-equivalents (CO₂-eq), using the equivalency factors of IPCC (2007). The GWP was calculated for the current crop rotation and for the modified system including grass in the crop rotation and using it for biogas production (Figure 1). The net result was calculated as the impact of the change from the current system, and systems expansion was applied to include the impact of changes in product output between the systems. All background details on crop yields, cultivation inputs and emissions, emissions related to the production of biogas and biofertilizer application, and the data used for the systems expansion can be found in Björnsson *et al.* (2016).

3. Results

Implementing the modified crop rotation in the investigated region would mean using 66 000 ha of arable land for grass production. This would give a biogas production corresponding to 4.8 PJ a⁻¹, which is more than the current production of biogas as vehicle fuel in Sweden (3.7 PJ in 2014) (SEA, 2015b). The crops lost compared to the current crop rotation would amount to 220 000 t DM a⁻¹ cereals and 116 000 t DM a⁻¹ oil seed rape. This can be compared to the Swedish production statistics for the last three years, where total cereal production has been 5.0-5.8 million t a⁻¹, whereof 2.2-2.5 million t has been used as animal feed and 1.0-1.9 million t has been exported (SJV, 2016). The use of rape seed based biodiesel (RME) in Sweden 2014 was 15 PJ, whereof 7% was produced from oil seed rape cultivated in Sweden, corresponding to 80 000 t DM rape seed (SEA, 2015a).

The GWP for the current and modified crop rotations are shown in Figure 2. While emissions related to cultivation inputs decrease slightly in the modified rotation, the field N₂O emissions increase, both due to the biofertilization and because grass contributes with nitrogen containing crop residues. This emission increase is, however, balanced by the annual SOC effect, where the crop rotation as a whole gives carbon sequestration. The positive impact on SOC of the increased addition of biomass (as biofertilizer and crop residues) thus largely outweighs the negative increase in N₂O emissions.

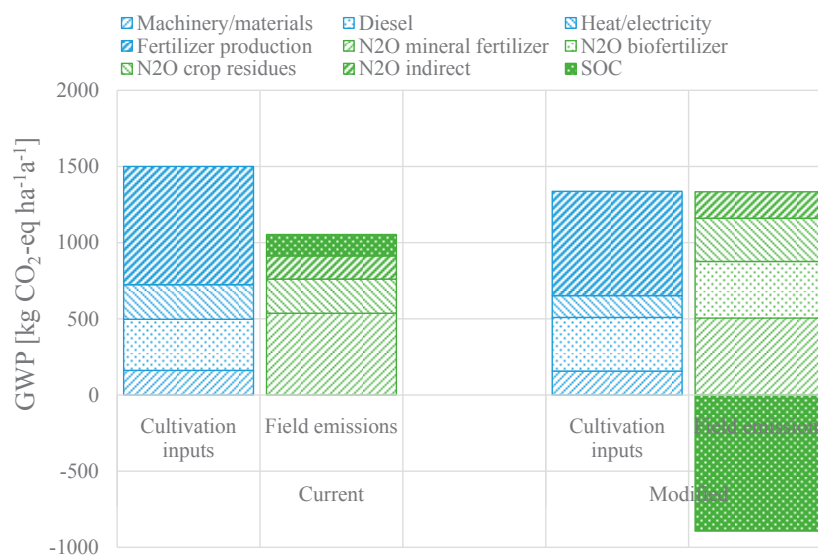


Figure 2: Greenhouse gas emissions in cultivation for current and modified crop rotations.

The net impact in cultivation of the change from the current to the modified crop rotation is shown in Figure 3 (left bars), totally an avoided emission of 0.8 t CO₂-eq ha⁻¹ a⁻¹. The emissions related to the biogas production are added, including the upgrading of the biogas to vehicle fuel quality, the

distribution to gas filling stations and end use emissions. In the systems expansion, the avoided emissions when replacing diesel with biogas for transport are included, giving net avoided emission of 1.2 t CO₂-eq ha⁻¹ a⁻¹ for the production and use of biogas. The emissions from cultivating the lost cereals and oil seed rape elsewhere in the region is, however, also added through systems expansion. The net impact, when the replacement of lost crops is added as a drawback for the fuel production, is an avoided emission of 0.7 t CO₂-eq ha⁻¹ a⁻¹ for the biogas part of the system (Figure 3, right bars).

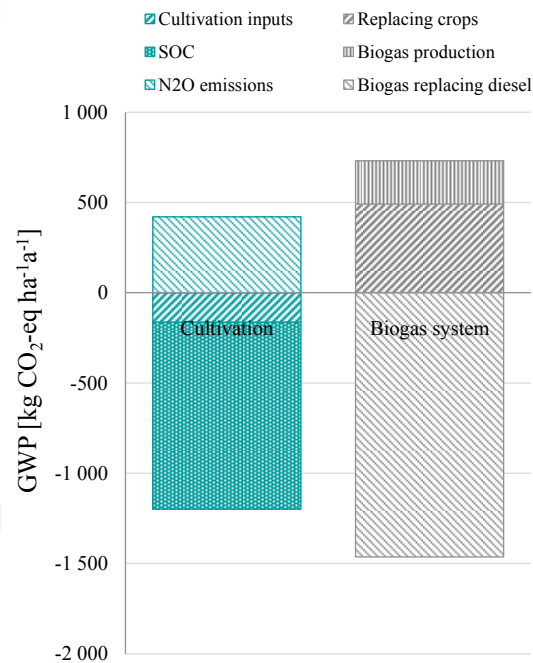


Figure 3: Greenhouse gas emission impact of changing from the current to the modified system.

4. Discussion

The fact that the four remaining food/feed crops in the crop rotation still give a net release of SOC (Prade, 2016) is outweighed by the large SOC impact of the grass and the biofertilizer, giving a carbon sequestration in the modified crop rotation as a whole. Mineral agricultural soils in Sweden on average lose SOC corresponding to 60 kg CO₂-eq ha⁻¹ a⁻¹ (SEPA, 2015). The current cereal dominated crop rotation studied here gives a loss twice that, 140 kg CO₂-eq ha⁻¹ a⁻¹. The studied modification turned the crop rotation as a whole to a carbon sink, giving a carbon sequestration of 900 kg CO₂-eq ha⁻¹ a⁻¹. The field emission of nitrous oxide would increase in the modified crop rotation, and if SOC changes were not taken into account in the assessment, the modified crop rotation would appear to give higher greenhouse gas emissions. Including the fact that the arable land as a whole in the modified crop rotation is turned into a carbon sink, however, gives a total decrease in emissions of 0.8 t CO₂-eq ha⁻¹ a⁻¹. This corresponds to a 30% decrease in GWP compared to the emission from the current crop rotation.

Implementing the modified crop rotation in the investigated region on all arable land under the current cereal dominated crop rotation (197 000 ha) would give an emission decrease of 0.16 Mt CO₂-eq a⁻¹ in cultivation. This can be compared to the present annual greenhouse gas emission from agriculture in Sweden of 10 Mt CO₂-eq (2013), where SOC losses from arable land represented 2 Mt CO₂-eq (SEPA, 2015). The cereal production that would be lost due to the introduction of grass would correspond to 10% of the present domestic cereal production used as animal feed. The lost rape seed production approximately corresponds to the domestic production of oil seed rape that in 2014 was used for biodiesel production, supplying 0.4% of the domestic vehicle fuel (SEA, 2015a). Instead, biogas corresponding to 1.5% of the present domestic vehicle fuel production would be produced, giving a climate benefit in the same range as the cultivation, 0.14 Mt CO₂-eq a⁻¹. The emission of

producing these lost crops elsewhere in the region are then included, but without adding any iLUC impact. The demand for animal feed (related to meat/milk production) is declining in Sweden, the present area of set-aside arable land in Sweden exceeds the area used for grass production in the modified scenario threefold (Olsson, 2015), and the area of recently abandoned farm land (last 15 years) amounts to 90 - 150 000 ha (Olofsson and Börjesson, 2016). Still, to decrease cereal/ oil seed rape production to allow for SOC increasing measures could cause iLUC, and the pros and cons of this should be further investigated. A research project on this issue (Land use change from a Swedish perspective) is ongoing (<http://f3centre.se/research/program/Biofuels-from-biomass-from-agricultural-land-land-use-change-from-a-Swedish-perspective>).

5. Conclusions

The overall aim of this study was to produce data to improve understanding of the broad perspective needed for decisions on sustainable use of arable land. The loss of soil carbon is not sustainable in the long term, and measures must sooner or later be taken to reverse this trend. A sustainable use of arable land should give the lowest possible contribution to greenhouse gas emissions while food production is safeguarded in the long term. To introduce grass cultivation in cereal crop rotations was shown to be one approach that can reverse the present carbon loss from arable land. The modified scenario investigated here would contribute to reduced greenhouse gas emissions, both in crop cultivation and in the transport sector.

The results of the present study stresses the importance of taking local conditions and spatial perspectives into account to avoid counterproductive measures when sustainability criteria are to be formulated. While SOC losses may not necessarily directly lead to decreasing crop yields, these losses render arable land to be sources of carbon emissions. Therefore, the benefits from a climate perspective of converting the crop rotations as a whole to carbon sinks should be taken into consideration when the sustainability in arable land use is discussed.

The modification, where the produced grass is used for biofuel production, also illustrates the conflict in the use of arable land for food/feed or bioenergy production. Concerns about food shortage has resulted in restrictions within the EU on the share of biofuels based on crops grown primarily for energy purposes on arable land, which should constitute no more than 7% of the final consumption of energy in transport in the member states after 2020 (EU, 2015).

The complexity in sustainability assessments of different uses of arable land requires an improved methodological framework. Future sustainability assessments for arable land use should include the dimension of soil carbon increase and decrease, e.g. by including a soil-specific bonus for crops contributing to increasing soil organic carbon. It is important to broaden the perspective, and to make sufficiently wide-ranging sustainability analyses of such complex systems as the use of arable land, taking local conditions and spatial perspectives into account in future policies.

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17. Modeling Potential Ecotoxicity Impacts due to Copper Fungicide Use in Vineyards

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ABSTRACT

Copper fungicides are frequently used in vineyards to protect grapes against mildew. Although it is well known that its toxicity depends on the surrounding environment, until now, impacts of pesticides are estimated without taking the spatial differentiation into account. The aim of this study was the inclusion of spatial differentiation in the assessment of freshwater and terrestrial ecotoxicity impacts and compare copper fungicides with the most common fungicide active ingredients (a.i) used in Europe for grape and wine production.

The potential ecotoxicity impacts derived from the use of copper-based fungicides were calculated for seven European water types and three soil scenarios. Impacts of six different active ingredients were evaluated using USEtox 2.0 as a characterization model.

The resulting freshwater ecotoxicity impact scores for the six different fungicides showed up to 4 orders of magnitude of variation and ranking copper base fungicides as the substance with higher ecotoxicity potential. This fact indicates that copper toxicity depends in a large way of the water chemistry. In the case of terrestrial ecotoxicity the impact scores show up to 3 orders of magnitude of difference (1.15×10^2 to 2.70×10^4), this is associated mainly with the variability in the soil organic carbon, and the effect of the soil pH in the bioavailability of copper.

The significance of site-dependent and special differentiation conditions are discussed, as well as options for better addressing the impacts of inorganic fungicides to be able to compare different farming systems especially because of their importance in organic agriculture.

Keywords: life-cycle assessment (LCA), organic agriculture, spatial differentiation, pesticides

1. Introduction

Copper fungicides such as Bordeaux mixture has been used in viticulture as a plant protection product against fungal diseases since the 18th century. Indeed, it was the first fungicide to be used on a large scale worldwide. Even today the fungicides allowed under organic standards that are efficient against *Plasmopara viticola*, the causal of grapevine downy mildew, are based on copper hydroxide and copper sulphate (Vitanovic, 2012). Given the extensive use of this fungicides over the years, from the ecotoxicological perspective, but not only, is important to determine the different forms of copper in the environmental compartments. The bioavailability of the copper applied in vineyards, the total amount of copper available to biota and its mobility are one of the most important factors in determining environmental impacts. Whether in the soil or aquatic environments, the free ion can interact and be transported by several processes. These processes are governed by the chemical nature of the metal, soil and sediment particles, and the pH of the environment. Therefore, its toxicity needs to be assessed taking into account the interactions with the surrounding environment (Peña et al. 2016).

Life Cycle Assessment (LCA) is a comprehensive methodology which allows a better understanding of the environmental impacts of the whole production chain. It is increasingly used to assess the environmental sustainability of agricultural and food products and is seen as a useful tool to evaluate ecotoxicity impacts of production systems (Roy et al., 2009). However, in LCA application to agricultural systems and especially to viticulture, one of the main drawbacks is the lack of agreement on how to handle Cu products. There is a gap in the quantification of Life Cycle Inventory emissions, as well as interim characterization factors in the stage of impact assessment.

The focus of this work is the inclusion of spatial differentiation in the assessment of potential freshwater ecotoxicity impacts (PFEIs) and potential terrestrial ecotoxicity impacts (PTEIs) and

compare copper-base fungicides with the most common fungicide active ingredients (a.i) used in Europe for disease control in vineyards.

2. Methods

To test the method and hypothesis of the importance of spatial differentiation for the potential ecotoxicity impacts derived from the use of copper-based fungicides the PFEIs was calculated for seven EU water types (Table 1) and PTEIs for three soil scenarios. The three soil scenarios were built from 760 different soils assessed by Owsianiak et al. (2013) as three extreme archetypes, based on pH, soil organic carbon and clay content. The PFEIs of six different active ingredients used to control Downy mildew were also estimated. The dose of active ingredients used was based on the recommended dose for protecting vineyards in Europe/Spain (MAAMA, 2016) (table 2). Furthermore, three application rate scenarios (3, 2 and 1,5 kg/ha) for copper were derived from the most restrictive use of copper in viticulture and the objectives for copper use in organic agriculture (EGTOP, 2014)

Table 1: Main properties for water archetype scenarios (Dong et al. 2014)

Scenarios	pH	DOC	Hardness
Wat1	High	Med	High
Wat2	High	Med	Med
Wat3	High	Low	Med
Wat4	Med	High	Med
Wat5	Med	Low	Med
Wat6	Med	Low	Low
Wat7	Low	Med	Low

To provide emission estimates, LCA practitioners and developers proposed generic assumptions regarding the varying percentages of applied active ingredient emitted to environmental compartments (Rosenbaum et al., 2015). In this case, we assumed a static percentage distribution of pesticide a.i (45% emitted to soil, 17% emitted to air, and 1% emitted to freshwater) into the different environmental compartments (Balsari et al., 2007; Pergher et al., 1979; Gil et al., 2014) to quantify the emission inventory of pesticides in the different scenarios.

Characterization factors

Characterization factors (CF) translate the elementary flow into its impact on the chosen indicator (water or soil) for the ecotoxicity impact category (Hauschild and Huijbregts 2015). CF's according to Udo de Haes et al. 2002 (eq 1) are expressed as the product of a fate factor (FF), an exposure factor (XF) for the exposure of sensitive targets in the receiving environment and an effect factor (EF) expressing the effects of the exposure on the targets for the impact category.

$$CF = FF \times XF \times EF \quad (1)$$

USEtox 2.0 was used as characterization model for PFEIs of the five selected a.i (Table 2). On the other hand, the European Commission (2009) approved the use of five different a.i of copper (copper (I) oxide, copper (II) hydroxide, Bordeaux mixture, copper oxychloride, and tribasic copper sulphate). For the analysis, all copper a.i are going to be represented by the CFs for Cu (II) cation proposed by Dong et al. 2014 for freshwater impacts PFEIs and from Owsianiak et al. 2013 for the terrestrial ecotoxicity PTEIs.

Table 2: Active ingredients, recommended dose (Kg/ha) and characterization factors for PFEIs.

Active Ingredients	Recommended dose (Kg/ha)	Characterization Factors (PAF. day. m ³ /kg)		
		Em. air	Em. water	Em. soil
Azoxystrobin	0,25	9,78E+03	1,13E+05	1,42E+04
Cymoxanil	0,121	1,93E+02	1,42E+04	5,47E+01
Mancozeb	1,6	1,83E+02	6,84E+04	1,42E+03

Folpet	1,5	1,24E+04	1,65E+06	2,71E+03
Maneb	1,86	1,73E+02	8,96E+04	7,48E+02
Cu (II)**	1,57*	7,64E+06	7,50E+05	1,40E+07

*Average of the recommended dose for the five copper-base a.i

**CF Recommended from Dong et al. 2014

Freshwater ecotoxicity impacts

Potential freshwater ecotoxicity impacts (Hauschild, M., 2005; Rosenbaum et al., 2015) were quantified using USEtox 2.0 (Rosenbaum et al., 2008; Fantke et al., 2015) as characterization model. In the case of copper the CF's are expressed as the product of a fate factor (FF; day) that represents the residence time of total metal in freshwater environment, and a Bioavailability Factor (BF; dimensionless) which is the fraction of truly dissolved metal within total metal and Effect Factor (EF; PAF. m³/kg) represents the ecotoxicity of truly dissolved metal, expressed as potentially affected fraction (PAF) of freshwater species (Gandhi et al., 2010; Dong et al 2014).

The PFEIs score is calculated based on the inventory estimates described above and on the method as suggested in Payet et al., (2014) and Rosenbaum et al., (2008) as follows (eq 2):

$$IS_{x,i} = M_{x,i} * CF_{x,i} \quad (2)$$

Where $IS_{x,i}$ is the impact score of the active ingredient x in the compartment i. $M_{x,i}$ is the mass of the active ingredient x emitted in the compartment i (Kg/ha), $CF_{x,i}$ is the characterization factor of the active ingredient x emitted in the compartment i (CTUe/kg_{em}). The total impact score was calculated using the additive approach of the impacts $IS = \sum IS_{x,i}$ (Payet et al., 2014).

Terrestrial ecotoxicity impacts

The method of Owsianiak et al. 2013 introduces the accessibility factor (ACF) into the definition of CF and modifying the definition of the bioavailability factor (BF) (eq 3):

$$CF = FF \times ACF \times BF \times EF \quad (3)$$

Where CF (m³/kg_{total emitted}·day) is the toxicity potential of total metal emitted; FF (day) is the fate factor calculated for total metal in the soil; ACF (kg_{reactive}/kg_{total}) is the accessibility factor defined as the reactive fraction of total metal in soil; BF (kg_{free}/kg_{reactive}) is the bioavailability factor defined as the free ion fraction of the reactive metal in soil; and EFs (m³/kg_{free}) is the terrestrial effect. The PTEIs score is calculated as above mentioned.

3. Results

The resulting PFEIs for the six different fungicides showed up to 4 orders of magnitude of variation and ranking copper base fungicides as the higher potential ecotoxicity (figure 1). Figure 1 also shows impacts produced in the different environmental compartment soil, water and air, being soil load impacts the larger ones due to the highest factor emission accounted for soil, but it also important to highlight the importance acquired by emissions produced in water compartment due to highest CF.

Focusing on PFEIs for copper fungicides results show that water ecosystems with low hardness and DOC, and medium pH (EU6 water type) have higher toxicity potential for copper fungicides than those with high pH and hardness (EU1 water type); this differences in water chemistry not only influence changes in the PFEIs but also may lead to ranking changes when comparing with other a.i., illustrating the relevance of spatial differentiation. This indicates that copper toxicity potential depends in a large way of the water chemistry (Table 3).

In the case of terrestrial ecotoxicity the PTEIs for copper fungicides show up to 3 orders of magnitude of difference (1,15x10² to 2,70x10⁴), this is associated mainly with the variability in the

soil organic carbon, and the effect of the soil pH in bioavailability of the copper (Table 4). In fact, variability regarding characteristics of the environmental compartment has shown to be more important than the doses applied of the a.i. Regarding the scenarios used, they are extreme ones and their CF's correspond to the information available at the moment, in the case of the most common values of pH, soil organic carbon and clay content in the agricultural soils for wine production they will be represented by the soil scenario 2.

Figure 1: Total freshwater ecotoxicity impact scores and emission fractions split by environmental compartment for the six fungicides

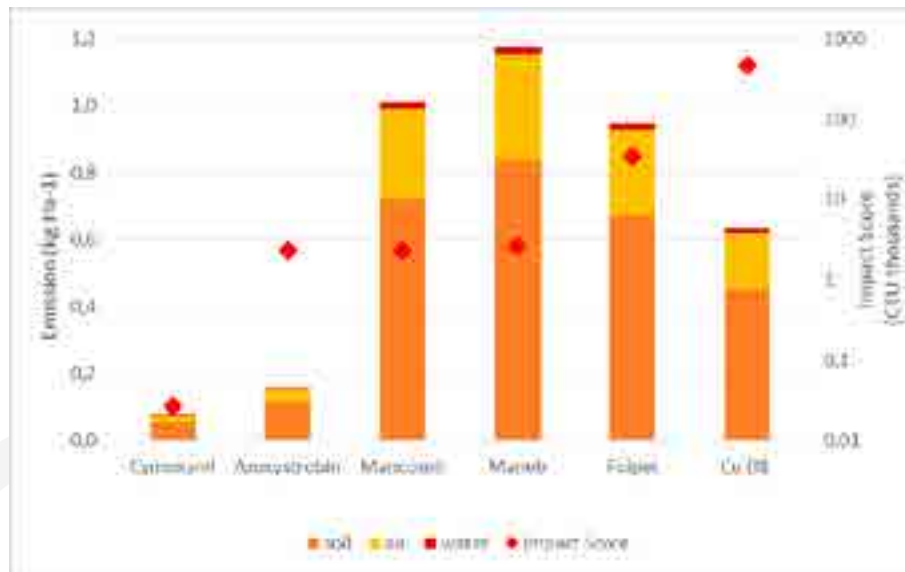


Table 3: Freshwater impact score results for copper fungicides by combined different water type scenarios

Water type	Impact Score		
	Min (1,5 kg/ha)	Med (2 Kg/ha)	Max (3kg/ha)
EU1	4,59E+02	6,12E+02	9,18E+02
EU2	1,91E+03	2,55E+03	3,83E+03
EU3	4,59E+03	6,12E+03	9,18E+03
EU4	1,10E+03	1,46E+03	2,19E+03
EU5	5,10E+04	6,80E+04	1,02E+05
EU6	1,22E+05	1,63E+05	2,45E+05
EU7	4,08E+04	5,44E+04	8,16E+04

Table 4: Terrestrial ecotoxicity impact score results for different application doses and different soil scenarios

Doses applied	Soil Impact Score		
	Soil scenario 1	Soil scenario 2	Soil scenario 3
1,5 kg/ha	1,15E+02	9,45E+02	1,35E+04
2 kg/ha	1,53E+02	1,26E+03	1,80E+04
3 kg/ha	2,30E+02	1,89E+03	2,70E+04

4. Discussion

Results have shown the significance of taking into account site-dependent and special differentiation conditions where emissions are produced, or pesticide applied.

Although copper fungicide has proved to present higher impact score than other organic fungicides, variability due to surrounding conditions made this impact also high variable.

On one hand, it has been seen that different approaches could provide quite different values, and on the other hand the importance of the geographic location of the activity not only because of the chemical-physical properties of the surrounding environment but also the distance to environmental targets (i.e. water bodies).

For the inventory we have used fixed values of emissions for the different environmental compartments, if we compare results with those presented in the work of Renaud-Gentié et al. (2014), our values result higher. Those authors adapted PESTLCI to be applied in the vineyard, this tool assumes part of the fate occurs in the inventory, on one hand, that also means the importance to agree on methods (Rosenbaum et al. 2015) but also shows the importance of focus on the development of characterization factors, which show higher variability.

The use of LCA tools to compare or advice for better techniques, in this case, use and substitution of pesticides, could be hampered if specific conditions of application and chemical/physical properties of pollutants and environment are not considered.

Special importance needs to be done to this issue for better addressing the impacts of inorganic fungicides to be able to compare different farming systems, especially in organic agriculture.

5. Conclusions

The present study has shown the importance of including spatial differentiation in the toxicity assessment. Accounting and comparing for pesticides substitution must be done about not only their intrinsic toxicity but also the surrounding environment where emissions are produced.

Methods to account for inorganics are not mature enough to be extensively applied, and more research is needed to capture potential ecotoxicity impacts better.

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98. Assessing potential pesticide-related ecotoxicity impacts of food products across different functional units

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ABSTRACT

The study aims to 1) demonstrate and apply a method for assessing the potential freshwater ecotoxicity impacts due to pesticide use in the primary production associated with six food products (chicken fillet, minced pork, minced beef, drinking milk, pea soup and wheat bread), and 2) evaluate how five different functional units (FUs) influence the results. Pesticide emissions were inventoried using an extended, updated and site-specific version of the PestLCI v. 2.0.5 model. In the impact assessment, USEtox v. 2.01 was used. The results show that the choice of FU has little influence on the outcome: four out of five FUs yield the same ranking of the animal-based food products: impact potentials decrease in the order minced pork > chicken fillet > minced beef > milk. The plant-based food products score considerably lower than the animal-based food products, regardless of FU. Notably, impact potentials of beef are lower than of chicken and pork, regardless of FU, contrary to typical carbon footprint and land use results for meat products. We conclude that the choice of FU did not influence the ranking of animal vs. plant-based food products. Also, we conclude that carbon footprints are inadequate proxy indicators of ecotoxicity impacts of food products and that ecotoxicity impacts need to be considered specifically, alongside other important impact categories.

Keywords: functional units, freshwater ecotoxicity, pesticides, USEtox, PestLCI.

Introduction

Estimates suggest that the planetary boundaries proposed to define the safe operating space for humanity have been transgressed for chemical pollution (Diamond et al., 2015), as well as for biodiversity loss (Rockström et al., 2009). Agricultural chemicals, such as pesticides, provide many benefits, but also contribute to chemical pollution, in e.g., surface waters (Stehle and Schulz, 2015), and to loss of biodiversity (Hallmann et al., 2014, Beketov et al., 2013, Whitehorn et al., 2012, Henry et al., 2012, Geiger et al., 2011). Despite being a highly relevant impact category in environmental assessments of food products, the ecotoxicity impacts from pesticide use are often excluded (Henriksson et al., 2012, de Vries and de Boer, 2010, Nemecek et al., 2016).

The choice of functional unit (FU) can have a large influence on results and conclusions in life cycle assessment (LCA) studies. The FUs should capture the primary function of the assessed product – such as nutrition in the case of food – but food LCAs usually only assess impacts in relation to kg food (Roy et al., 2009, de Vries and de Boer, 2010, Nijdam et al., 2012). Sonesson et al. (2016) developed new FUs based on the quality and/or quantity of protein, as well as the dietary context, with the intention to contribute to more relevant and useful information about the environmental impacts of food products. FUs based on protein quantity and/or quality are relevant since proteins are essential nutrients and associated with widely different environmental impacts depending on origin, and production methods.

The aim of this study is to 1) demonstrate and apply a method for assessing the potential freshwater ecotoxicity impacts due to pesticide use in the primary production associated with six food products and 2) evaluate the influence of different FUs on the results.

Methods

Six food products are considered: chicken fillet, minced pork, minced beef, drinking milk, pea soup and wheat bread. These food products are based on eight crops: rapeseed, feed wheat, bread wheat, barley, oats, grass/clover, peas and soybean. Food products are produced in the county of Västra Götaland, South West of Sweden. Seven of the crops are locally produced, and one (soybean) is produced in Mato Grosso, Brazil.

The pesticide application data represent current, typical and realistic use of pesticides in the studied crops and region, and were primarily obtained from Sonesson et al. (2014) which compiled information about current agronomic practices in the studied crops and regions (SLU, 2015). Pesticide application data for soybean were obtained from Nordborg et al. (2014) and represent cultivation of conventional soybean (not genetically engineered).

Pesticide emissions were calculated using an extended, updated and site-specific version of the pesticide emission model PestLCI v. 2.0.5 (Dijkman et al., 2012). This model has been described as the most advanced pesticide emission inventory model currently available for use in agricultural LCAs (van Zelm et al., 2014). PestLCI takes into account the physico-chemical properties of pesticides (e.g., degradation rates), local field conditions (e.g., slope), pedoclimatic conditions at the time and place of application (e.g., air temperature and soil clay content), and agronomic practices (e.g., tillage type). These parameters were adjusted to local conditions for the assessed crops and regions.

In the impact assessment, USEtox version 2.01 (www.usetox.org, Fantke et al., 2015a, Rosenbaum et al., 2008), released in February 2016, was used. USEtox is an emission route-specific impact assessment model developed in a “scientific consensus” process that “merged” several toxicity impact assessment models (Hauschild et al., 2008). It is generally recognized as the most advanced model currently available for comparative assessment of chemicals and their toxic effects on humans and freshwater ecosystems (see e.g., Hauschild et al. 2013) and recommended by several influential organizations and authorities (Fantke et al., 2015a). We used site-generic characterization factors at midpoint level. Characterization factors represent an estimate of the Potentially Affected Fraction (PAF) of species in (freshwater) space and time per unit emission, measured in the unit Comparative Toxic Unit ecotoxicity (CTUe) per kg emitted substance, where $1 \text{ CTUe} = \text{PAF} \cdot \text{m}^3 \cdot \text{day}$. New characterization factors were calculated for nine pesticide active substances, which were not available in the USEtox 2.01 database. In total, 26 pesticide active substances are included in the study.

Potential freshwater ecotoxicity impacts per kg harvested crop were first calculated as described above. Impact scores per kg food product consumed in the household were then derived by calculating the amount of crop(s) needed to produce each food product, taking into account representative conversion efficiencies in the assessed production systems, and downstream food processing (milling, slaughter, and cooking in the household). The production chains are described in Sonesson et al. (2014).

Impacts were assessed in relation to five FUs: food mass (kg), food energy content (Mcal), and three FUs that take protein quantity and/or quality into account: “kg protein”, and the newly developed FUs “kg digestible protein” and “kg PQI-adjusted food (AD)” where PQI stands for protein quality index and AD stands for average Swedish diet (Sonesson et al., 2016). The PQIs are dimensionless coefficients based on the composition of nine essential amino acids in the food product, the true ileal digestibility of each amino acid, the composition of the amino acids in the total dietary intake, and the nutritional requirements for the amino acids. The PQIs are thus dependent on the dietary context: the higher the PQI, the more valuable the product in a given diet. The idea is that products with a higher nutritional value (in relation to the dietary supply) will get more favorable LCA results (i.e. lower environmental impacts), and vice versa. Sonesson et al. (2016) developed PQIs for three Swedish diets with different supply of protein, but found that the dietary context was of little importance when ranking food products with regard to environmental performance. Therefore, only PQIs for one of the diets, AD, was included here.

Results

The potential freshwater ecotoxicity impacts for the six food products are presented in Figure 1. Beef scores lower than chicken and pork; pea soup and wheat bread have much lower impact potentials than the animal-based food products, and milk scores in-between the meat products and the plant-based food products. These results are stable across all five FUs.

The four mass-based FUs yield the same ranking of the animal-based food products: impact potentials decrease in the order minced pork > chicken fillet > minced beef > milk (Figure 1). In

relation to food energy content, chicken fillet scores higher than pork, since the energy density (Mcal kg⁻¹) of chicken fillet is 25% lower than of minced pork, hence less valuable from an energy perspective.

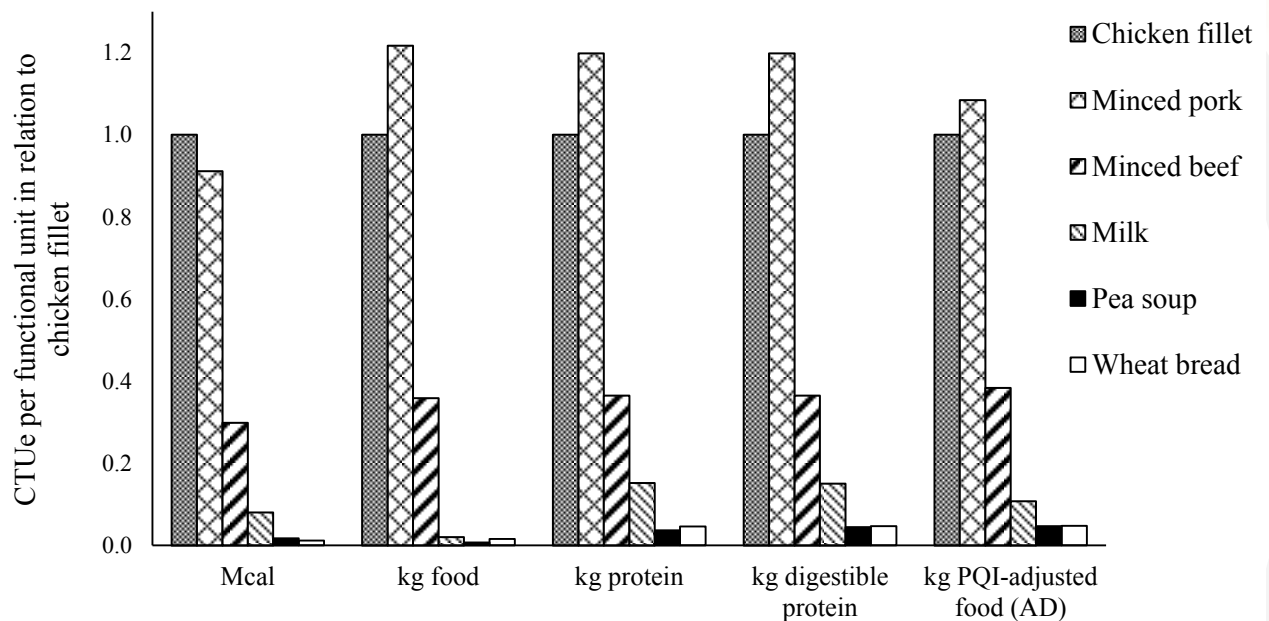


Figure 1: Potential freshwater ecotoxicity impacts of food products in CTUe (Comparative Toxic Units ecotoxicity) per functional unit (FU), in relation to chicken fillet. PQI = protein quality index, AD = average Swedish diet. Results are presented in relation to chicken fillet since we are primarily interested in how the different FUs rank the food products, and since the FU “kg PQI-adjusted food (AD)” represents a fictitious mass flow, rendering the absolute values difficult to interpret in terms of actual impacts and non-comparable to impact potentials expressed in relation to FUs that represent physical mass flows.

Discussion

The plant-based food products have considerably lower potential freshwater ecotoxicity impacts than the animal-based food products. This is primarily due to animal-based food production systems being less efficient at converting inputs (feed crops) to outputs (meat, milk or eggs), than plant-based food production systems, due to losses of energy and nutrients associated with an additional trophic level in the food chain. Therefore, the total use of pesticides per unit product becomes higher for animal-based food products, compared to plant-based food products, unless animal-based food production systems rely on grazing with very little feed crop supplement.

The four mass-based FUs yield the same ranking of the animal-based food products: impact potentials decrease in the order minced pork > chicken fillet > minced beef > milk. In particular, chicken and pork score higher than beef, for all five FU, despite poultry and pigs having higher feed conversion ratios and shorter cycle lengths than cattle. These results are primarily explained by the food products being based on different crops (Table 1) that are subject to different pesticides in the primary production, and consequently (widely) different potential freshwater ecotoxicity impacts. In relation to kg harvested crop, the impact potential of grass/clover is $1.7 \cdot 10^{-5}$ CTUe, while the impact potentials of feed wheat, bread wheat, peas, rapeseed, oats, barley and soybean are 10, 11, 14, 24, 42, 51 and 642 times greater, respectively. In Västra Götaland, 36% of the beef comes from specialized beef cattle and 64% comes from the dairy production system. Grass/clover is an important feed in both production systems, but contribute only 4% to the impact potentials of beef and milk (Table 1), due to very low pesticide use in cultivation. In contrast, the high impact potentials of chicken and pork are explained by the feed rations of poultry and pigs containing large amounts of soymeal produced from soybeans, with much higher pesticide inputs in cultivation.

Table 1: The contribution from crops to the potential freshwater ecotoxicity impacts in CTUe (Comparative Toxic Units ecotoxicity) per kg food product. The “-“ indicate that the crop is not used in the production of the food product. The percentages all sum up to 100%.

	Bread	Chicken fillet	Minced pork	Minced beef	Milk	Pea soup
Wheat	100%	4%	4%	-	-	-
Rapeseed	-	1%	2%	-	-	-
Soybean	-	95%	66%	23%	26%	-
Barley	-	-	21%	40%	38%	-
Oats	-	-	7%	32%	31%	-
Peas	-	-	-	-	-	100%
Grass/clover	-	-	-	4%	4%	-

The finding that beef scores better than chicken and pork, for all five FUs, is noteworthy since it contradicts findings in studies quantifying carbon footprints and land use of meat products. Such studies typically attribute larger impacts to beef, than to chicken and pork, due to lower feed conversion ratios and reproduction rates in beef production system, as well as methane emissions from enteric fermentation contributing to climate impacts (Nijdam et al., 2012, Westhoek et al., 2011).

Despite a detailed and site-specific inventory of pesticide usage and emissions in the studied crops and regions, the results are subject to uncertainties and limitations. Some data display large spatial and/or temporal variability, such as type and amount of pesticides applied and soil and climate conditions (influencing emissions). In addition, feed rations vary within production systems, in particular in beef production systems (Westhoek et al., 2011). Intensive feedlot systems with little or no grass and more soybeans or other protein-rich feed crops would likely score higher. More research is needed to assess how different beef production systems perform. Less variation can be expected for chicken and pork production in the industrialized world, since these production systems are more standardized.

More comprehensive assessments are needed where ecotoxicity impacts are considered specifically, alongside other important impact categories in environmental assessments of food products. For example, besides the amount of land needed to support production, the quality of land is also an important factor to account for. van Zanten et al. (2016) showed that from a land use efficiency perspective, some ruminant production systems outperform both monogastric and food crop production systems.

While plant-based food products have lower impact potentials than the animal-based food products, caution should be applied before generalizing this finding, since only six food products were assessed. Some fruits and vegetables may score higher than meat products due to the use of high-toxicity pesticides in the cultivation.

Conclusions

Despite being a highly relevant impact category, ecotoxicity impacts from pesticide use are often excluded in environmental assessments of food products. Here, we assessed the potential freshwater ecotoxicity impacts due to pesticide use in the primary production of six food products produced in Sweden. We also assessed how the results vary across five different FUs.

The plant-based food products have much lower impact potentials than the animal-based food products, for all five FUs. The choice of FU was thus not critical to the degree that it influenced the ranking of animal vs. plant-based food products (but it partly influenced ranking within these categories). However, only six food products were assessed here. Ecotoxicity impacts of a wider range of food products, e.g., tropical fruits, need to be assessed in order to establish whether plant-based food products always score better than animal-based food products.

We also found that beef has lower freshwater ecotoxicity impacts than chicken and pork. This result, which is stable across the FUs, stands in sharp contrast to typical carbon footprint and land use results. Carbon footprints are sometimes used as proxy indicators of environmental impacts. We conclude that carbon footprints are inadequate proxies of the ecotoxicity impacts of food products.

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132. Estimating pesticide emission fractions for use in LCA: a global consensus-building effort

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ABSTRACT

A practical challenge in LCA for comparing pesticide application in different agricultural practices is the agreement on how to quantify the amount emitted, while only the amount applied to the field is known. Main goal of this paper is to present an international effort carried out to reach agreement on recommended default agricultural pesticide emission fractions to environmental media. Consensual decisions on the assessment framework are (a) primary distributions are used as inputs for LCIA, while further investigating how to assess secondary emissions, (b) framework and LCA application guidelines and documentation will be compiled, (c) the emission framework will be based on modifying PestLCI 2.0, (d) drift values will be provided by German, Dutch and other drift modelers, (e) pesticide application methods will be complemented to develop scenarios for tropical regions, (f) climate, soil and application method scenarios will be based on sensitivity analysis, (g) default emission estimates for LCA will be derived from production-weighted averages, and (h) emission fractions will be reported spatially disaggregated. Recommendations for LCA practitioners and database developers are (a) LCA studies should state whether the agricultural field belongs to technosphere or ecosphere, (b) additional information needs to be reported in LCI (e.g. pesticide mass applied), (c) emissions after primary distribution and secondary fate processes should be reported, (d) LCIA methods should allow for treating the field as part of technosphere and ecosphere, (e) fate and exposure processes should be included in LCIA (e.g. crop uptake), (f) default emission estimates should be used in absence of detailed scenario data, (g) and all assumptions should be reported. The recommended pesticide emission fractions results and recommendations are presented and disseminated to strive for broad acceptance at a dedicated stakeholder workshop back-to-back with the current LCA Food 2016 conference in Dublin.

Keywords: life cycle assessment, emission quantification, agricultural pesticides

1. Introduction

Life Cycle Assessment (LCA) helps establishing and comparing environmental performance profiles of products or services. For agricultural LCA in particular, one of the major challenges is the comparison between different farming practices, comparing for example various pesticides applied in conventional agriculture to alternative (functionally equivalent) solutions, such as organic or integrated farming practices. LCA thereby aims at identifying the “best-in-class” solution(s) among all considered practices.

What is required to quantify impacts on humans and ecosystems related to the pesticides used is the amount of pesticides emitted to the different environmental media (air, water, soil, crop residues) under different practices (Dijkman et al., 2012). However, this information is mostly not available to LCA practitioners, while typically the pesticide amount applied to the agricultural fields is known (Fantke et al., 2012). This constitutes a practical challenge in LCA for comparing the application of pesticides in different agricultural contexts and practices. Different tools and approaches exist to address the quantification of pesticides emissions in LCA, yielding inconsistent outcome, thereby hampering to consistently account for impacts related to the use of pesticides in agricultural LCA.

In response to this challenge, a global effort was initiated¹ with the objective to estimate and agree on recommended default agricultural pesticide emission fractions to environmental media. It is the

¹ <http://www.qsa.man.dtu.dk/Dissemination/Pesticide-consensus>

aim of the present paper to summarize the findings of this effort, to outline the followed approach, the recommendations that have been agreed upon until now, and to provide an outlook on how to set the achieved results and recommendations into practice.

2. Methods

Three expert workshops have been organized involving more than 70 specialists representing industry, government, and academia from 24 countries and 5 continents. Main objectives of the workshops were to streamline and coordinate the global effort on quantifying emission fractions for use in LCA.

The first workshop (scoping workshop) was held in 2013 in Glasgow (UK) with focus on providing guidance on the delimitation between life cycle inventory and impact assessment in LCA with respect to pesticide use in agricultural practices. Consensual recommendations were reached as result of this workshop on a consistent accounting of emissions and impact assessment. These recommendations have been fully peer-reviewed and published in Rosenbaum et al. (2015).

The second workshop (framework workshop) was held in 2014 in Basel (Switzerland) with focus on agreeing on how to consistently model the fractions of applied pesticides that enter air, water, soil, and agricultural crops as emissions under different agricultural pesticide application practices. Outcome of this workshop was a defined set of data and models that can be used and that can be consistently combined to arrive at an overall emission quantification framework that can be ultimately operationalized to quantify pesticide emission fractions to the environment for a defined set of scenarios applicable for LCA.

The third workshop (consensus workshop) was held in 2015 in Bordeaux (France) with focus on presenting and discussing intermediate results of the follow-up work after the first two workshops. In the consensus workshop, agreement was reached on (a) the modeling framework, (b) the set of default scenarios to be recommended for LCA, and (c) the format of the emission results along with associated data requirements for implementation into current LCA software.

The consensus building process until the workshop in Bordeaux in 2015 is illustrated in Figure 1.

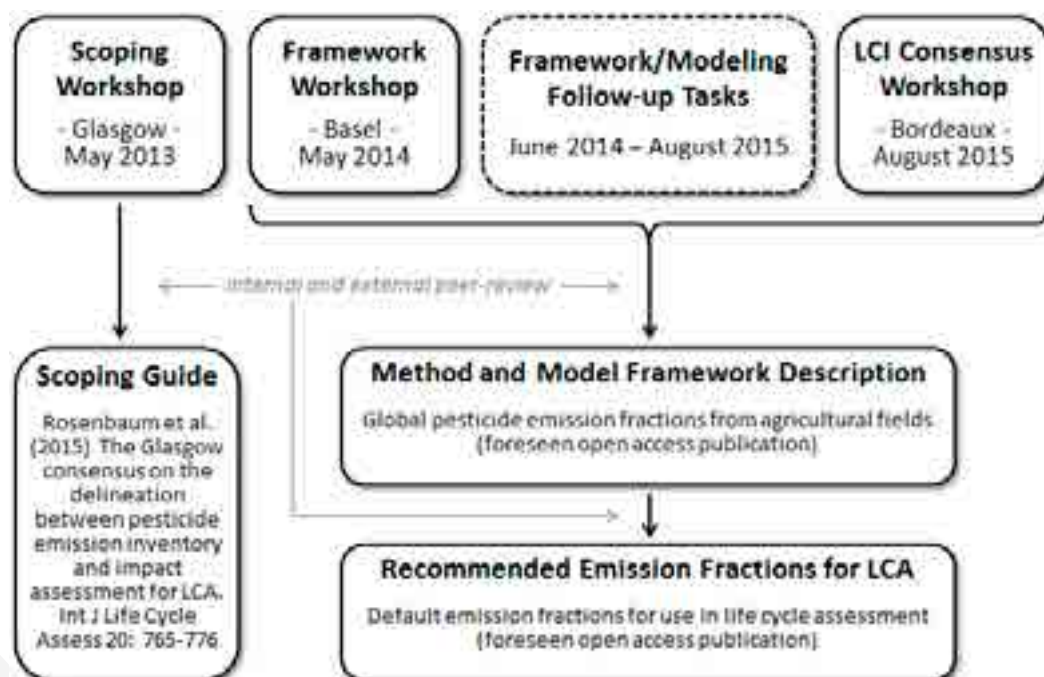


Figure 1: Overview of the global consensus building process to arrive at recommended default pesticide emission fractions readily usable in life cycle assessment.

3. Results

A set of consensual decisions of how to consistently quantify pesticide emission fractions to the environment is summarized in the following as outcome of the three workshops and follow-up work by several international research teams:

(a) At this stage, only primary pesticide distributions are used as direct inputs for life cycle impact assessment (LCIA) and further investigation is required about how to couple secondary emissions to LCIA models,

(b) Guidelines and a full documentation will be compiled regarding the model framework and how to combine LCI primary pesticide distribution and secondary emissions results with LCIA toxicity characterization results,

(c) The emission quantification framework will be based on a model that builds on a completely reworked and extended PestLCI 2.0 including adjustments for the implemented drift functions,

(d) Average drift values are needed and will be provided by German (Rautmann et al., 2001) and Dutch (Holterman and van de Zande, 2003) drift modelers as well as additional functions for scenarios not covered by the available drift functions,

(e) To ensure a broad coverage of assessing agricultural practices, pesticide application methods will be complemented as input to build emission scenarios for tropical regions,

(f) Climate, soil and pesticide application method scenarios will be selected based on sensitivity analysis,

(g) Default emission estimates for LCA will be derived as pesticide production (mass)-weighted averages across detailed emission results,

(h) Emission fractions will be reported spatially (geographically and politically) disaggregated as function of climate zones, crop production, and administrative regions (e.g. countries),

The conceptual framework of how to model pesticides from application to agricultural crops to emission fractions reaching different environmental media and receptors as developed and agreed in Basel in 2014 (Figure 2) was used as starting point for improving the PestLCI 2.0 model towards the agreed points.

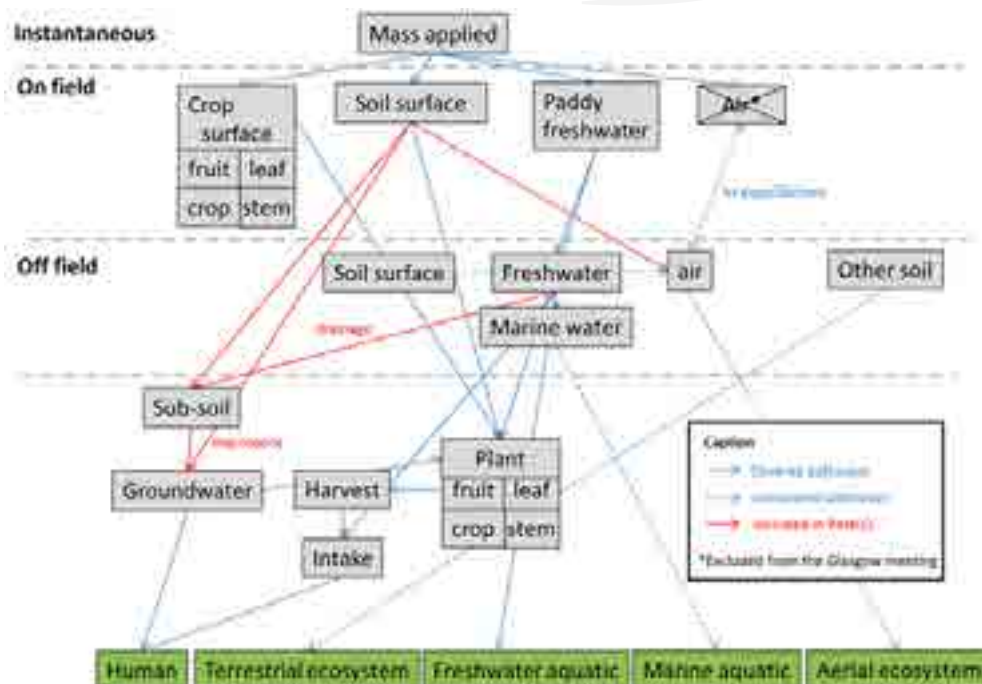


Figure 2: Conceptual framework of how to model pesticides from application to agricultural crops to emission fractions reaching different environmental media and receptors. Emissions or residues will be reported in LCA databases for the following compartments “Soil surface”, “Freshwater”, “Marine water”; “Air”, “Groundwater”, and “Plant”.

The set of scenarios agreed to combine crop class, pesticide target class and application methods is shown in Figure 3 and was used to cover the main globally occurring combinations.

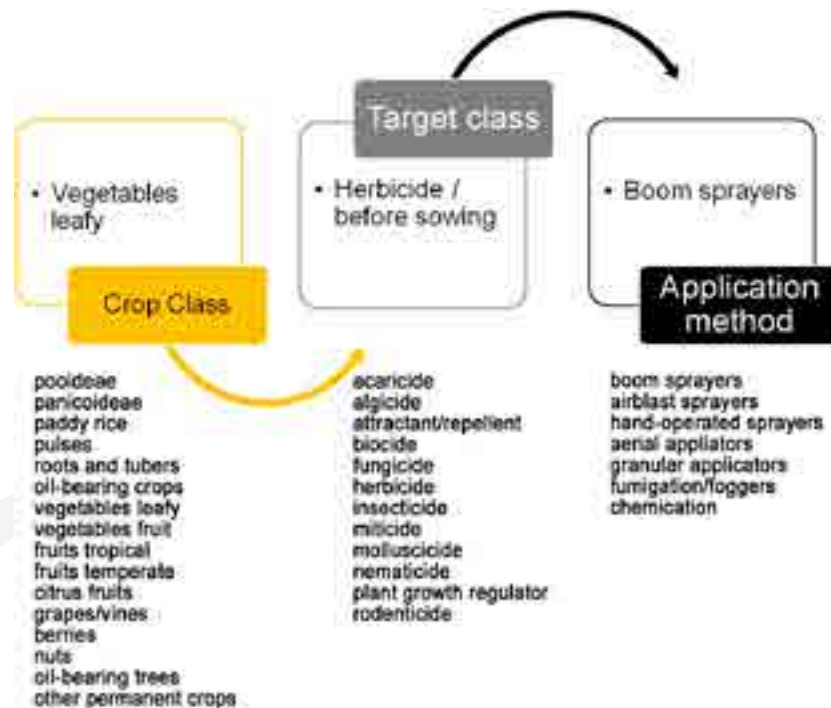


Figure 3: Archetypal classification of crop class-pesticide target class-pesticide application method scenarios used for estimating initial pesticide emission fractions for LCA.

Based on the set of agreed decisions about the assessment framework, a set of recommendations was established for LCA practitioners and software developers:

(a) In the goal and scope section of an LCA, it should be stated whether the agricultural field belongs to the technosphere or to the ecosphere,

(b) Pesticides, crops, pesticide mass applied to agricultural fields, application method, presence of buffer zones, application location and time, and adherence with good agricultural practice should be reported as part of the LCI to quantify pesticide emissions from agricultural fields,

(c) Both, emissions after primary distribution processes (i.e. immediate) and after secondary fate processes (i.e. with longer time horizon) should be reported during the LCI phase,

(d) LCIA methods should allow for treating the agricultural field as part of the technosphere and as part of the ecosphere,

(e) In LCIA, secondary drainage, runoff, degradation/dissipation, volatilization, leaching, crop uptake and related food residue exposure, and exposure of bystanders, applicators and field workers should be included,

(f) If detailed information on pesticide application location and time is not available, default emission estimates for LCA should be used as derived as pesticide production (mass)-weighted averages across detailed emission results,

(g) All assumptions about buffer zones, application scenario, impact assessment methods, considered fate and exposure processes and pathways, and spatiotemporal resolution should be reported by LCA practitioners.

4. Discussion

After implementing, testing and fully documenting all consensus-based agreements, results are presented and disseminated together with agreed recommendations for LCA practitioners and database and model developers to strive for broad acceptance at a dedicated Stakeholder Workshop back-to-back with the LCA Food 2016 conference in Dublin (Ireland) on October 18, 2016². Objective of this workshop is to seek broad stakeholder acceptance and agreement on recommended default agricultural pesticide emission fractions to environmental media in LCA. The target is the feasibility to implement these pesticide fractions along with associated data requirements into LCI databases to improve current LCA practice with respect to impacts from the use of agricultural pesticides.

5. Conclusions

The international consensus building effort emphasizes the importance of involving the broad range of stakeholders for which a consistent consideration of agricultural pesticides in LCA is relevant. A combination of state-of-the-art science-based methods and feasible scenarios and assumptions are required to arrive at agreement for advancing the assessment of the various agricultural practices in LCA. For practitioners, the results of the global effort constitute an improvement with respect of comparing agricultural practices, striving towards a consistent interface between LCI and LCIA, and a clear guidance of how to evaluate pesticides. Follow-up efforts are required to adapt LCIA models, especially with respect to groundwater emissions and on-field impacts, and to also address agricultural nutrient emissions in LCA.

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² <http://www.qsa.man.dtu.dk/Dissemination/Pesticide-consensus/Stakeholder-workshop-Dublin-2016>

244. Bioplastics: Does biodegradability have any ecological advantage per se?

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ABSTRACT

We analysed the environmental (dis)advantage of bioplastics with regard to their biodegradability in several studies. Data for composting and anaerobic digestion of bioplastics were derived from standardized composting and digestion studies. For the interpretation of the environmental impacts the ecological scarcity method was used for decision support and as a validation compared to the ILCD single score method. Biodegradability does not have an advantage per se from an environmental perspective. Composting of bioplastics leads to its disappearance in the best case. But there is no added value such as fertilisation or soil improvement (organic matter, humus) that normally occurs by composting organic waste. The same applies to digestate of bioplastics. Anaerobic digestion produces methane that can substitute natural gas. Whereas only about 60 % of the energy content can be “harvested” with anaerobic digestion, thermal exploitation in an incineration plant has higher yields. Biodegradability has no environmental advantage per se if compared to other end of life treatment options.

Keywords: bioplastics, composting, anaerobic digestion, thermal utilization, soil improvement

1. Introduction

The market of biodegradable bioplastics is growing each year. And the property biodegradable – among others such as renewable – is used for the promotion of their ecological benefits. Nowadays, many day-to-day consumer goods such as take-a-way packaging is made of biodegradable bioplastics. But how does the optimal end of life treatment look like – except for recycling which was not part of this study? Shall we compost biodegradable plastics, or put them to an anaerobic digestion plant, or is thermal utilization in an incineration plant also a advisable option? On behalf of the Amt für Umwelt und Energie, Basel (AUE) and the Amt für Abfall, Wasser, Energie und Luft, Zürich (AWEL) a study was done answering these questions.

2. Methods (or Goal and Scope)

Goal and Scope

The goal of this study was to analyse the end of life treatment options anaerobic digestion, thermal utilization or composting of biodegradable bioplastics by means of life cycle assessment. As functional unit 1 kg of biodegradable bioplastic was chosen. Only processes of the waste streams were included. The production and the use phase of the bioplastics were excluded because they are outside the system boundary and not necessary to answer the questions of the study. The different end of life treatments deliver different products: methane and digestate for anaerobic digestion, electricity and heat for thermal utilization and some sort of compost for composting. In order to make the end of life treatments comparable, credits were given for the different products assuming that they replace similar products on the market (e.g. electricity from a thermal utilization plant replaces otherwise produced electricity etc.).

Inventory data

Data for composting were derived from Pladerer et al. (2008). Data for anaerobic digestion of bioplastics are based on a preliminary study about the degradability of bioplastics in anaerobic digestion plants. In this preliminary study the actual degradability rates and methane yields were measured under standardized digestion settings (Baier, 2012).

Data for energy recovery in incineration plants were taken from the Ryttec report (2013) on energy efficiency of Swiss incinerations plants.

The data and methodology of Dinkel et al. (2011) described in Zschokke et al. (2012) was used to derive credits for organic matter.

Impact assessment methods

Different environmental impacts were analysed. But in order to guarantee sound and effective decision support aggregated single-score results were used (Kägi et al., 2016). Therefore, the ecological scarcity method (Frischknecht & Büsser Knöpfel, 2013) was used for the interpretation of the environmental impacts. This method was also used as a validation compared to the ILCD single score method (European Commission-Joint Research Centre, 2011), using the weighting scheme suggested by Huppel et al. (2011).

3. Results

Figure 1 shows the overall results of three end of life treatments of the three bioplastics polylactic acid (PLA), starch blend and cellulose acetate. It is obvious that incineration is always among the best end of life treatment options, whereas composting seems always to perform worst.

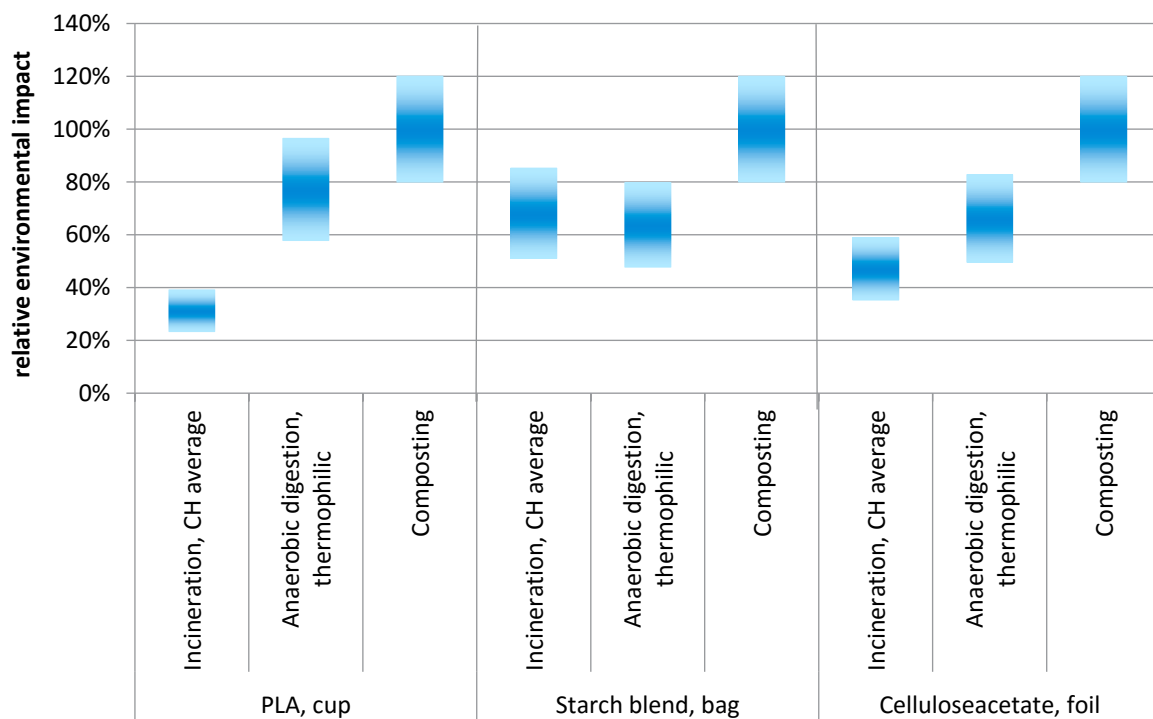


Figure 1: Relative environmental impact of different end of life treatments of bioplastics (PLA, starch blend, cellulose acetate) using the ecological scarcity method 2013 and the basket of benefits concept.

The reason can be better understood in analysing figure 2 which shows the environmental impacts and benefits of different end of life treatments of bioplastics such as cellulose acetate and biomass (example of palm leave; presented here to better understand credits for organic matter).

Inspecting the environmental impacts due to emissions only (figure 2, example of cellulose acetate), incineration shows the highest impact due to air emissions followed by anaerobic digestion and composting. For biomass (figure 2, palm leave) the results look differently: Anaerobic digestions and composting show higher impacts than incineration mainly due to the heavy metal emission to soil (digestate and compost). As there are no such heavy metals in bioplastics, the corresponding emissions do not exist at all.

Looking at credits only with the example of cellulose acetate, incineration shows the highest credits due to sold electricity and heat (replacing marginal electricity and heat), followed by anaerobic digestion with credits for sold biogas (replacing natural gas in a co-generation plant). The biogas credits are lower because – among other reasons – only a certain fraction of the embedded energy is transferred to methane (the remaining carbon is transferred to CO₂ or is not converted at all and remains in the organic residues). No credits are given for organic matter (humus) and for fertilisers.

This stays in contrast to biomass, for which quite high credits are given. This is due to the fact that the considered bioplastics do not contain any substantial nutrients such as nitrogen, phosphorus, or potassium. They are also lacking any structural molecules that could lead to humus build-up or complex top soil structures. Humus is only formed if some sort of lignin or complexing agents are included, which is the case for biomass but not for bioplastics (Dinkel & Kägi, 2013; Zschokke et al., 2012). This implies that the carbon in bioplastic digestate or compost is weakly bound and will therefore be metabolised to CO₂ sooner or later.

Over all, the credits are higher than the impacts, therefore leading to negative total results as only the end of life step was considered. In general, the anaerobic digestion and composting show equal or worse results than the incineration path for bioplastics.

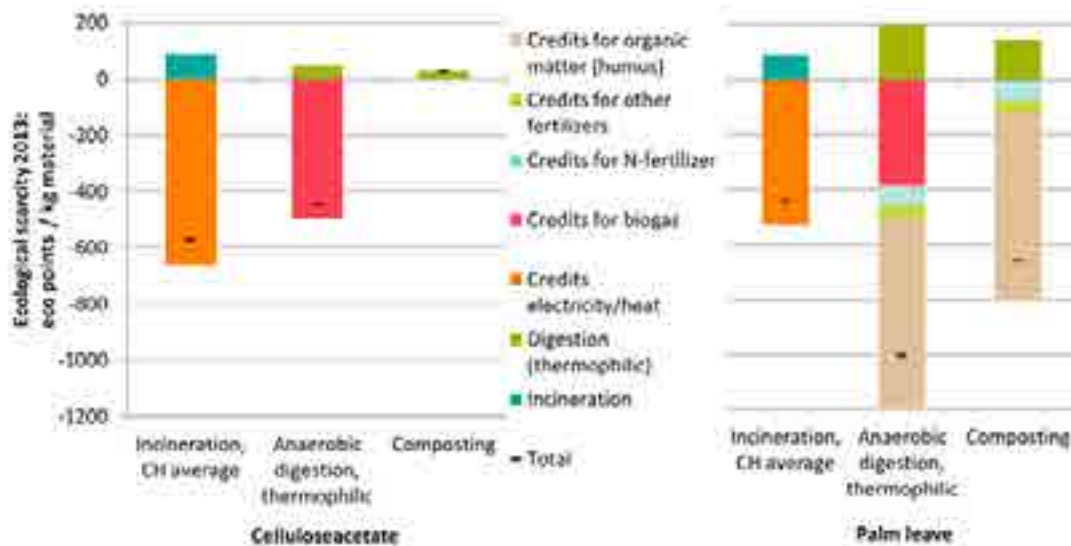


Figure 2: Environmental benefit of different end of life treatments of bioplastics (cellulose acetate) and biomass (palm leave) as an example for credits for organic matter using the ecological scarcity 2013 method and the avoided burden concept.

4. Discussion

Biodegradability does not have an advantage per se from an environmental perspective. Composting of bioplastics leads to its disappearance in the best case. But there is no added value such as fertilisation or soil improvement (organic matter, humus) which normally occurs by composting organic waste. The same is true for digestate of bioplastics. Anaerobic digestion leads to methane that can substitute natural gas. But whereas only about 60 % of the energy content can be “harvested” with anaerobic digestion, thermal exploitation in an incineration can mineralise almost all of the organic matter and shows always similar or even better results than anaerobic digestion.

5. Conclusions

Biodegradability has no environmental advantage per se if compared to other end of life treatment options. Biodegradability of bioplastics does not reduce the environmental footprint. On the contrary, the biodegradation of bioplastics often leads to higher environmental footprints compared to incinerating them. Our results are of course only valid for countries in which incineration is combined with energy recovery. In other countries where landfilling is normally employed and incineration plants are missing, the biodegradability of bioplastics may have advantages.

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ORAL PAPERS

MEAT PRODUCTION 1

6. Meat production (1)

6. Is Danish venison production environmentally sustainable?

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ABSTRACT

The objective of this study is to quantify the environmental impact of venison production from six wild life species in Denmark: Red deer, roe deer, fallow deer, wild boar, mallard and pheasant, and compare it with the environmental impact of commercially produced beef, pork and chicken in Denmark. The method for quantifying the impact of venison applied original LCI data obtained for the complete life cycle of Danish venison production of all six species, supplemented with data from Ecoinvent[®] and LCAFOOD on materials and processes involved in production of venison and industrial meat. Fodder, foraging on farmers' fields, infrastructure, hunter/hunting and abattoir processes were analyzed separately using Simapro software applying the Stepwise[®] method. The results indicate that Danish venison production ranges from being slightly less, over being equally, to most often being far more environmentally harmful than the production of comparable industrial meat types. The main environmental impact originated from feed and foraging on farmer's fields and mileage driven by the hunters was surprisingly high. Danish industrial meat from domestic animals is typically more environmentally friendly than Danish venison.

1. Introduction

It is a popular notion in Scandinavia that we should increase ingredients in our diets that are gathered, caught or hunted in nature rather than bred, grown and harvested on farmed fields, and in stables and waters. These new ingredients include commodities like seafood, seaweed, mushrooms, herbs and venison, i.e. meat from free-ranging wildlife. In the recommendations for the New Nordic Diet, Danish consumers are advised to consume 35 % less meat, with more than 4 % of the consumed meat being venison (Meyer et al. 2011). But the present study contradicts that the Danish venison production alone will be able to support that goal. A total of 2.6 million wild animals are reported killed by hunters in Denmark each year, and the possibility for increasing Danish venison production is limited, considering Denmark's sustainable conservation policy and the limited area not already occupied by buildings and roads or exploited by agriculture. Presently Danes consume only 0.8 % of their meat as venison, and the consumption is very unevenly distributed; most hunters keep the main portion for themselves. The "wild" ingredients in a modern diet are in general assumed to be both healthy and environmentally sustainable. But is that necessarily true? The present study seeks to answer the question if Danish venison production has less impact on the environment than Danish production of meat from domestic animals. Six types of venison were considered.

2. Methods and materials

The goal of this study is to provide information to hunters, nature managers, distributors, retailers, consumers – and to the project sponsors (15 Juni Fonden, Nordea Fonden) – on the sustainability of Danish venison production at *Klosterhedens Vildt*, one of the two major Danish abattoirs, specialising in venison production, and its contracted suppliers.

A 15-page questionnaire was forwarded to the abattoir, game keepers, huntsmen, traders and a wide range of businesses and experts at relevant private and public institutions ahead of a series of interviews to collect relevant LCI (Life Cycle Inventory) data on game management, infrastructure, hunting, feed and foraging, transport, processing, packaging, materials, energy, waste, distribution, and more. The extensive notes from these interviews were processed and returned for approval to those interviewed. After communication back and forth by email and in person, the results obtained formed the basis of a complete LCI of Danish venison production from six species – red deer, roe deer, fallow deer, wild boar, mallard and pheasant – processed at Klosterhedens Vildt abattoir in Jutland (Western Denmark). The study focused on the activities in 2010/11 considered to be representative of the venison production at Klosterhedens Vildt abattoir even today. The LCIs were

elaborated separately for each of the six investigated species, all including three main stages: (1) from animal birth to the abattoir, (2) slaughter, packaging and waste processes at the abattoir, and (3) from abattoir to retailers. The first step was divided into (a) infrastructure, (b) fodder and foraging on farmers' fields and (c) hunters and hunting.

Production data for industrially produced meat from domesticated animals were taken from the Ecoinvent[®] and LCAFOOD databases. The environmental impact of venison production was analysed by consequential life cycle assessment (cLCA) using Simapro[®] 8.04. The consequences considered were the substitution of industrial meat types with venison. The Stepwise[®] 1.05 method (Weidema 2009), analysing 16 environmental impact categories associated with all activities, energy- and resource consumption from soil-to-supermarket or restaurant, was chosen as the most appropriate LCA method with the option of monetizing. Monetizing summarises 15 impact categories in a common expression, thus revealing the socioeconomic cost associated with the environmental impact of Danish venison production within the scope of this study. Monetizing makes it possible to compare the environmental impact of widely different products and services. The monetised environmental impact of venison production from red deer, roe deer, and fallow deer was compared to the monetised environmental impact of beef production, wild boar production with pork production, and mallard and pheasant was compared with chicken production.

The functional unit (FU) was 1 kg of meat as there was not enough data on venison's nutritional content to select e.g. protein content as the FU. The geographical scope included venison from Jutland (the Western Danish 29,652 km² peninsula + the island of Samsø 114 km²), and only meat produced at Klosterhedens Vildt abattoir during the complete hunting season 2010/11. It is assumed that the studied venison production is a fair estimate of the overall Danish commercial venison production.

Red deer were supplied to Klosterhedens Vildt abattoir from three locations: Oksbøl and Ulfborg-Klosterhede plantations managed by the Danish Nature Agency in Western Jutland, and Aage V. Jensen Naturfond's nature conservation and wild life protection area Lille Vildmose in North Eastern Jutland. Each of these data suppliers provided distinctive LCI data. The three areas represent different types of nature management. In the first two, the red deer were free ranging. At Oksbøl they had access to a foraging field planted inside the forest to keep them from foraging neighbouring farmer's fields. At Ulfborg-Klosterhede the deer foraged to a certain extent on neighbouring farmer's fields, and in Lille Vildmose the red deer (and wild boar) were fenced inside a large area where they roam freely but have to be fed to maintain the stock.

LCI data on roe deer came from Brattingsborg estate on the Southern tip of the Island of Samsø where a 2,367 ha fenced area was used for combined agriculture, forestry, pork production and hunting. The roe deer browsed in the forest and fed on the crops. The hunted area was populated with roe deer, fallow deer and pheasants. In this study we only include data on the roe deer from Brattingsborg. 74 roe deer were shot during the hunting season of 2010/11, of which 37 were sold to Klosterhedens Vildt abattoir.

LCI data on fallow deer were obtained from a deer park run by Ørumgård estate near Vejle. This is the only supplier of fallow deer to Klosterhedens Vildt abattoir included in this study, although they only delivered 9 of the 140 fallow deer processed at Klosterhedens Vildt abattoir in 2010/11. The results are thus less representative than for red deer, and possibly representing high-end impact values for fallow deer since they were fenced in with free access to agricultural fields. However, many fallow deer were delivered from the Central Danish island of Funen and other long-distance locations to the abattoir where environmental impact of transport had higher impact.

LCI data on wild boar came from Lille Vildmose where an electric fence kept them at a safe feeding distance from tourists and thus away from the risk of swine fever. The wild boars were fed, and 165 sold to Klosterhedens Vildt abattoir.

Data on mallards were based on 4,000 mallards raised at Bakkegårdens Vildtopdræt and put out in constructed lakes at Frijsenborg who supplied them to the abattoir after hunting. Data on pheasants were based on 8,000 pheasants raised by Frijsenborg estate. Only 40 % of the pheasants at Frijsenborg estate were shot during the estate hunts during 2010/11, while 30 % were estimated taken by neighbours and predators, and 30 % survived until the following seasons and thus tended to migrate.

The LCI of deer foraging on farmers' fields was estimated from expert testimony regarding the species and amount of feed plants consumed and disturbed (trampling and eating reproductive organs) on relevant locations, stomach content of dead deer (Petersen, 1998), informal statements by local

farmers and Kanstrup et al. (2014). The LCI of supplied fodder was based on the types and amount of fodder with known compositions targeted at the animals to be fed at each location. The LCI of infrastructure included fences (the share related to wildlife), electricity to supply e.g. cold rooms and electric fences, wood and metal for fences and shooting towers, transport for construction, inspection and replacement of fences and shooting towers, local transport of hunters, and transport of dead animals to the abattoir. The LCI of the hunters and hunting was based on the mileage driven by hunters to the hunting and training sites, and consumed equipment (e.g. bullets and cartridges). The LCI of the abattoir material consumption and processes was allocated to the six species according to weight. It included energy consumption (30,110 KWh), water, people and product transport, cardboard boxes, plastic bags, transport and processing of waste, including gain from incineration at a local plant, and transport of the end products to the retailers. There were no data on enteric emissions and emissions from manure from venison to be included in this study. Saxe (2015) gives a detailed and complete description of all calculations for all six species.

Regarding allocation of the environmental impact of venison production there are several options – choices to me made. In this study it was decided that ‘the joy of hunting’ (1 million hunting licences in Denmark) was counterbalanced by ‘the disturbance by hunters’ (4.7 million Danes enjoy nature without a hunting licence). Some postulate that hunting equals ‘nature management’ so that without it, the Danish deer populations would ‘run out of control’, increasing crop loss from farmed land and ultimately leading to deteriorating health and collapse of starving deer populations. Since population control could be carried out more efficiently than hunting, and may not even be needed, this aspect was not included. Of all mammals and bird species that breed and thrive in Denmark, the 90 % not-hunted species manage perfectly well to stay within sustainable population sizes without requiring hunting for regulation. For all six species it was decided to allocate all of the impact to the meat value in this study, and none to the ‘joy of hunting’, ‘disturbance’, or ‘nature management’, since the latter impacts are subjective (‘immaterial/intangible goods’) compared with the physical and accountable impacts of hunting. The immaterial aspects of Danish nature were discussed by Jacobsen et al. (2014).

There were several sources of uncertainty in the calculations of environmental impact of venison production; the major source being the estimation of foraging, i.e. the feed that deer take from farmers’ fields and the impact on plant growth and development by trampling. This is at the same time the largest single source of environmental impact of free ranging deer species.

3. Results

The total meat production at Klosterheden Vildt abattoir in 2010/11 is given in Table 1.

Table 1. Venison from six species processed at *Klosterhedens Vildt* abattoir.

Species	Number of animals	Tot kg meat weight	Ave kg per animal	Net kg produced
Red deer	774	36,378	47.0	25,465
Fallow deer	140	4,340	31.0	3,038
Roe deer	252	2,873	11.4	2,011
Wild boar	174	4,727	27.2	3,545
Mallard	3,082	2,620	0.75	1,965
Pheasant	12,721	8,269	0.85	1,819
Total	-	-	-	37,843

3.1 LCI

The hotspots for environmental impact of venison include feed/fodder, infrastructure, and the hunter/hunting. The latter include an unexpected high mileage travelled by the hunters in private cars. The LCI flow chart (Figure 1) for commercial production of red deer venison at Klosterhedens Vildt abattoir is an example of the flow diagrams used as the basis for cLCAs for all six species.

3.2 Global Warming Potential (GWP)

The impact on the GWP of producing 1 kg deer venison (Table 2) was found to be within or below the impact range of producing 1 kg of beef (Cederberg et al. 2011, Parsson et al. 2015). Dairy cattle emit more methane than deer, both in total and per kg (Swainson et al. 2008). The GWP associated

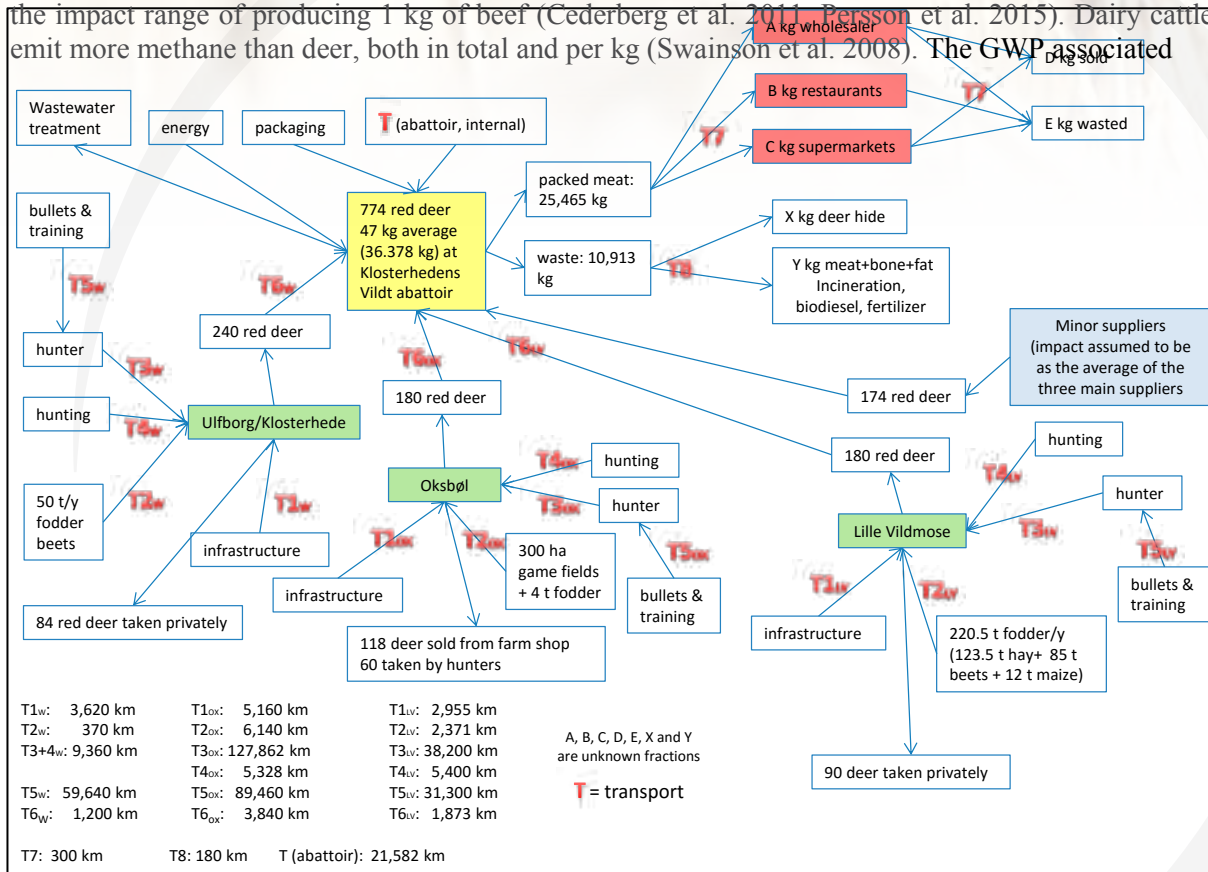


Figure 1. LCI of red deer venison production during 2010/11 involved 3 main suppliers (green boxes) that covered 78 % of the red deer venison produced by *Klosterhedens Vildt* abattoir. The environmental impact associated with minor suppliers (blue box) was estimated as an average of the main suppliers. For all suppliers, a share of the abattoir impact (yellow box), waste and transport to the consumer was added to arrive at the overall environmental impact.

Table 2. The GWP associated with Danish venison production.

Species and hunting grounds	Red deer, Ll.Vildmose	Red deer, Oksbøl	Red deer, NST West	Red deer, weighted average	Roe deer, Samsø	Fallow deer, Ørumgaard	Wild boar, Ll. Vildmose	Mallard, Frijsenborg	Pheasant, Frijsenborg	Beef, shop	Pork, shop	Chicken, shop
GWP, kg CO _{2eq} /kg venison	11.3	24.4	44.8	28.6	10.5	9.8	10.2	34.9	145.2	20-46	3.3	3.1

with meat production was three times larger for wild boar than for industrially produced pork, and 11 times larger for mallard, and 47 times larger for pheasant than for industrially produced chicken.

Figure 2 illustrates that the largest overall contribution to the monetized environmental impacts of venison was caused by fodder and foraging on farmers' fields and less on infrastructure and even on the excessive mileage driven by hunters to the selected hunting grounds, though many came from Eastern Denmark in their individual 4WD vehicles to hunt in Western Denmark. The monetized environmental impact of producing 1 kg deer venison was – like for the GWP (Table 2) – within or below the monetized impact range of producing 1 kg of beef, and the impact depended, besides on the deer species, on the origin of the venison, as illustrated for red deer. The monetized environmental impact of wild boar was twice that of pork, and for mallard and pheasant it was 19 respectively 61 times more than for chicken. The negative impact values for birds slaughtered at the abattoir were caused by high values of incineration of discarded specimen and waste, generating energy, substituting fuel elsewhere in the system. Of the 15 monetized impact categories, GWP and particulate pollution dominated the monetized impact for all species, with some contributions from

non-carcinogenic human toxicity and area use, and little or no contribution from the rest of the impact categories.

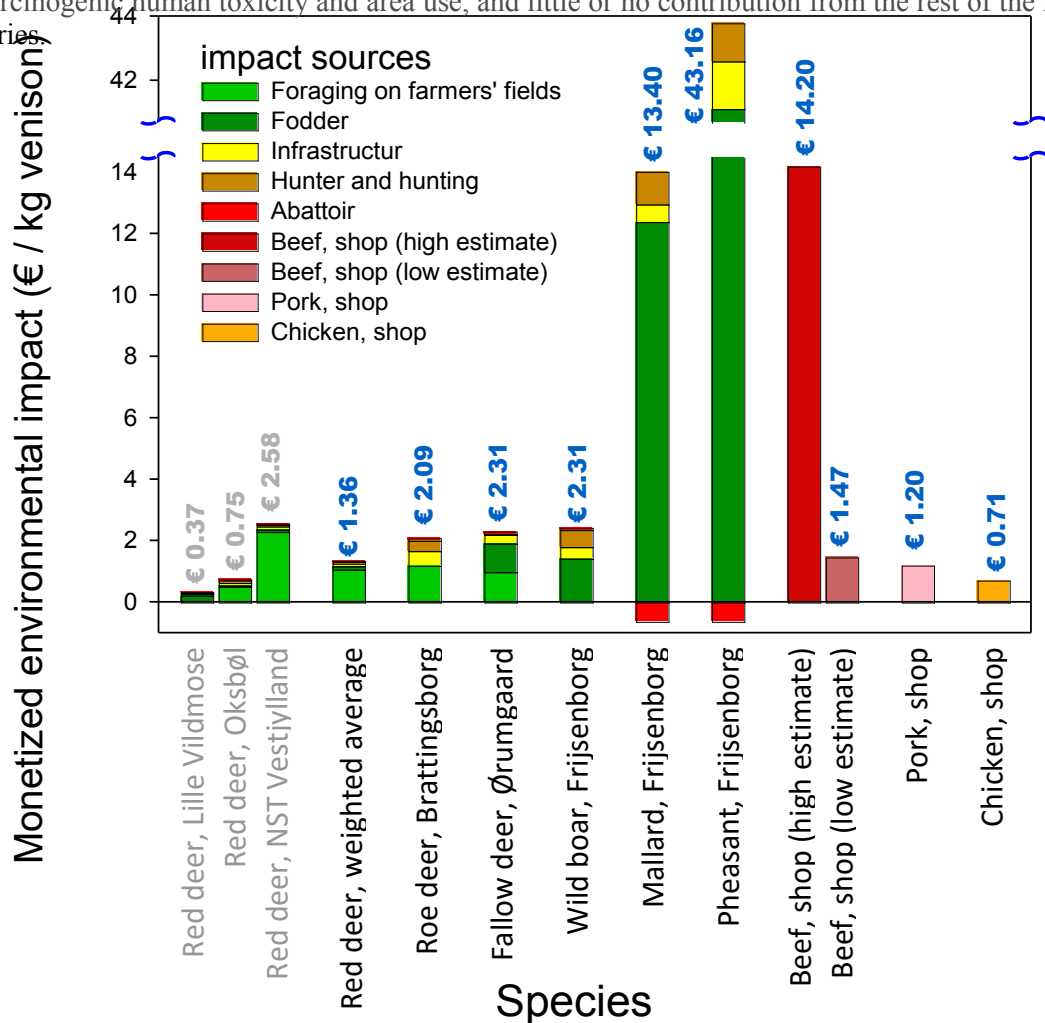


Figure 2. Monetized environmental impacts of red deer (from three locations), roe deer, fallow deer, wild boar, mallard, pheasant and commercially produced beef, pork and chicken. The individual impact contributions of fodder, feeding on farmers' fields, infrastructure, hunter and hunting and abattoir are distinguished by individual colour codes.

4. Discussion

The environmental impact of commercially available venison produced in Denmark has not previously been studied. The findings in this paper offered several new insights discussed below.

4.1 Transport

There was surprisingly much transport involved in the production of venison: Hunters driving to hunts, hunters driving to buy equipment and acquire their licence, and test-shooting their weapons every season; there was transport of fodder, transport of carcasses, transport of produce and transport of waste; transport of materials for the infrastructure, e.g. fences and shooting towers; and driving to check and repair fences after storms, and after hunting that could have caused e.g. red deer to run into and damage fences. For the 180 red deer originating from Oksbøl there was about 240,000 km of transport involved (Figure 1). But in terms of environmental impact, transport only made up a minor part of the overall impact associated with the commercial production of venison. Transport was mostly associated with the hunter/hunting including helpers and retrievers (brown colour in Figure 2).

4.2 Fodder and feeding/trampling on farmers' fields

The main environmental impact of commercially produced venison at Klosterhedens Vildt abattoir was what the animals consume, either when fed (as e.g. red deer and wild boar at Lille Vildmose or pheasant at Frijsenborg estate – dark green colour in Figure 2) and/or when trampling and foraging on farmers' fields (light green colour in Figure 2). The environmental impact of fodder and/or foraging on farmers' fields was from 1.3 (roe deer) to 20 (pheasant) times larger than the sum of all other impacts associated with venison production (green colours in Figure 2). For wild boar it was 1.4 times, for fallow deer 4.9 times, for red deer 6.7 times and for mallard 12.3 times larger.

The foraging and trampling on farmers' fields is a difficult component to estimate, and at the same time in this study it was found to be the most important component in the environmental impact of commercially produced deer venison. For free ranging red deer it averaged more than 90 % of what the deer consumed, and for fallow deer it was about 50 %, while for confined animals, wild boars and red deer at Lille Vildmose, and mallards and pheasants at Frijsenborg estate it was zero. Farmers, hunters and nature managers often have opposing interests in the magnitude of wild life populations, though farmers, even when losing crops may also enjoy hunting.

One conclusion from the above is that wild game and raised game most likely have very similar environmental footprints because they need equal amounts of feed/foraging. The distinction between the two in this respect is difficult to make; the transition between wild and raised game is gradual. In fact very few wild game individuals are truly wild in the Danish landscape – they are most likely living off a significant amount of agricultural crops one way or another. This may be different in e.g. Sweden where there are much larger natural fields and forested areas (Wiklund and Malmfors, 2014).

4.3 Commercial vs. privately taken venison

This study investigated only commercially produced Danish venison in order to compare the environmental impact hereof with commercially produced meat from cattle, pig and chicken. Some may assume that private hunting by individuals has a far smaller environmental footprint than the commercially produced venison from organized hunts described in this paper. However, in line with the above, since intentional feeding and arbitrary foraging on farmers' fields is the rule rather than the exception for the studied Danish game, even for raised fowl that escape their owners after months living under well-fed conditions, the sum of this feed typically constitutes the major part of the environmental impact of the venison, even the venison taken privately by individual hunters. Thus both commercial and privately taken venison must have a similar large environmental footprint. Furthermore, private hunting often results in a lower yield than organised hunting for commercial venison where 3-4 large animals may be bagged in a single two-man rifle hunt, or 30-40 birds in a single battue, while many hunters return empty handed from private hunts, thus driving many kilometres in vain.

4.4 Representativeness

How well do the results in this study represent all the commercial venison produced in Denmark? Klosterhedens Vildt abattoir produces the majority of Danish venison sold to consumers from Danish shops and restaurants, and by weight this is mostly red deer and wild boar. Kivan Food Ltd., the other major Danish venison supplier produces similar amounts, but mostly as fowl. The rest comes from minor suppliers.

In Denmark there are 50 species of mammals and 300 bird species that are in principle protected. But for 10 mammal species and 33 bird species hunting is permitted for some months every year. Annual statistics are prepared for all wildlife hunted and killed (Asferg 2014). In recent years the annual hunting yield has been around 2.6 million 'wild' animals. Roe deer, hare, fox, pheasant, wood pigeon and mallard are the most frequently hunted. In 2010/11 128,200 roe deer, 7,400 red deer, 6,000 fallow deer, 61,300 hares and wild rabbits, 39,300 foxes, 721,400 pheasants, 299,500 wood pigeons and 485,400 mallards were shot in Denmark (Asferg 2011).

According to the above numbers and to Table 1, Klosterhedens Vildt abattoir produced 10 % of all red deer shot in Denmark during the 2010/11 hunting season, 2 % of the fallow deer, 0.002 % of the roe deer (nearly all roe deer were taken privately, i.e. not sent to the abattoir), nearly all wild boar,

0.06 % of the mallards (nearly all taken privately) and 0.2 % of the pheasants (nearly all taken privately). Even though Kivan Ltd. produced most of the Danish bird venison, breeding and feeding of mallards and pheasants would be more or less the same as for birds delivered to Klosterhedens Vildt abattoir.

In conclusion, the environmental impact numbers in this study are reasonable representative for all commercial Danish venison, but less representative of the overall hunting yield in Denmark, where most is taken by the hunters themselves. But according to section 4.3, the environmental impact may not be much smaller for game taken by the private hunters compared to game sold for commercial production consumer. Venison sold by private hunters directly to consumers is legally a grey zone.

4.5 Positive and negative impacts of game and hunting - perspectives

The largest environmental impact of commercially produced venison is caused by feed consumption (Figure 2). The environmental benefits envisioned for venison from wild animals in terms of them being sustained by wild flora, i.e. foraging on areas that are not farmed, turned out to be a false pretence – and more so with some species than others. The production of venison from mallard and pheasant were proven to be respectively 19 and 61 times more harmful to the overall environment than chicken meat; production of wild boar venison was 3 times more harmful to the environment than pork, while production of deer meat was equally or less harmful to the environment than production of beef, but far more harmful than pork or chicken.

The bottom line is that there is little or no environmental advantage in choosing any type of venison over industrial meat, mostly because there are little or no savings on feed for wild animals compared with domestic animals, sometimes on the contrary – and extensively so. This is because of the inefficiency of most venison production. Feeding is inefficient because it is not targeted, and from a production point of view it is therefore often ‘wasted’ when other wild life consumes it, or it is left to decompose in nature. ‘Harvesting’ the ‘wild’ animals is also inefficient as many of the well-fed animals happily escape out of the production systems, as most clearly seen with mallards and pheasants, and as hunters compared with industrial butchers transport themselves over long distances, sometimes without taking home any game at all.

The damage to farmers’ fields and foraging on those fields has never been precisely accounted for, and may be underestimated by wildlife managers and hunters who may have an interest in trivialising this subject. Farmers may condone wildlife foraging or may even try to attract wildlife by putting out fodder if they want to hunt wildlife themselves. If farmers could agree that they would prefer to have less wildlife feeding off their fields, the Nature Agency could possibly (at a cost) supply more feed or plant fields inside forests to keep the deer in the forests and natural areas. Furthermore, the nature agency could put up more fences if they wanted to protect farmers’ fields and road traffic against deer collisions, and protect the deer against unwanted hunting, rather than only putting up fences to protect new plantations against deer browsing. These are charged questions as input to a currently ongoing debate on wildlife management and hunting in Denmark for which this study hopefully contributes with new knowledge.

4.6 Not enough venison to satisfy the New Nordic Diet

The existing populations of hunted wildlife in Denmark yielding approximately 1.4 g venison per Dane per day only satisfy a quarter of the recommended venison intake suggested by the New Nordic Diet (Meyer et al. 2011), even if the goods were evenly shared among Danish consumers and all were following the healthy recommendations. If the Danish venison production was to be increased, it would take significantly more feeding and fencing; and in that scenario it is a big question whether the game could still be considered free ranging wildlife, both in terms of animal welfare, human health, taste and environmental impact.

Animal welfare is also part of the ongoing debate on venison versus industrial meat. Meat from mallard and pheasant is extremely inefficient in environmental terms – they have very high environmental impacts per kg meat compared with chicken meat. However, both mallards and pheasants live at least a few months of their lives in nature, sometimes even years before they are killed. In contrast, commercial chicken meat comes from birds that typically never saw daylight or

truly natural living conditions. Deer and wild boar may also be considered experiencing superior animal welfare compared with cattle and pigs in conventional Danish animal production.

5. Conclusions

Venison typically has higher environmental impact than comparable industrial meat types. Regarding wild life, the reasons for this is a combination of the following: (1) inefficient feeding, (2) harmful foraging on farmers' fields, (3) loss of animals to carnivores, disease, or to freedom, (4) hunters' staggering mileage driving to hunting grounds, (5) the small scale production typical for venison. However, the fact, that deer emit less methane than dairy cattle work against the above.

From Table 2 and Figure 2 it is concluded that both the GWP and the overall (monetized) environmental impact of 15 impact categories associated with consumed meat may or may not improve if the consumers chose to eat commercially produced Danish deer venison rather than beef; it is less harmful to the environment to consume pork than commercially produced Danish wild boar, and much less harmful to the environment to consume chicken rather than commercially produced Danish mallard or pheasant. All in all, production of venison is much more harmful to the environment than presumed by diet recommendations like e.g. the New Nordic Diet (Meyer et al. 2011). Furthermore, realistically there can never be produced enough meat to satisfy the recommendations of the New Nordic Diet.

The wildlife is not out there just 'for the taking', free of environmental impact. On the other hand, if we did not take advantage of the available wild life for venison production, some may consider it a waste of resources. Others may prefer to enjoy the wild life without killing it - to live and let live.

Acknowledgement

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35.Reducing site-specific and life-cycle environmental impacts and achieving a price-premium for beef from a nitrogen-constrained catchment

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ABSTRACT

In the Lake Taupo catchment of New Zealand, all farms have a farm-specific limit on nitrogen (N) leaching per hectare. This study determined the site-specific and life-cycle emissions of N and carbon for beef production options from a case study beef cattle finishing farm in the catchment. Four main finishing farm scenarios were assessed. Three scenarios were optimised for profitability based on the farm remaining within the N leaching cap (< 20 kg N/ha/year); 1. a base farm with traditional flexible beef supply, 2. a farm with an assumed beef price premium and flexible beef supply, and 3. a farm with a beef premium and a system requiring steady supply of beef to meet retail requirements. The fourth scenario was based on no N leaching constraints and was optimised for profitability (including with higher production through the use of N fertiliser to increase pasture growth) with traditional flexible beef supply. Scenario analyses included evaluation of sourcing surplus young beef cattle from breeding farms or from dairy farms. A life cycle assessment method was used to estimate all reactive N emissions through the life cycle of beef. Leaching of N from the finishing farm was <20 kg N/ha/year from N-constrained scenarios and 42 kg N/ha/year from the scenario with no N leaching constraint. Profitability decreased with N constraints and further decreased with regular beef supply requirements, but the effects of this were countered by a price premium. The N footprint from beef production ranged from 93 to 155 g N/kg meat, being least from the N-constrained scenarios. The carbon footprint (14.4-21.4 kg CO₂-equivalents/kg meat) was lowest from the N-constrained scenarios 1 and 2, but highest for scenario 3 which required regular beef supply over time. Use of dairy-derived calves decreased the N and carbon footprints of meat compared to a traditional beef cattle breeding system due to allocation of significant breeding dairy-cow emissions to a milk co-product. The farm stage dominated the life-cycle N footprint (~79% of the total) with the only other significant contributor being the final waste (sewage) stage at ~21% of the total, based on a traditional urban waste-water treatment system. Land application of sewage to pasture for silage production and feeding to cattle, which is currently used for Taupo town sewage, was estimated to further decrease the N footprint of beef over the life cycle by ~20%.

Keywords: Carbon footprint, New Zealand, nitrogen footprint, nitrogen leaching, sewage.

1. Introduction

Water quality degradation in agricultural catchments with livestock farming has been linked to non-point source pollutants including the nutrients nitrogen (N) and phosphorus (P) (e.g. Howarth et al. 2002; Larned et al. 2004). In New Zealand, Lake Taupo is the largest lake and research has shown that while it is currently pristine with very high visibility, there is a very slow decline in water quality associated with increased N inputs and increased growth of phytoplankton (Vant and Huser 2000). Phosphorus levels in the lake have been stable over time and are non-limiting (or minor co-limiting) for phytoplankton growth.

Local government regulations for Lake Taupo have been in place since 2011 and this involves all agricultural land within the lake catchment having a maximum farm-specific N leaching value (WRC 2016). Pastoral farmland contributes >90% of the manageable N load and most of this is in relatively extensive sheep and beef farming. The regulations require farmers to manage their farm system and N inputs so that N leaching remains at or below the farm's N leaching limit or 'cap' (set according to farm system practices in 2001-2005; WRC 2016). The N leaching is calculated using a nationally-accepted and validated model (OVERSEER[®] nutrient budgets model, hereafter called OVERSEER; Wheeler et al. 2003).

These regulations mean that farmers are greatly restricted in their farming options for increasing production in the catchment. One enterprising farmer group has set up a company (Taupo Beef) that produces, processes and markets beef from a low environmental impact system and charge a price premium for the beef in local restaurants and national retail outlets (Taupo Beef 2016).

The aim of this project was to determine the site-specific and life-cycle emissions of N and carbon for beef production options on a case study Taupo Beef cattle farm with an N leaching cap in the Lake Taupo catchment. The economic implications on farm of a price premium linked with control of the value-chain to restaurants were assessed, to determine whether greater economic returns could counter the environmental constraints and costs. Effects of land application of waste from consumed meat (sewage) were also evaluated.

2. Methods

The case-study farm system in the Lake Taupo catchment of the Waikato region of NZ was based on a real cattle finishing farm (120 ha flat-rolling grassland) that purchases yearling beef cattle and sells them prime at about 2-years-old. Cattle (about 1.2 cattle wintered/ha) graze year-round on long-term pastures of perennial grasses and white clover on a coarse-textured pumice soil under relatively high rainfall (c.1300 mm/year).

Four finishing farm scenarios were assessed. Three scenarios were optimised for profitability based on the farm remaining within the N leaching cap (< 20 kg N/ha/year); 1. a base farm with traditional flexible beef supply (with main sales in summer and autumn when pasture availability decreased), 2. a farm with a 25 cents (NZ)/kg beef (carcass weight) premium and flexible beef supply, and 3. a farm with a 25 cents/kg premium and a system requiring steady supply of beef to meet retail requirements. Thus, differences for scenarios 1-3 relate to the stocking rate and timing of cattle purchases and sales to meet pasture growth pattern and requirement for beef supply (scenario 3), all with no N fertiliser use (Table 1). The fourth scenario was based on no N leaching limit and was optimised for profitability (e.g. permitting higher production through the use of N fertiliser to increase pasture growth) with traditional flexible beef supply. All four scenario analyses were based on the yearling cattle purchased from a traditional beef farm (based on an average Class 4 North Island hill sheep and beef farm assumed to be in the Waikato region; Beef+LambNZ 2010) or derived from surplus dairy calves reared to weaning on milk powder, grain and pasture, and then reared from weaning to one-year-old on a traditional beef farm.

Farm system analyses included farm production and economics (using the INFORM farm systems model; Rendel et al. 2013) and N leaching from the breeding and finishing farms (using OVERSEER). Profitability of the finishing farm was calculated as earnings before interest, taxes, depreciation and amortization (EBITDA).

A cradle-to-grave LCA was based on cattle produced and processed in the Taupo/Waikato region and consumed in a restaurant in the Taupo town. The functional unit was 1 kg meat consumed, while the reference unit at the farm-gate was 1 kg live-weight (LW). Co-product handling was based on biophysical allocation for meat and milk from dairy cattle on-farm and economic allocation for meat and co-products (e.g. tallow, hides, renderable non-edible product) during processing (LEAP 2015; Lieffering et al. 2012). An LCA-based N footprint was calculated, which accounted for loss of all reactive N forms including leached N (dominated by nitrate), ammonia (NH_3), nitrous oxide (N_2O) and other nitrogen oxides (NO_x , e.g. from fossil fuel use). The NZ greenhouse gas (GHG) inventory methodology (MfE 2015) was used in calculating NH_3 emissions and N_2O emissions, with the latter including direct and indirect (from NH_3 and leached-N) emissions, from the breeding and finishing farms. Environmental emissions from the background processes were derived from Ecoinvent database version 3.1, but were modified using NZ-specific data where available. The four farming scenarios were modelled using SimaPro (version 8.1.0.6), for the production, transport and use of farm inputs including N fertilisers. Emissions associated with meat processing (based on primary data

from NZ plants, including waste-water processed via multi-pond treatment and output to waterways; Lieffering et al. 2012), transport (100 km from farm to plant, and from plant to consumer), consumption (including refrigerated storage for 1-day and cooking using natural gas) and waste treatment stages, based on consumption in a restaurant in the Taupo town were accounted for. The ‘waste treatment’ stage refers to meat-N excreted into the sewage system after meat consumption and assumed to be processed using a typical NZ municipal waste water treatment system, with secondary processing and discharge to surface water (based on Muñoz et al. (2008), with NZ-specific modification). Data from all life cycle stages was also used to determine the carbon footprint of meat using methods outlined in LEAP (2015).

A life cycle scenario analysis of the waste treatment stage assessed the effect of the treated waste water being recycled by application onto land and used in pasture silage production systems instead of discharging it to surface water. This land application of sewage (after secondary processing) for silage production and use of the nutrients for pasture silage production on farms. The latter was based on the actual sewage processing system used by the Taupo town, with secondary-treated waste-water applied on land outside the Lake Taupo catchment. A system substitution method was used in the analysis, whereby nutrients (N and P) in the treated waste water were assumed to displace nutrients from chemical fertilisers that are typically used for silage production, based on LCA data for NZ fertiliser production by Ledgard et al. (2011).

3. Results

On the finishing farm, N leaching per hectare was calculated (using OVERSEER) at 18-19 kg N/ha/year in the N-constrained scenarios and 43 kg N/ha/year for the non-N-constrained base scenario, respectively. Profitability (EBITDA) decreased by 32% for the Base farm in the N-constrained scenario compared to the corresponding unconstrained scenario (Table 1). A premium of 25c/kg carcass weight on the N-constrained farm resulted in a similar profit to that for the unconstrained base farm. However, profit was much lower when that same farm had to manage a system that required a regular supply of finished cattle to meet the required retail supply (i.e. \$457/ha for scenario 3 compared to \$738/ha for scenario 2). This was associated with a requirement for more frequent purchases of young cattle over time to match pasture growth and availability.

The net amount of live-weight (LW) sold from the finishing farm was lower under the N-constrained scenarios than the non-N-constrained scenario (Table 1). While this was associated with lower N leaching from the N-constrained farm scenarios, it resulted in a greater land requirement for production of yearling cattle from the breeding farm. Overall, the land requirement was lowest for the more intensive non-N-constrained scenario.

Table 1: Description of some farm characteristics for four scenarios for a 120 ha finishing farm in the Lake Taupo catchment with an N leaching constraint or no N constraint, optimised for profitability. Yearling beef cattle were purchased for the finishing farm from either a traditional beef breeding farm or a dairy farm (surplus dairy calves reared off-farm and raised to yearlings on a breeding farm).

	Constrained N leaching			
	1	2	3	4
	Base	Base + Premium	Base + Premium +Supply	Base (no N +constraint)
Finishing farm only:				
Fertiliser N (kg N/ha/yr)	0	0	0	160
t live-weight brought-in	147.3	147.2	232.5	122.8
t live-weight sold	241.5	241.2	302.5	227.3
t net live-weight sold	94.2	94.0	70.0	104.5
Profit NZ\$/ha/yr	\$488	\$738	\$457	\$725
Finishing + Breeding farm:				

Breeding farm area needed (ha)	276	276	461	223
Finishing +Dairy-derived Breeding farm:				
Breeding farm area needed (ha)	148	149	302	105
Dairy farm area needed (ha) ¹	18	18	23	17

¹ Area for production of milk powder used for rearing calves from 4-days-old to weaning

For the finishing farm, the losses of reactive N were mainly from N leaching (predominantly nitrate) and NH₃ emissions (from animal excreta and fertiliser-N), which were of a similar magnitude (Figure 1). The N footprint of LW sold decreased slightly from scenarios 1 to 3, but was about two-fold higher for scenario 4. When the breeding farm component was included there was little difference in the N footprint of LW sold between all N-constrained scenarios, while it was approximately 50% higher for the non-N-constrained scenario 4 (Figure 2a). The N footprint of LW sold for beef derived from dairy farms was 7-12% lower for the N-constrained scenarios compared to that for beef from the traditional breeding farm and 5% lower for scenario 3 (Figure 2a).

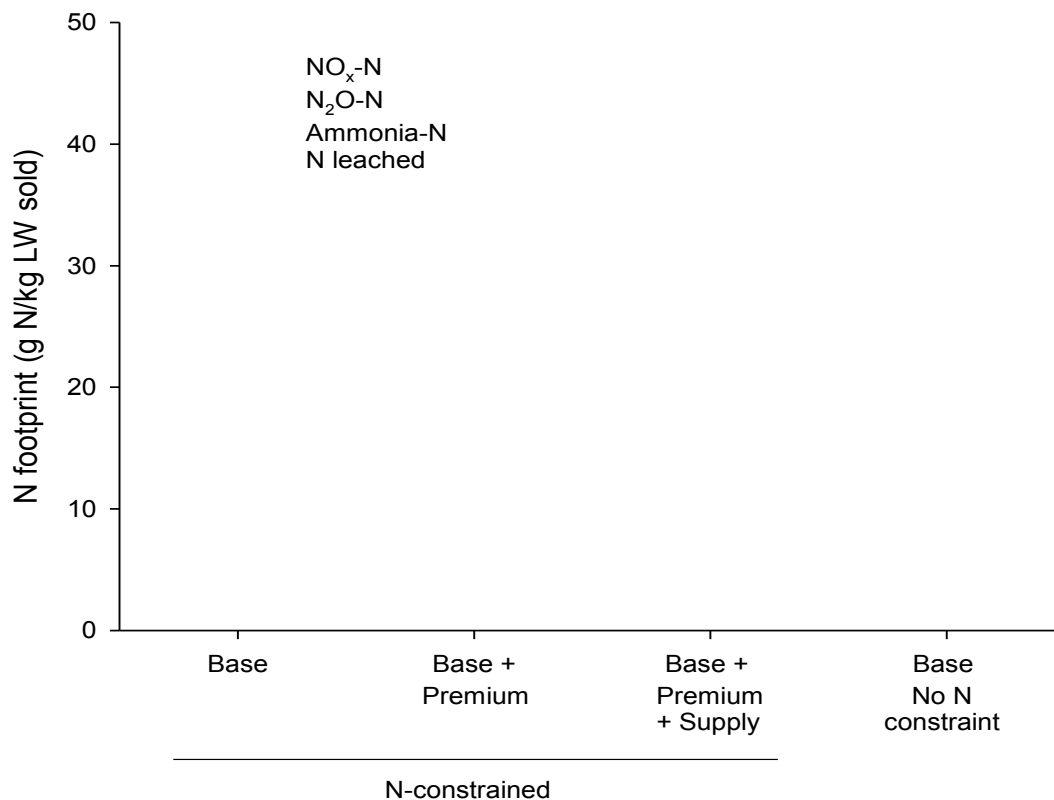
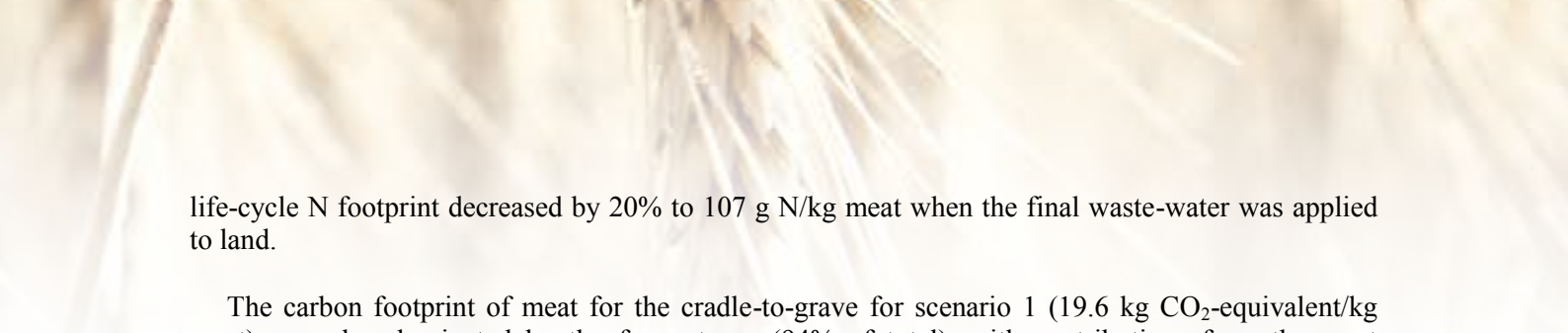


Figure 1: N footprint (i.e. reactive N loss from leaching, ammonia, N₂O and NO_x emissions) per kg live-weight (LW) sold for a 120 ha finishing farm in the Lake Taupo catchment for four scenarios with or without an N leaching constraint, optimised for profitability.


The carbon footprint of LW sold for the cradle-to-farm-gate showed similar results across all scenarios for the cattle from the breeding farm, being slightly higher for scenario 3 (Figure 2b). However, for all scenarios, it was 18-21% lower for dairy-derived cattle than for cattle from the breeding farm.

Assessment of the cradle-to-grave N footprint for scenario 1 (134 g N/kg meat; derived from traditional breeding farm) showed contributions of 79% from farm stages, <1% from each of the meat processing, transport and consumer stages, and 21% from the waste treatment (i.e. sewage) stage, based on a municipal processing system where final waste-water goes to waterways. In contrast, this



life-cycle N footprint decreased by 20% to 107 g N/kg meat when the final waste-water was applied to land.

The carbon footprint of meat for the cradle-to-grave for scenario 1 (19.6 kg CO₂-equivalent/kg meat) was also dominated by the farm stages (94% of total), with contributions from the meat processing, transport, consumer and waste stages contributing 2.3, 0.2, 2.8% and 0.5%, respectively. Land application of waste-water was estimated to decrease the life-cycle carbon footprint by 0.3% to 19.5 kg CO₂-equivalent/kg meat.



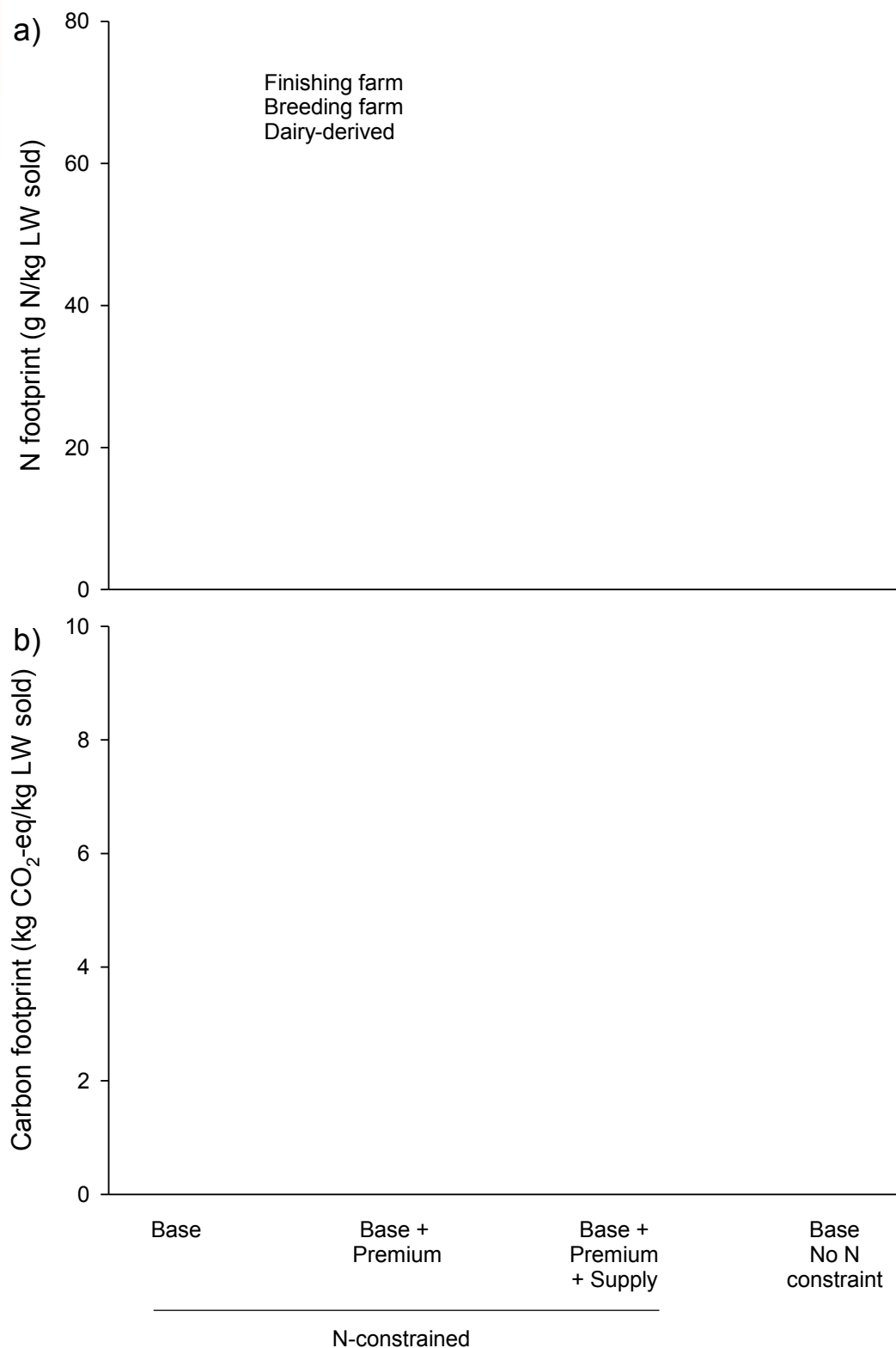


Figure 2: a) N footprint and b) carbon footprint per kg live-weight (LW) sold for the cradle-to-farm-gate for four scenarios with or without an N leaching constraint, optimised for profitability. Cattle for the finishing farm were either derived from a traditional breeding farm (left columns) or from surplus dairy calves reared off-farm and raised to yearlings on a breeding farm (right columns).

4. Discussion

All N-constrained scenarios for the finishing farm had been optimised to remain below the N leaching cap of 20 kg N/ha/yr. This resulted in the base N-constrained farm scenarios having less than one-half the N leaching per-hectare compared to the corresponding non-N-constrained scenario but having one-third lower profitability. This highlights the impacts faced by farmers in the Lake Taupo catchment where there is a limit on N leaching losses. The case study farmer had changed their farm practices over time, including ceasing use of N fertiliser and relying on clover N₂ fixation, but one of the largest changes was a shift away from having breeding cattle on farm to sourcing one-year-old cattle for finishing. This change avoided the need for breeding cows which produce only 0.9 calves/cow on average per year in NZ and contribute significantly to farm N leaching via their excreta deposited during grazing. This finishing farm system has enabled an increase in product output (LW sold) and profitability while meeting their N leaching cap. However, the one-year-old cattle must be produced somewhere and the main source is from hill country sheep and beef farms, which were used in this analysis. These hill country farms are also relatively low N emitters since they rely on perennial grass/clover pastures, use little N fertiliser (c. 10 kg N/ha/year) and use year-round grazing with no animal housing, no brought-in feeds and minimal use of on-farm forage crops or pasture silage/hay (Beef+LambNZ 2010). Their average N leaching is 15 kg N/ha/year (Ledgard et al. 2014).

Overall, the N footprint for LW sold, covering the cradle-to-farm-gate stages, was low at approximately 42 g N/kg LW sold for the N-constrained scenarios and 62 g N/kg LW sold for the non-N-constrained scenario. This was based on cattle derived from traditional beef and was lower at 38 and 59 g N/kg LW sold for the dairy-derived beef, respectively. This 5-12% decrease can be attributed to most of the maintenance-related emissions from the breeding cow being allocated to milk for dairy cows, with the surplus calves being a minor co-product. These latter values equate to approximately 95 and 148 g N/kg boneless-meat, and are much lower than the average values reported for European beef at about 700 g N/kg meat (Leip et al. 2014). The latter was based on a 'top-down' approach using full N flows and so was a more indirect estimate of reactive N emissions. However, higher values for European cattle systems would be expected based on their use of N fertiliser, annual crops, and cattle housing systems. These housing systems require manure collection and application, which result in much larger ammonia emissions compared to that for excreta deposited directly onto soil in grazing systems (e.g. Jarvis and Ledgard 2002).

Carbon footprint analysis showed little difference between scenarios, but average values were 18-21% lower for dairy-derived cattle than for cattle from the breeding farm. Again, this reduction can be attributed to allocation of most of the maintenance-related enteric methane emissions from the breeding dairy cow to milk rather than to the surplus calf. The values for dairy-derived beef for the cradle-to-farm-gate stages equate to approximately 15-19 kg CO₂-equivalent/kg boneless-meat, which are of a similar order to that estimated for average European beef of 23 kg CO₂-equivalent/kg meat (Lesschen et al. 2011) and within the range for 16 studies across OECD countries of 14-32 kg CO₂-equivalent/kg meat (de Vries and de Boer 2010). This similarity in carbon footprint of meat across different systems is probably due to the dominant effect of cattle enteric methane associated with feed consumption, which will be broadly similar for pasture or crop-based feeding systems.

The cradle-to-farm-gate stage dominated the life-cycle N footprint (~79% of the total) with the only other significant contributor being the final waste (sewage) stage at ~21% of the total, based on a traditional urban waste-water treatment system. Land application of sewage to pasture for silage production and feeding to cattle, which is currently used for Taupo town sewage, was estimated to decrease the N footprint of beef over the life cycle by ~20%. This illustrates that if the town had discharged its treated wastewater to the lake it would have been a major contributor to potential water quality deterioration and that land application is an efficient option for N recycling and saving on fertiliser requirements. In contrast, the waste-water stage had a minimal contribution (0.5%) to the life-cycle carbon footprint, while the farm stages constituted about 94% and provided the main reduction potential.

From an LCA perspective, a carbon footprint relates to one environmental impact (climate change) whereas an N footprint sums contributing reactive N sources on a g N basis and does not relate to any one specific environmental impact category. Indeed, the N sources contribute differently to a range of different impact categories (EC-JRC 2011). Thus, it does not align well to LCA methodology. However, it has been popularized by relating an N footprint to a human's contribution to the environment in a general sense through their various N emissions (e.g. Leach et al. 2012). An N footprint is broadly similar to the marine eutrophication potential indicator (EC-JRC 2011), except that for the latter each of the contributing reactive N sources has a different characterization factor depending on the source and fate. Paradoxically, the freshwater eutrophication potential indicator (EC-JRC 2011) is only driven by phosphate emissions and for Lake Taupo, which is a large freshwater lake the concern about actual eutrophication and local government regulations are based on N as the recognized source of water quality degradation (WRC 2016). Payen and Ledgard (2016) expand on this important area of site-specificity of LCA-based water quality indicators and recognize the appropriate contributing sources, including using Lake Taupo as a case study.

5. Conclusions

A case-study beef farm system, which was optimised to meet a site-specific N leaching cap, was found to use a range of farm practices to increase N conversion efficiency into meat product and achieve a low N footprint across the cradle-to-farm-gate stages. While the finishing farm had a low N leaching loss and low associated N footprint, the farm profitability was less than what could have been achieved if there was no N-leaching constraint. The case study farmer markets their Taupo Beef at a premium based on its low impact on Lake Taupo, but this required a price premium to counter the cost of the increased farm system complexity to regularly supply their beef to restaurants.

Whole farm system analysis required accounting for the breeding farm to produce the young cattle for the finishing farm. Use of dairy-derived calves decreased the N and carbon footprints of meat compared to a traditional beef cattle breeding system due to allocation of significant breeding cow emissions to a milk co-product and avoiding the sole need for a beef breeding cow.

While the farm stages dominated the whole life cycle, the only other significant contributor was the final waste stage for consumed meat-N excreted into the sewage system. Land application of sewage to pasture for silage production and feeding to cattle, thereby substituting for fertiliser and closing the beef life cycle, provided a further large decrease in the N footprint of beef.

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52. Environmental impact of different types of Danish beef production - focus on meat processing

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ABSTRACT

The objective of the work was to examine the environmental impact of 10 Danish beef production systems covering the entire chain from the farm until the edible products and side streams leave the slaughterhouse. The functional unit '1 kg edible product' was used. For each type of beef, data from slaughterhouse was used to divide live weight into edible products, hides, other by-products and specified risk materials. The following impact categories were considered; carbon footprint, land use, biodiversity damage, primary energy use and eutrophication. The primary production accounts for a major share of the impact per kg edible products. The slaughtering process itself is very energy- and resource efficient, when taking into account the alternative use of by-products and hides. The environmental impact of dairy-based beef was lower than for meat from beef breeds. There is a huge potential for reducing the environmental impact per kg edible product if it is possible to obtain a higher utilization of the slaughtered animal by producing new edible products not conventionally produced.

Keywords: Slaughtering, Edible products, Carbon Footprint, Biodiversity damage.

1. Introduction

Meat is an important part of the human diet and at the same time one of the foods carrying a high environmental footprint. Beef is in particular perceived as having a high environmental footprint, but at the same time there are huge differences in the way different beef products are produced at the farm, and it is well known that this impacts to a high degree on the environmental profile. While a number of studies have been carried out at the farm level and translated into environmental impact of the carcasses produced, comparatively less is known on resource use and exploitation of the carcass at the slaughterhouse from different types of cattle. Since only about half of the weight of the living cattle is present in the carcass, and differs between the types of cattle, the translation of the impact related to the live animal and the products produced is not straight forward.

Beef is produced in many different ways. A main distinguishing is between meat from dairy cattle and meat from specialized beef breed cattle. While the dairy cattle breeds are mainly for milk production less importance have been put on the quality of the carcass for beef production. Contrary in the specialized beef production systems the quality of the carcass has been given attention, but huge differences exist in types of cattle breeds optimized for carcass quality and cattle breeds that are robust and can rely on relatively poor feeding. Another important distinguishing is that the cattle are slaughtered at different ages.

The objective of this study was to examine the environmental impact of 10 Danish beef production systems covering the entire chain from the farm until the edible products and side streams leave the slaughterhouse with the main focus was on the slaughtering stage.

2. Methods

The functional unit for the life cycle assessment was '1 kg edible product'. The system covers the primary production **at the farm where the animals are raised and the slaughtering process**. This study includes 10 specific types of cattle delivered for slaughtering. They originate from two categories of production systems: beef from dairy production and beef from beef cattle breeds. The dairy system includes the dairy cow, heifer calves raised for replacement and bull calves raised for slaughtering; breeding is based on artificial insemination. Three types of beef production are based on

male calves from dairy production slaughtered at 8.9, 13.5 and 26.3 months respectively. In the first two systems the calves are fed intensively and housed indoor, in the last system, the bull calves are castrated and the system is more extensive based on grazing during summer. A fourth system is the dairy cow when slaughtered and replaced with a heifer reared in the herd. The beef breed systems consists of a suckler cow with a heifer raised for replacement and a calf weaned at 6 months of age and then raised in a separate fattening unit, breeding is based on artificial insemination. Two different beef breeds were included: Highland and Limousin representing typical extensive and intensive beef breed production systems, respectively. Input and output data and emissions related to the primary production were based on Mogensen et al. (2015) where further detail can be found. Key inputs of the primary production include feed, straw, electricity, heat and a calf in the dairy based systems.

The live animal is transported from the farm to the slaughterhouse where it is processed into four types of products: edible products, hides, a variety of other by-products that can be utilized and specified risk materials (SRM) that need to be destroyed (Table 1). The slaughtering process consumes heat, electricity and water and generates wastewater that goes to treatment. The manure produced by the animal during transport and in the slaughterhouse was assumed utilized for biogas production and afterwards applied to fields as fertilizer. Slaughterhouse data were based on Mogensen et al. (2016).

This LCA includes the following impact categories: carbon footprint (CO₂-eq), land occupation (m²), primary energy use (MJ), and eutrophication (NO₃-eq.). The effect on biodiversity (PDF) from producing different types of beef products was estimated according to Knudsen et al. (2016). By this method the number of vascular plants is used as a proxy for biodiversity due to the relation between number of plant species and other organisms in the agricultural land scape. As a sensitivity analysis, contribution of GHG emissions from soil carbon changes and indirect land use change was included. The contribution from carbon changes in soil was calculated using the method described by Pedersen et al. (2013), where the type of crop grown affects whether C is sequestered or released. The indirect land use change effect (iLUC) was estimated according to Audsley et al (2009) with an average iLUC emission factor of 143 g CO₂/m² used for crop production. An exception was the use of permanent pastures and natural areas, which we assumed do not contribute to iLUC, since these areas do not have an alternative use like cultivation of another crop.

Both in primary production of beef and at slaughterhouse, it was necessary to distribute the environmental burdens of the production process to various co-products. In primary production, we used a modification of step 2 from the ISO 14044 standard (ISO, 2006), i.e., allocating the total environmental impacts to the different products based on the underlying physical connection between them, here feed consumption. With regard to handling co-products from the slaughtering process, it was possible to use system expansion for those co-products that are used either as feed or for biogas production, whereas for the hides we used economic allocation.

From Table 1 appears that between 44.6-57.5% of the amount of live weight end up as edible products (Pontoppidan and Madsen, 2014). These numbers represent the 'actual utilization' of the slaughtered animals at real markets. As a potential mitigation option, Pontoppidan and Madsen (2014) have estimated amount of edible products for 'an optimal utilization' of the slaughtered animal. That includes by-products that at the moment are used for something else, but has a potential for use for human consumption at a global market in the future. To reach that optimal utilization, increased demand is needed for these products. Probably these by-products also need some treatment before sale. Such possible extra resources were, however ignored in the calculations.

Table 1. Input and output from the slaughtering process of one animal for 10 types of cattle, kg ⁶⁾

Type of cattle Months ¹⁾	Dairy				Beef breed			Beef breed		
	Holstein		Frisian		Highland			Limousin		
	Calf	Bull	Steer	Cow	Bull	Heifer	Cow	Bull	Heifer	Cow
	9	14	26	65	18	24	91	14	20	95
Input										
Live weight, kg	391	458	611	653	432	354	436	533	504	687
Electricity, kWh ⁷⁾	34	37	42	41	36	33	35	43	41	47
Natural gas, kWh ⁸⁾	29	29	29	29	29	29	29	29	29	29
Water, l ⁹⁾	686	686	686	686	686	686	686	686	686	686
Output, kg										
Total edible products ²⁾	194	222	290	294	210	172	204	304	276	361
- Meat without bones	154	176	228	218	168	133	155	260	228	294
- Edible by-products	35	40	55	71	38	35	45	39	43	62
- Bones for food	4	5	7	5	5	4	4	5	6	5
Other by-products ³⁾	149	170	230	254	159	140	165	163	176	229
SRM ⁴⁾	18	32	43	64	29	22	42	31	27	63
Hides	30	35	48	40	33	19	25	34	24	35
Optimal utilization ⁵⁾										
Total edible products	245	287	379	381	272	223	263	366	342	444

1) Age at slaughtering, months

2) Total edible products is the sum of Meat without bones - sold, edible by-products and bones for food

3) By-products used for animal feed, for biogas production, and for other use like medicine

4) Special risk material for destruction,

5) As a mitigation option, amount of edible product by an optimal utilization of the slaughtered animal was estimated

6) Based on Mogensen et al., 2016

7) Consumption of electricity for cooling and other operations is influenced by weight of the slaughtered animal (Pontopidan & Madsen, 2014)

8) Natural gas for heating the buildings and for hot water production is assumed to be the same per animal slaughtered for all systems (Pontopidan & Madsen, 2014)

9) Water for handling the slaughtering and for cleaning is assumed to be the same per animal slaughtered for all systems (Pontopidan & Madsen, 2014)

3. Results and discussion

It appears from Table 2 that there are significant differences in environmental impact for the different types of beef but also that the different impact categories rank differently. Beef from young cattle from the dairy system has a lower environmental impact than beef from young cattle from a beef breed system across all impact categories, except biodiversity damage. Thus, carbon footprint and eutrophication amounts about 1/3 in the dairy system compared with the beef systems. On the other hand the beef system beef has in fact a negative biodiversity damage index, which means that this system actually contributes to improved biodiversity. Beef from Highland cattle shows a higher carbon footprint and a better impact on biodiversity than beef from the Limousin, which is related to the fact, that these animals are assumed to graze natural grassland. Beef from adult cattle includes beef from steers and beef from culled cows. Among the different types of cows, only small difference is seen in carbon footprint and eutrophication. Beef from beef cows require a lower expenditure of primary energy and also impact positively on biodiversity compared to beef from dairy cows. Looking across all types of beef only small differences exist in carbon footprint within the dairy based systems, except that the carbon footprint of beef from steers are considerable higher than from the other types.

In table 2 impacts related to changes in soil carbon and to indirect land use changes were not taken into account, since it is generally agreed that these impact should be reported separately. However the impact can be very different for different types of beef systems, and therefore the importance hereof for the carbon footprint has been estimated as well. Grassland based systems sequester carbon and thus reduce the carbon footprint compared to systems based on arable crops. Emissions related to indirect land use changes (iLUC) are related to the occupation of land which can be cultivated. In Figure 1 is shown the importance of including soil carbon sequestration and indirect land use in the assessment of different types of beef. In general the carbon footprint of the dairy based calf and cow beef are increased by 11-19 % when including these impacts, while for the beef based systems these two impacts are to a certain degree counter balanced. Thus, the differences between beef from different systems tend to diminish.

If it is possible to increase the amount of the live weight that is utilized as edible product, it can have huge impact on the environmental impact measured per kg edible product. The suggested optimized utilization (Table 1) results in a 17 to 23% lower GHG emission per kg edible product. For example for a Holstein veal calf, at present 50% of the 391 kg live weight of the animal is utilized as edible products. With optimal utilization it was estimated that 63% of LW could be utilized. If that is possible, carbon footprint per kg edible product could be reduced by 20 % from 10.4 to 8.3 kg CO₂/kg edible product.

Table 2. Environmental impact from beef production per kg edible product ('meat')

Type of cattle	Dairy Holstein Frisian				Beef breed Highland			Beef breed Limousin		
	Calf	Bull	Steer	Cow	Bull	Heifer	Cow	Bull	Heifer	Cow
Carbon footprint (CF), kg CO₂-eq.										
Prim. Production	10.6	10.6	19.8	11.3	42.7	46.4	13.1	31.5	31.1	11.5
Slaughtering process	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.1
By-products	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1
Hides	-0.2	-0.2	-0.5	-0.2	-0.9	-0.7	-0.3	-0.5	-0.4	-0.2
Total CF	10.4	10.5	19.4	11.1	41.9	45.8	12.9	31.0	30.8	11.3
Average system²⁾	10.9				36.2			25.3		
Land use, m²										
Total ³⁾	14.1	15.5	19.9	12.7	168.6	240.5	62.9	55.1	57.1	21.7
Arable	14.1	15.5	19.9	12.7	18.3	19.0	5.1	25.3	21.5	7.5
Biodiversity damage										
PDF-index ¹⁾	7.2	8.1	1.7	4.6	-50.6	-77.0	-19.9	-4.4	-10.3	-4.3
Primary energy use, MJ										
Total	36.0	38.5	28.6	30.2	27.4	28.6	7.5	37.2	30.1	9.9
Eutrophication, NO₃-eq g										
Total	794	773	1460	710	3273	3143	835	2281	2140	776

1) PDF-index = average PDF/m² for each crop in feed ration *total land use (m²). A positive number means BD loss

2) Average meat output from either the dairy system, the Highland beef breed system or the Limousine beef breed system, i.e. average weighted composition of meat produced in each system

3) Including arable land, permanent pasture and nature grass

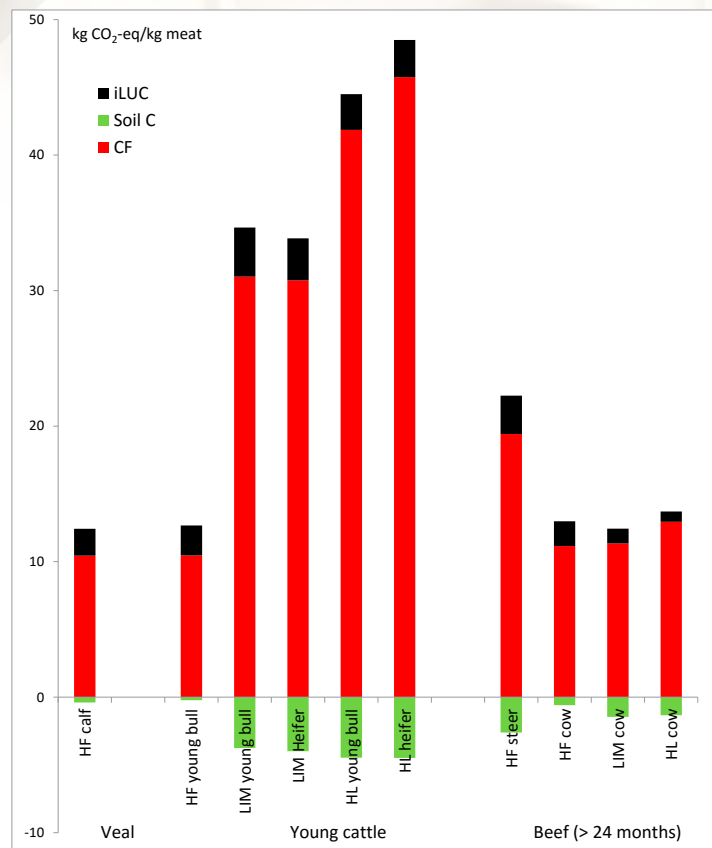


Figure 1. Carbon footprint (CF) without taking into account soil carbon changes (Soil C) and indirect land use change (iLUC), and the contribution from Soil C and iLUC for the 10 beef products.

Figure 2 illustrates the overall environmental impact of average beef produced in one of the three overall systems: the dairy system and the two beef breed systems; Highland and Limousin production system. It can be seen that carbon footprint, eutrophication and land use follow the same picture due to the relation to use of feed per kg edible product produced. Therefore these emissions are lower in the most efficient system, the dairy system with the lowest feed use per kg edible product and highest in the extensive Highland cattle system with a high feed use per kg edible product. There are only smaller differences in energy use between systems per kg edible product, whereas the effect on biodiversity differs to a high degree between the three systems. There is close to a neutral effect on biodiversity of the dairy system compared with the reference system, an average semi-natural forest in Europe, a small positive effect of beef production in the Limousin system and a large positive effect on biodiversity in the extensive Highland system.

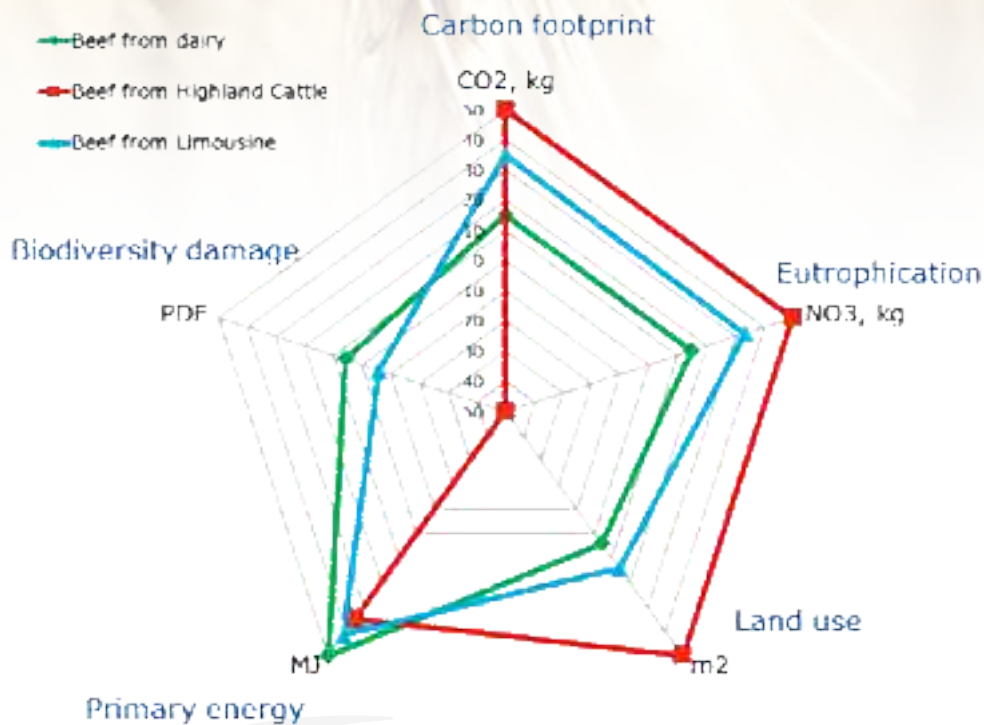



Figure 2: Environmental impact of average beef from the dairy system, the Highland and the Limousine production system (average for different types of cattle from each system: calf, young bull, steer, heifer and cow) – lowest impact in the centre for the figure

4. Conclusions

The major environmental burden of beef is related to the farm level stage and innovations to reduce impact should be given high attention. **The production of dairy-based beef results in a lower carbon footprint and a lower eutrophication per kg edible products than of beef from beef breed cattle. Beef from beef breeds, especially from the extensive Highland cattle system on the other hand, has a very positive effect on biodiversity.** The slaughtering process itself is very energy- and resource efficient. A major innovation to reduce environmental impact of the meat produced will be to ensure a higher utilization of the animal into new edible products not conventionally produced. Also, for beef products there is a significant tradeoff between impact on GWP and impact on biodiversity.

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125. Land use efficiency of beef systems in the Northeastern USA from a food supply perspective

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ABSTRACT

One widely recognized strategy to meet future food needs is reducing the amount of arable land used for livestock feed production. Of all livestock products, beef is the largest land user per unit output. Whether beef production results in feed-food competition or a net positive contribution to the food supply, however, may depend largely on whether marginal land is used for forage. Van Zanten et al. (2015) developed the land use ratio (LUR) to identify livestock systems that provide more food than could be produced by converting their suitable feed land to food crops. While Van Zanten et al. (2015) used country and farm-level suitability data, the former may not be high enough resolution for large countries, and the latter may not be available in many countries. We developed a method that integrates geospatial data to estimate crop suitability and yield at multiple scales into the LUR, broadening its applicability. We illustrated this approach with grass-fed beef (GF) and dairy beef (DB) case study systems in the Northeastern USA, including multiple scenarios limiting land conversion. All systems had LURs greater than one, indicating they produce less protein than conversion of their suitable land bases to food cropping would. Because a large fraction of the forage land used in the GF system was suitable for crop production and moderately productive, its LUR was 3-6 times larger (less efficient) than the DB system. Future research should explore mechanisms to improve LUR and life cycle environmental burdens of these regional production systems.

Keywords: Land use ratio, beef production, food security

1. Introduction

Over one-third of global land is used for agriculture, the majority of which (75%) is dedicated to livestock (Foley et al., 2011). To meet future food needs in a sustainable way, critical examination of the current allocation of land to livestock is necessary (Foley et al., 2011; Garnett, 2009). Van Zanten et al. (2015) developed the land use ratio (LUR) to identify livestock systems that provide more food than could be produced by converting their suitable feed land to food crops – a perspective that is not yet addressed in LCA. Systems that utilize byproducts and/or land unsuitable to cultivate food crops can be efficient in terms of human protein production (van Zanten et al., 2015). For example, beef production systems that rely on marginal land may result in a net positive contribution to the food supply (de Vries et al., 2015; Eisler et al., 2014; van Zanten et al., 2015). However, roughly half of global pasture land is marginal, with the other half suitable for food crops (van Zanten et al., 2016). Whether the production of beef results in feed-food competition or a net positive contribution to the food supply may depend largely on whether marginal land is used for forage.

To estimate the land use ratio (LUR) for dairy systems in the Netherlands, Van Zanten et al. (2015) combined data on land suitability and food crop yields at the country level for purchased feeds and at the farm level for grassland. Country-level data may be appropriate for the Netherlands, which has a small and relatively homogenous agricultural land base, and for imported feeds when the region of origin is unknown. However, this approach is less precise for production systems in large countries with diverse land uses, geographies, and climates, such as the USA. Even if such a country is suitable for production of food crops, its regions may have quite different capabilities. Additionally, the grassland data available at the farm level in the Netherlands may not be available in other countries. Because ruminant production systems may occupy large grassland areas, having representative suitability data is critical to determine their LUR. Therefore, an intermediate approach between farm and country scale estimation is needed to make the method more widely applicable.

Our primary objectives in the present work were to: (i) enhance the LUR method developed by Van Zanten et al. (2015) by incorporating geospatial analyses to assess land suitability and yield potential of food crop production on different land cover types at multiple scales (e.g., field-scale, regional); and (ii) illustrate this approach with two case study systems in the Northeast USA:

management-intensive grazing (MiG) grass-fed beef and confinement dairy beef. Additionally, the system boundary for dairy beef was expanded to include milk production to estimate results for the whole dairy and beef system.

2. Methods

The land use ratio (LUR) is estimated using Equation 1:

$$LUR = \frac{\sum_{i=1}^n \sum_{j=1}^m (LO_{ij} \times HDP_j)}{HDP_a}$$

where LO_{ij} is the whole herd land requirement for the production of feed ingredient i ($i=1,n$) in country j ($j=1,m$), resulting in the production of one kg of animal source food (ASF), and HDP_j is the maximum amount of human-digestible protein that could be produced per year from conversion of suitable land to human food crop production in country j (van Zanten et al., 2015). This sum is divided by the HDP_a , the amount of human-digestible protein from one kg of ASF produced by the system. We enhanced the LUR by including multiple sub-country scales of production (i.e., field and region scale) and estimating food crop production potential on different land cover types at those scales. The enhanced LUR is estimated using Equation 2:

$$LUR = \frac{\sum_{i=1}^n \sum_{j=1}^m \sum_{k=1}^p (LO_{ijk} \times HDP_{jk})}{HDP_a}$$

where all variables and indices are as in Eq. 1, except k ($k=1,p$), which indicates the livestock feed land requirement at the sub-country scale for country j . As in Van Zanten et al. (2015), the enhanced LUR is computed in four major steps, which are described in the following sections.

2.1.1. Quantifying Land Requirements of Feed Production

We used data from a recent life cycle assessment of Northeast grass-fed beef (GF) and dairy beef (DB) as the basis of the production systems (Tichenor et al., in review). We defined the Northeast region in accordance with the U.S. Department of Agriculture (USDA-NIFA, 2012). The GF system is a 30 cow herd that produces approximately 24 market-weight steers and heifers per breeding cycle (Table 1). Producers practice MiG, moving cattle between paddocks 0.4 – 6 times per day. During the grazing season, herd feed requirements are met with grass-legume pasture, milk from the dams, and a mineral mixture. During the winter, cattle are fed grass hay or grass-legume bale silage and a mineral mixture.

Table 1. Technical data for Northeast grass-fed beef system (Tichenor et al., in review)

Technical parameter	Value	Technical parameter	Value
Breeding stock		Birth to market weight (days)	
Number of mature cows	30	Steers	712
Number of mature bulls	1	Heifers	682
Number of replacement cattle	7	Market weights (kg)	
Calving percentage (%)	97.7	Steers	544
Cull rates (%)		Heifers	478
Brooding cows	9.7	Beef production (kg liveweight/year)	
Breeding bulls	20.0	Market steers and heifers	12,463
Mortality rates (%)		Culled cattle	1,822
Mature cows and bulls	1.5	Feed intake parameters (kg DM)	
Replacement heifers and bulls	2.1	Grass-legume pasture	142,887
Pre-weaned calves	3.6	Grass hay	103,781
Growing and finishing cattle	0.5	Grass-legume bale silage	15,874
Length of grazing season (days)	200	Mineral mix*	1,440

* No associated agricultural land use.

The DB system is a combination of two production systems: dairy production and finishing of dairy beef calves (Tables 2 and 3). The dairy system is a 328 cow herd, which produces milk, culled cows and bulls, and surplus calves to be raised for either veal or dairy beef. Cattle are fed a mixture of

harvested forages, concentrates, and a mineral mixture. Allocation between milk and beef was performed using a biophysical allocation equation developed for the USA (Thoma et al., 2013), as described in (Tichenor et al., in review). Newborn calves destined for dairy beef are first sent to starter operations to be weaned and then shipped to grower/finisher operations to be raised to market weight on a high concentrate ration (Table 3).

Table 2. Technical data for Northeast dairy system, system boundary at the dairy gate (Tichenor et al., in review)

Technical parameter	Value	Technical parameter	Value
Breeding stock			
Number of cows	328	Calving percentage (%)	86
Number of mature bulls	1	Beef production (kg liveweight/year)	
Number of replacement cattle	232	Calves (veal and dairy beef)	6,742
Cull rates (%)			
Breeding cows	28.2	Culled cattle	60,398
Breeding bulls	20.0	Feed intake parameters (kg DM)	
Mortality rates (%)			
Mature cows and bulls	6.1	Maize silage	1,328,928
Replacement heifers and bulls	1.5	Grass-legume hay	842,964
Pre-weaned replacement calves	7.0	Alfalfa haylage	596,284
Milk production			
Milk production per cow (kg/year)	10,732	Shelled maize	382,969
Total milk production per farm (kg/year)	3,522,404	Soybean meal	212,468
		DDGS, dry	53,117
		Milk replacer	4,261
		Mineral mix*	123,283

*No associated agricultural land use.

Table 3. Technical data for Northeast dairy beef production system, system boundary at the growing/finishing gate (Tichenor et al., in review)

Technical parameter	Value	Technical parameter	Value
Calves from dairy system	86	Birth to market weight (days)	485
Biophysical allocation beef (%)	9.4	Feed intake parameters (kg DM)	
Mortality rates (%)			
Pre-weaned calves	9.4	Shelled maize	122,134
Growing and finishing cattle	1.1	Maize silage	51,457
Market weights (kg)			
Steers	602	DDGS, dry	26,905
Heifers	529	Mostly-mixed grass hay	16,278
Beef production (kg liveweight/year)		Soybean meal	4,240
Market steers and heifers	44,523	Milk replacer	2,935
Culled cattle	60,398	Mineral mix*	2,705

*No associated agricultural land use.

Using whole herd feed requirements and crop yields (Table 4), we calculated the feed land required for the GF and DB systems.

Table 4. Feed crop yields at region and country scales 2000-2010

Crop and production scale	Feed crop yield (kg DM/ha)^a
Northeast region	
Grass-legume pasture	2,697
Grass-legume bale silage	4,641
Hay (mostly mixed grass; grass-legume)	3,166
Maize silage	12,166
Alfalfa haylage	5,966
Shelled maize	6,794
Country (USA)	
Shelled maize	8,204
Soybean meal	3,762
DDGS, dry	13,523

^aYields are adjusted for storage and feeding losses. Economic value allocation is used to further adjust yields for soybean meal and DDGS, dry. (Source: Tichenor et al., in review)

2.1.2. Estimating Suitability for Human Food Crop Production

We estimated the suitability of livestock feed land to produce the same five human food crops as Van Zanten et al. (2015), which included maize, soybeans, wheat, potatoes, and rice. To estimate suitability for food crop cultivation at the field and regional scales, we needed to spatially identify systems' feed land, classify it as arable or non-arable, and apply productivity indices to estimate food crop yields on arable feed land.

For the GF system, field-level data was collected from a sample of Northeast grass-fed beef producers practicing MiG (n=9). All forage land owned, leased, or managed for the maintenance of their herds was mapped using the Google My Maps application (Google, 2015). Farm parcels were then exported to ArcMap version 10.2 for spatial analyses (Esri, 2014). To assess arability, the Gridded Soil Survey Geographic (gSSURGO) Database was used, which includes a raster layer of the highest resolution classification of soils in the U.S. (Soil Survey Staff, 2014a). We used the non-irrigated Land Capability Class (LCC) attribute data within gSSURGO to classify each cell within the farm parcels as either arable (LCC 1 through 4) or non-arable (LCC 5 – 7) (Soil Survey Staff, 2014a; USDA-SCS, 1961). For forages purchased beyond the boundaries of the farms, a similar method to estimate arability of soils at the regional level was used. The 2014 Cropland Data Layer (CDL), a raster agricultural land cover data layer, was used to spatially classify land used for hay and pasture production (hereafter, hay/pasture) (USDA-NASS, 2014). The LCC dataset was again used at the regional level to classify hay/pasture land as arable or non-arable.

For the DB system, field-level data was not available. However, we assumed all forages and most maize (except the fraction used in calf feed, which is purchased from companies) in the ration were produced regionally. As with regional hay/pasture, the CDL was used to spatially identify regional cultivated cropland and the LCC to classify cropland as arable or non-arable.

Two of the coproduct feeds in the DB system, soybean meal and distillers dried grains with solubles (DDGS), were globally traded commodities. The U.S. is the top global producer and exporter of soybeans, with less than one percent of the domestic average annual supply (2000-2010) of soybean and soybean meal imported (USDA-ERS, 2015a, 2012). As such, we assumed all soybean meal was produced domestically. Similarly, the U.S. is the top global producer of corn, which is used in ethanol production, the primary source of the co-product DDGS in the U.S. (USDA-ERS, 2015b, 2015c). Less than one percent of the total domestic average annual supply of DDGS (2000-2010) is imported (USDA-ERS, 2015c). Thus, we assume all DDGS were produced domestically. The country level suitability assessment from Van Zanten et al. (2015) was used for the land used to grow these feeds.

2.1.3. Estimating Human-digestible Protein Production from All Suitable Land

For each food crop, the approach of Van Zanten et al. (2015) was followed to estimate human digestible protein output. Results were also calculated on an energy basis for an additional perspective. To estimate food crop yields, we used a tiered approach at the field, regional and national scales. At the field scale (GF system only), the National Commodity Crop Productivity Index (NCCPI) version 2.0 within gSSURGO was used to estimate maize, soybean, and wheat yields on arable land used for forage production (Soil Survey Staff, 2014b). The NCCPI corn and soybeans (hereafter, corn/soy) module and small grains modules estimate non-irrigated yields as a function of soil, climate and landscape factors (USDA-NRCS, 2012). Weighted average NCCPI corn/soy and small grains values on arable pasture and arable hay/bale silage land were calculated within ArcMap. Those values were then applied to adjust the following maximum yields: 15,063 kg ha⁻¹ maize, 4,705 kg ha⁻¹ soybeans, and 8,065 kg ha⁻¹ wheat (Dobos, Personal communication). For potato, we developed an alternative approach, as no direct productivity index was available (see below). We assume no suitability for rice at the field or regional scale, because it is a sub-tropical or tropical crop and thus could not be grown under normal circumstances in the temperate Northeast (Samanta et al., 2011).

At the regional scale, weighted average NCCPI corn/soybean and small grains values on arable hay/pasture land and cultivated cropland were calculated. For potatoes, a similar index was not available to apply at the field and region scales. Instead, region average yields (2000-2010), weighted by land area from the data of Griffin et al. (2014) were used as a proxy for yields on all cultivated cropland. We assumed potato yields on regional hay/pasture land were 15.4 percent lower than on cultivated cropland, based on the proportional difference between NCCPI values on arable cultivated cropland and hay/pasture land. At the country level, U.S. average yields (2000-2010) were applied for all five food crops (USDA-NASS, 2011) (Table 5).

2.1.4. Estimating Human Digestible Protein of Animal Source Food

A conversion factor of 0.35 edible weight beef per liveweight cattle was applied, accounting for decreased carcass conversion of grass-fed, Holstein breed, and culled cattle (Duckett et al., 2013; Neel et al., 2007; Scaglia et al., 2012; Stackhouse-Lawson et al., 2012). Conversions from raw ASF output to human digestible protein and energy were from Van Zanten et al. (2015), with the exception of milk protein (252.1 g kg DM⁻¹) and energy content (21.8 MJ kg DM⁻¹) (Thoma et al., 2010; USDA-ARS, 2015)

2.2. Scenario Analyses

Cropping is possible on arable forage land but not necessarily advisable from a conservation or potential profitability perspective. Clearing and cropping woodland pasture, for example, may have a high economic opportunity cost, carbon emission, and biodiversity impact. Additionally, economic viability of cropping on pasture may be limited by low yields. Fully addressing these tradeoffs is complex and beyond the scope of this analysis. However, to explore how such considerations may impact the results, we developed “risk averse” (RA) scenarios for hay/pasture land used in the production systems. For the GF system, we first reclassified all arable land currently managed as woodland pasture on farms as non-arable (RA1). For both systems, we used NCCPI statistics on cultivated cropland at the region-scale to define thresholds to limit the conversion of all hay/pasture land cover based on potential productivity. These thresholds were the mean NCCPI corn/soy and small grains values on regional cultivated cropland minus 2, 1, and 0.5 standard deviations (RA2 – 4), which were generated using the Zonal Statistics as Table tool in ArcMap (Esri, 2014). All other steps in the LUR calculation were the same for these scenarios.

3. Results

The majority of forage land at the field and region scales was arable (Table 5). Estimated yields on pasture and hay/bale silage land at the field scale were similar to the regional scale. For example, when field scale pasture and hay/bale silage land were combined, the weighted average NCCPI corn/soy was 0.39 (results not shown) compared to 0.41 at the region scale.

Table 5. Suitability, productivity index values, and estimated yields at farm, region and country scales for feed land

Feed land and production scale	Percent arable	NCCPI corn/soy	NCCPI small grains	Food crop yield on arable land (kg/ha)				
				Maize	Soybeans	Wheat	White potatoes	Rice
Field scale								
Gross-legume pasture land	89.1	0.43	0.32	6,538	2,042	2,600	26,272	-
Hay and bale silage land	89.1	0.34	0.31	5,062	1,581	2,486	26,272	-
Regional scale								
Pasture and hay land ^a	83.4	0.42	0.36	6,333	1,978	2,878	26,272	-
Cultivated cropland ^b	100.0	0.50	0.41	7,552	2,359	3,280	31,049	-
Country scale	100.0	-	-	9,271	2,729	2,814	43,294	7,582

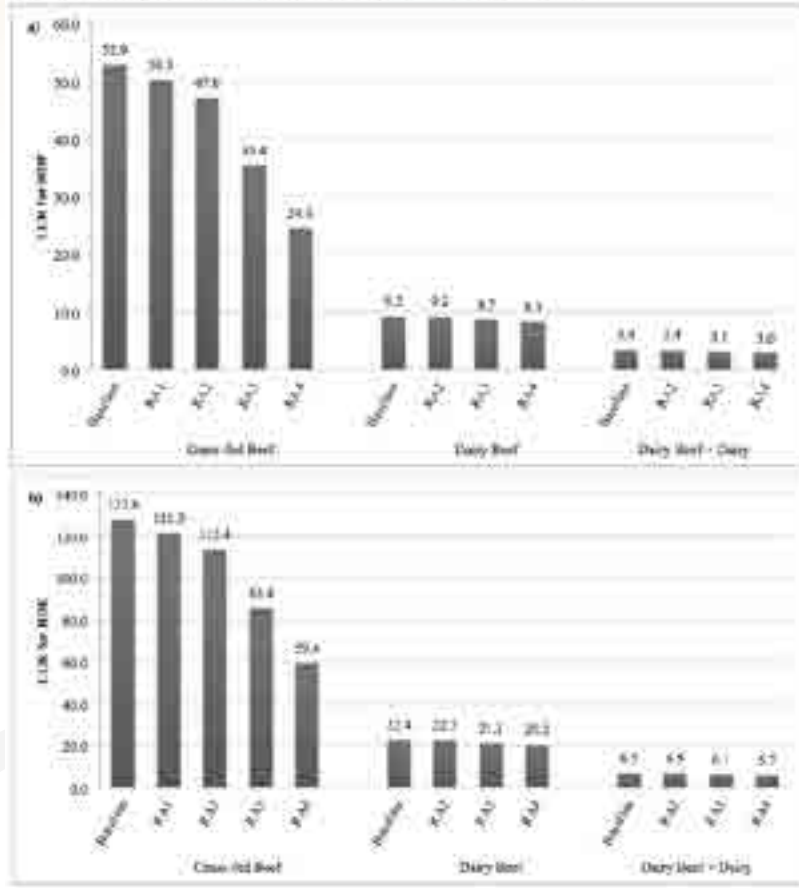
^a Pasture and hay land includes: gross-legume bale silage, gross-legume hay, mostly-mixed grass hay
^b Cultivated cropland includes: maize silage, alfalfa haylage, shelled maize

When compared to national averages, estimated food crop yields on regional cropland were lower for all crops except wheat. Unsurprisingly, estimated food crop yields on arable regional hay/pasture land were lower than on regional cultivated cropland. However, the difference was not that large, ranging from 12% lower for wheat to 16% lower for corn and soy.

The baseline LURs on human digestible protein (HDP) and energy (HDE) bases for both beef systems were much greater than 1 (Figure 1). For the GF system, the high arable fraction of forage land and moderately high estimated productivity resulted in a large LUR. DB land use was 86% lower per unit output compared to GF, accounting for part of the reason why the LUR was much lower despite its high reliance on cultivated cropland. The protein based LURs of GF and DB systems mean that converting their arable feed land bases to food crops could yield 52.9 and 9.2 times more human digestible protein than is currently produced. Overall, the protein based LUR of the GF system was about six and sixteen times greater than the DB and DB plus milk systems at baseline, respectively. Expanding the system boundary to include milk production from the dairy calving system resulted in a lower LUR compared to DB, though it was still greater than 1 (Figure 1). Although 55% of the land required for the dairy system was cultivated cropland, high milk productivity and HDP output partially compensated for this. Adding milk to the DB system resulted in a more than a 17-fold increase in HDP output with only seven times more land.

None of the RA scenarios resulted in a LUR less than one (Figure 1). For GF, removing woodland pasture from the arable land base at the farm scale (RA1) reduced the arable fraction to 0.83, resulting in a 5% reduction in the LUR (HDP and HDE bases). In each RA scenario, the GF system had the largest LUR, followed by DB, and then DB plus milk. However, the magnitude of the differences between GF and the dairy based systems decreased with each additional scenario. Limiting the fraction of forage land that could be converted based on productivity potential (RA 2 - 4) had a much more powerful result in the GF system, due to its complete reliance on forages. Major reductions in the LUR were only realized in the GF system in RA 3 and 4, where pasture and hay land conversion was limited to land with productivity greater than or equal to the mean cropland productivity, minus 1 or 0.5 standard deviations. This illustrates substantial overlap between distributions of potential productivity of pasture/hay land and cultivated cropland for these food crops at the regional level. Although the LUR of the GF system was 53% lower than baseline in RA4, the differences between the production systems remained large. The protein-based LUR of the GF system was approximately three and eight times larger than the DB and DB plus milk systems, respectively.

Figure 1a,b. Land use ratios (LUR) on a) human digestible protein (HDP) and b) energy (HDE) bases for Northeast beef systems. Baseline and risk events (RA) scenario results are shown.



4. Discussion

While the enhanced LUR is an important metric to understand land use efficiency, there are limitations to this method. Van Zanten et al. (2015) described some of the limitations related to the nutrient based functional unit. Our suitability and productivity estimates rely on national soil survey data that is continuously updated on a project basis, though the currency of full soil surveys at the county-level varies tremendously (Soil Survey Staff, 2014a; USDA-NRCS, 2016, n.d.). Although the NCCPI accounts for many soil attributes that could impair productivity, such as erosion class, actual erosion could vary from site to site due to the nature of the data. In many cases, it is uncertain how much erosion has occurred since it is estimated using general class ranges at a point in time and because the actual starting condition of the soil surface is uncertain (Pers. Communication, S. Finn, 5.19.16). That being said, this dataset is the highest resolution classification of soils in the U.S., and thus, was best possible option for this analysis.

The enhanced LUR method simulates potential human food crop productivity at multiple sub-country scales, on multiple land cover types. While the spatial datasets used were specific to the USA, agricultural land cover data and productivity indices are available in other countries, making this approach broadly applicable. For the dairy systems of Van Zanten et al. (2015), the only grassland used was on the farms, with suitability and potential food crop yield determined by broad soil type (sand versus peat). For the present cases, we sought to estimate suitability and yield potential of cropland and grassland at one or more sub-country scales. Using a productivity index made this possible and increased the specificity of food crop yield estimates. Given the latter, as well as differences found between national and estimated regional food crop yields on cropland and grassland, this approach increased the accuracy of estimating the LUR for ruminant systems in the USA.

We have demonstrated that even when beef production is completely reliant on forages, there may be a significant opportunity cost of land use regarding human food production. However, these findings are specific to these case studies in the Northeastern U.S. Raising grass-fed beef on arid rangeland in the Western U.S. likely has a starkly different LUR, which is an area for future research. Furthermore, as was mentioned earlier, although food production is technically possible on regional hay/pasture land does not mean it is economically feasible. We addressed one aspect of economic feasibility by limiting the converted land base based on productivity potential in the RA scenarios. However, farmers allocate land to uses they believe will result in the greatest benefit over time, estimating expected returns to land as a function of, for example, output value, input costs, current policies, land quality, skills, and personal preferences (Lubowski et al., 2006). In New York, Peters et al. (2012) estimated low and negative weighted average land use values for the production of grains and meat, respectively. The land use values, which were calculated assuming average quality land and conventional production systems, were also found to have high sensitivity to yield changes (Peters et al., 2012). Producing a high-value, niche product like grass-fed beef, therefore, may be an attempt to establish a business model that generates positive returns on less productive land in the region.

Significant opportunities may exist to reduce the LURs of these systems. Substituting food waste or crop byproducts for feeds is one potential strategy (Tichenor et al., in review), which may have policy momentum in the region due to an increasing number of organic waste landfill bans (Edwards et al., 2015). Future research should explore the net benefits of incorporating food waste and byproducts into these systems. In addition to process improvement, examining these systems with a more holistic lens may reduce their LURs, particularly for the GF system. Ecosystem services, such as carbon sequestration and cultural value provided by maintaining grasslands are not currently accounted for in the LUR. Partial accounting for this multi-functionality has produced starkly lower life cycle burdens compared to only considering marketed products in Spanish case studies (Ripoll-Bosch et al., 2013). Similar results could be true for the LUR, which merits further research.

5. Conclusion

The enhanced LUR provides a high-resolution, broadly applicable approach to estimating the opportunity costs of land used in livestock systems for human food production. The case study systems, MiG grass-fed and confinement dairy beef in the Northeastern U.S. have LURs greater than one, meaning they produce less protein than conversion of their suitable land bases to food cropping would. However, the LUR does not consider the ecosystem services provided by regional grasslands, which are likely important both from a conservation and social value standpoint. At the very least, the LURs provide additional clarity on the tradeoffs regarding different regional beef production systems in terms of the future food supply.

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49. Modelling Nitrogen and Phosphorus Footprints, Case Finnish Beef Chain

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ABSTRACT

Even though nutrient balances and emissions of nitrogen and phosphorus have been extensively monitored, particularly in farming, an indication of nutrient use efficiency through the entire food chain has been lacking. In this study, we defined N and P flows associated with Finnish beef production and consumption chain from a product-specific point of view using the Nutrient Footprint methodology and data from a Finnish beef LCA study. Nutrient footprint is an indicator, which combines the amount of nutrients captured for use in the production chain and share of nutrients utilized either in the primary product itself or in the entire production chain, accounting also for secondary products. 1000 kg of consumed Finnish beef from suckler cow-calf system require 1700 kg N and 188 kg P during its life cycle. The share of virgin nutrient is more than a half for N but only 24% for P. The nutrient use efficiency (NUEs) is for N 1% and 47% and for P 0.2% and 75% (in the primary product and in the entire chain, respectively). The nutrient footprint offers information about the nutrient usage and utilization efficiency in a simple and comparable form. In transition towards sustainable nutrient performance economy it is essential to define the hot spots of nutrient leakage in order to be able to close them and improve food chains.

Keywords: Nitrogen, Phosphorus, Food chain, Nutrient use efficiency, Life Cycle Assessment

1. Introduction

Global flows of nitrogen (N) and phosphorus (P) have increased over recent decades, due to increased request of mining of phosphorus apatite and conversion of nitrogen into its reactive form, for fertilizer use. Even though nutrient balances and emissions of nitrogen and phosphorus have been extensively monitored, particularly in farming, an indication of nutrient use efficiency through the entire food chain has been lacking.

In this study, our aim was to develop further the basic nutrient footprint methodology introduced recently by Grönman et al. (2016) by applying it to an animal food product, beef. The methodology was originally tested on oat flakes and porridge. Beef was chosen because it has been shown in previous Life Cycle Assessment studies to have relatively big environmental impacts related to plant and other animal products (Reijnders & Soret 2003, Williams et al. 2006, Carlsson-Kanyama & Gonzalez 2009, Audsley & Wilkinson 2012, Leip et al. 2014). Most LCA studies compare different kinds of production systems and, therefore, they stop at the farm gate. Some exceptions for beef exist (Mieleitner et al. 2012, Carlsson-Kanyama & Gonzalez 2009, Opio et al. 2013, Rivera et al. 2014), but to the best of our knowledge no studies on animal products from cradle-to-grave, until waste management exist.

The nutrient footprint methodology presented by Grönman et al. (2016) combines the amount of captured nutrients [kg of N and P] for use in the production chain and the share of nutrients utilized [%] either in the primary product itself or in the entire production chain, accounting also for secondary products. The captured nutrients are further divided to virgin and recycled nutrients. All phases of the production chain are included from fertilizer production to human food product and further to wastewater treatment. The method offers information about the nutrient usage and utilization efficiency in a simple and comparable form. Thus the nutrient footprint complements typical LCA studies on global warming, eutrophication and acidification potential.

A different concept of the nutrient footprint has previously been presented by Leach et al. (2012). They have developed a nitrogen footprint tool which calculates the nitrogen losses to the environment caused by food consumption per capita per year. For each food category they defined a Virtual N factor representing the total N loss in the production chain divided by the N that remains in the consumed product. Also Leip et al. (2014) have calculated nitrogen footprints of food products as direct N losses to the environment per unit of product. However, in this study production chain phases beyond slaughtering were not included. Leip et al. (2014) also present a Nitrogen investment factor, representing total external N required to produce one unit of product in terms of N contained. However, these approaches include only N and do not consider nutrient recycling. Our approach on

the other hand gives a more holistic view of the nutrient circulation in the food chain by combining nutrient use and emission data including all phases of the production chain until the treatment of human wastewater.

2. Methods

2.1. Nutrient footprint methodology

The Nutrient Footprint methodology has been presented in detail by Grönman et al. (2016). In short, the methodology takes into account 1) the amount of nutrients [kg of N and P] taken into use, 2) whether nutrients are virgin or recycled nutrients and 3) the efficiency of these nutrients [%] utilized in the particular production chain. Nutrient losses at each life cycle phase are identified. The nutrients bound to the primary product and the secondary products are calculated separately.

Virgin nutrients are extracted from nature and converted into a reactive form for the studied production chain (typically inorganic fertilizers), while recycled nutrients have already been captured into a previous production process and then recycled to the studied production chain (typically manure, sewage sludge and food processing industry side flows).

2.2. System boundary and functional unit

The system boundaries of the case calculation of beef production system are represented in figure 1. Nutrient use efficiency of the further processing of secondary products or waste materials is not taken into account. Therefore, nutrients bound to secondary products of which the nutrient content is utilized as a food product, fertilizer or animal feed, are considered potentially utilizable nutrients. Also, the animal skin is used in leather production and thus its nutrient content is considered as potentially utilizable.

The functional unit for the case was 1000 kg of Finnish beef from suckler cow-calf system eaten by the consumer. This is slightly different from the previous study by Grönman et al. (2016), where the functional unit was 1000 kg of oat flakes reaching the consumer. Furthermore, Grönman et al. (2016) include food waste treatment into the system boundary, but exclude energy consumption in food preparation. In the present study we exclude food waste treatment, because we consider it as a side flow not belonging to the main product chain. Due to the functional unit setting, we include energy consumption during food preparation.

2.3. Data acquisition

The data on the production of inorganic fertilizers was derived from the manufacturers: for nitrogen from Yara (2012) and for phosphorus from Prud'Homme (2010). Data on feed crops and animal production is based on a Finnish national LCA project on beef (FootprintBeef). In that project a model integrating biological plant and animal models and environmental life cycle assessment approach was developed. Nutrient flows through the whole beef production system are modelled using dynamic biological functions. The model connects animal growth, feeding intensity and composition, feed production and manure and fertilizer use on different soil types. The model used assumes that all feed crops are cultivated on farm, and all manure of the cattle is spread on its own feed crops. The average data for a male calf originating from Finnish suckler cow-calf system was used in the present calculation. A share of nutrient inputs and emissions of a suckler cow was allocated to the calves based on physical causality: The energy requirement for maintenance was allocated for lactation, pregnancy and growth according to their shares of their summed energy requirement, and thus each calf was assigned the energy requirement of one pregnancy, one lactation, and their equivalent share of maintenance of a suckler cow. The energy requirement of a suckler cow for growth was assigned completely to the suckler cow's meat, as well as energy requirement of one pregnancy and lactation (as a proxy for the emissions of suckler cow's dam during its pregnancy and lactation). This resulted in allocating 43 % of emissions to meat of a suckler cow and 57 % to calves. Estimate of animals died/put down on Finnish farms was obtained from Hartikainen et al. (2014).

According to the EC regulation No 999/2001 (EC 2001) these are defined as class 1 risk material of transmissible spongiform encephalopathies (TSEs) and thus must be eradicated.

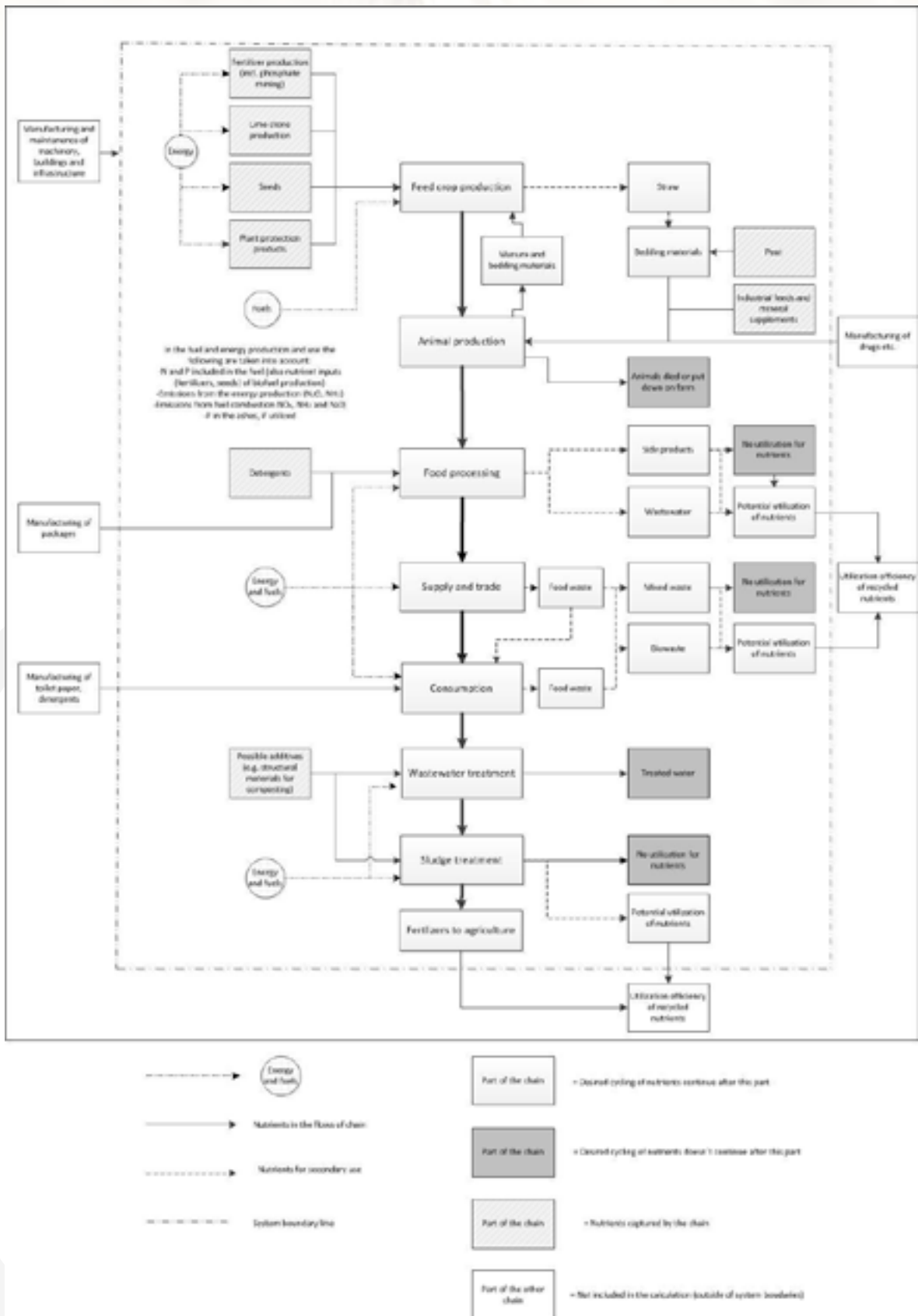


Figure 1. Simplified life cycle stages and system boundaries in the beef chain

After slaughter, meat and other organs are separated and either used for different food, feed and fertilizer products after processing or disposed as waste. We obtained data from a meat processing company and a recycling company of animal-based side flow and waste materials. Supplementary data from literature and nutritional databases was used for the share of different body parts and organs of the live weight as well as where they end up during the processing (Kauffman 2012, EC 2001, Aalto 2010, Huuskonen 2012). Data on the specific nutrient contents of bovine meat and organs were obtained from USDA nutritional database (2014), bone N content from Kauffman (2012), bone P content from Beighle et al. (1994) and blood and hooves N and P content from Fineli food composition database (2013).

Estimates for the energy use in storage in the retail chain were obtained from Taipale (2011). Estimate on food waste share in retail chains was obtained from (Eriksson et al. 2014). Estimates for the energy use in storage at the consumer as well as food preparation by the consumer were obtained from Taipale (2011). Weight loss of beef during food preparation was estimated 26% (Sääksjärvi & Reinivuo 2004). Food waste was assumed to occur after food preparation. With normal adults digestive system all eaten nitrogen and phosphorus is assumed to be excreted.

Household food waste originating from purchased beef and pig meat has been estimated to be 3.4% (Hartikainen et al. 2013). There is available limited information about the treatment of food waste in households. However, it can be estimated that 21% (23-29 million kg annually) of households food waste is separately collected as bio waste (HSY Helsinki Region Environmental Services Authority 2011, Silvennoinen et al 2012, Silvennoinen et al 2013, Statistics Finland 2012a). The remaining 79% (98-100 million kg) ends up in municipal mixed waste.

The calculation of nutrient flows in wastewater treatment was performed as described in Grönman et al. (2016).

3. Results

1000 kg of consumed beef require 1700 kg N and 188 kg P during its life cycle. The share of virgin nutrient is more than a half for N but only 24% for P. Inorganic fertilizers are the main source of virgin N and P, while recycled nutrients are derived mostly through the use of manure as fertilizer and cereal straw as bedding material. The nutrient use efficiencies (NUEs) are presented in figure 2.

The results show that there is potential in improving the NUEs in several phases of the production chain. The biggest nutrient losses occurred in the wastewater treatment phase. NUE (N) was also relatively low in the feed crop cultivation phase, indicating that N fertilization could be improved. Apart from wastewater treatment, NUE (P) was lowest in consumption and food processing phases. In the food consumption phase, P losses originated from the fuels used in Finnish electricity production (Alakangas 2000, GaBi 6 2012). In the food processing phase, P was lost in the body parts (skull, brain, spinal cord and vertebrae), that are defined as class 1 risk material of transmissible spongiform encephalopathies (TSEs) in the EC regulation No 999/2001 (EC 2001) and thus must be eradicated.

In the food processing phase, 22% of animal N and 53% of animal P ends up in secondary use: food, fur animal feed, pet food and fertilizer products. Also, 14% of animal N and 1% of animal P is in animal leather, which is utilized in leather industry. In leather products, the nutrients can stay bound for long time periods isolated from the nutrient circulation. However, they can later potentially be brought back to nutrient use and thus we did not consider them as lost nutrients.

4. Discussion

Compared to oat flakes (Grönman et al. 2016), beef NUEs are lower in the phases of crop production, food processing, supply and trade and consumption, as well as in the whole chain (Table 1.). The lower NUEs in crop cultivation phase are likely at least partly caused by the larger share of manure in the fertilization, because manure nutrients are more slowly soluble than mineral fertilizer nutrients.

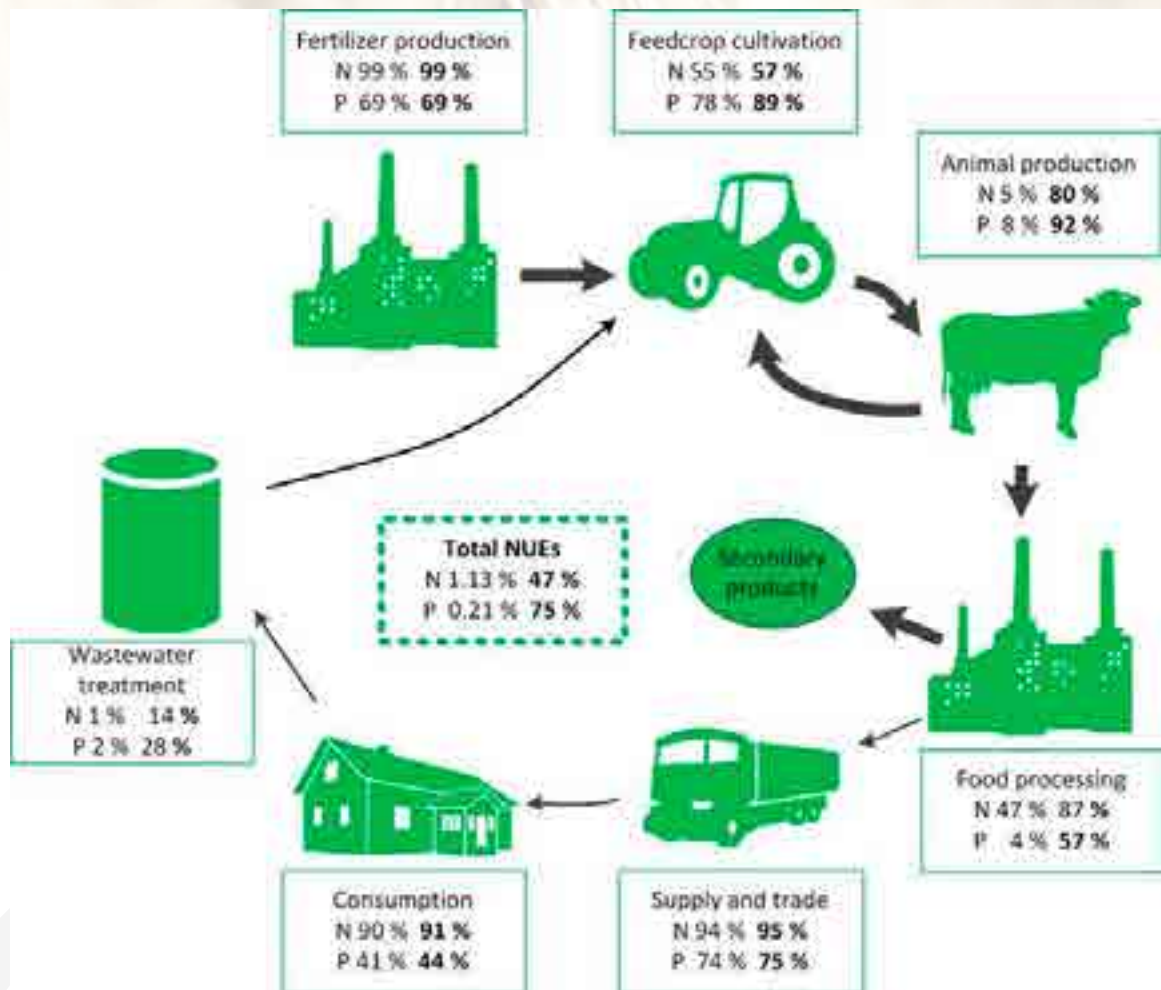


Figure 2. NUEs in the different phases of the beef production chain, considering nutrient use from beginning of the production chain to the primary edible product (in normal type) and accounting also for secondary products (in bold type). Total NUEs of the production chain (NUEs of the different phases summed together) are presented in the box with dashed outline in the center. Main nutrient flows are illustrated as black arrows. Arrow thickness represents relative volume of nutrient flow.

Table 1. NUEs of N and P of beef compared with oat flakes, %, utilization in the entire chain

Phase	Nitrogen		Phosphorus	
	Beef	Oat flakes	Beef	Oat flakes
Crop production	57	74	89	100
Food processing	87	92	57	94
Supply and trade	95	100	75	100
Consumption	91	95	44	95
Whole chain	47	71	75	99

Previously Leip et al. (2014) have calculated nitrogen footprint (direct N losses to the environment per kg carcass weight), and nitrogen investment factor (kg total external N required to produce one kg carcass weight in terms of N contained) of beef production system in EU 27, using a farm gate system boundary, including slaughtering. In their study the nitrogen footprint was ca. 500 g/kg product and nitrogen investment 15-20 kg N/ kg N in product. In the present study these values are relatively similar: 432 g/kg product and 35 kg N/ kg N in product.

According to Chatzimpiros & Barles (2013) NUE (N) in French feed crop cultivation on beef farms is 76%. They also calculated the overall NUE (N) of the livestock system as total N in retail products divided by total N inputs. Especially their NUE (N), 7,2%, is higher compared to the present study, where NUE (N) is only 1,2% when the same production stages are included.

Nguyen et al. (2010) have calculated N and P farm gate balances and efficiencies of typical beef production systems in EU, including a suckler cow-calf system resembling the system in the present study (Table 2). The reported N and P balances (calculated as nutrients in imported fertilizer and feed inputs minus nutrients in live animals sold) per 1000 kg animal slaughter weight are slightly greater than in the present study. However, also the nutrient efficiencies are greater.

Table 2. N and P inputs, outputs, balances and efficiencies in the present study and typical suckler cow-calf system in EU presented as kg N, P/1000 kg slaughter weight.

	The present study	Nguyen et al. 2010
Slaughter weight, kg	394	348
Age at slaughter, months	19	16
N balance, kg	401.9	437.7
N efficiency	0.05	0.09
P balance, kg	10.6	12.4
P efficiency	0.34	0.50

Leach et al. (2012), and Pierer et al. (2014) have calculated Virtual N factors (representing the total N loss (kg) in the production chain divided by the (kg) N that remains in the consumed product) for beef (Table 3.). When calculating in a similar way, the Virtual N factor for beef in the present study is higher. However, the results of the previous studies represent averages of national beef production systems, while in the present study only the suckler cow-calf system is studied. According to Nguyen et al. (2010) nutrient use efficiencies are greater in dairy bull-calf systems than in the suckler cow-calf system.

Table 3. Virtual N factors of beef

Reference	Country	VNF
Leach et al. 2012	US	8.5
Pierer et al. 2014	Austria	5.4
The present study	Finland	45.1*/ 12.9**

* Including inputs and emissions allocated from suckler cow

** Including only the slaughtered animal

5. Conclusions

The nutrient footprint methodology has potential in assessing the nutrient balances of food chains as well other bio-based production chains. It offers information about the nutrient usage and utilization efficiency in a simple and comparable form. In transition towards sustainable nutrient performance economy it is essential to define the hot spots of nutrient leakage in order to be able to close them and improve food chains.

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122. Carbon footprint analysis of pasture-based beef production systems based on rich primary data at the North Wyke Farm Platform in Devon, UK

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ABSTRACT

The objective of this study was to investigate the carbon footprint (CF) of three different pasture-based beef production systems at the North Wyke Farm Platform in Devon, UK. The three systems were based on permanent pasture, high sugar grass and white clover/high sugar grass mix. Detailed records were collected on farm for herd performance and IPCC calculations were used to estimate gaseous emissions. The white clover/high sugar system generated the lowest CF, predominately as a result of lower N fertiliser requirements in the presence of white clover. The permanent pasture system produced less greenhouse gas emissions than the high sugar grass system, due to better animal production efficiency. Our analysis to date suggests that these results are most sensitive to the application rate of inorganic fertilisers and the growth rate of cattle liveweight, which contribute to N₂O and CH₄ emissions, respectively.

Keywords: livestock; global warming potential; experimental farm; primary data.

1. Introduction

The worldwide population consumed 42.9 kg of meat per capita in 2013, whereas this figure rises to 75.9 kg per capita when the calculation is limited to people from developed countries (FAO, 2014). By 2050, the human population is expected to increase to 9.15 billion, and a 70% increase in total global food production is believed to be required subsequently (FAO, 2009). Amongst a wide range of food groups, livestock production generates 7.1 Gt CO₂-eq year⁻¹, of which beef and dairy cattle contribute 65% (Gerber et al., 2013). Given this forecasted increase in nutritional demand and the vast greenhouse gas (GHG) emissions associated with ruminants, identifying sustainable methods of beef production is critical to ensure long-term food security.

Feedlot-based beef production systems are believed to generate less global warming potential (GWP) when compared with extensive production systems due to increased animal productivity (Pelletier et al., 2010, Peters et al., 2010, Nguyen et al., 2010). However, these systems have also been shown to be the most inefficient users of human-edible cereals (Steinfeld, 2006), while cattle from pasture-based systems consume forages produced on land unsuitable for arable crop production (Eisler et al., 2014, de Vries et al., 2015) and largely indigestible for monogastric animals including humans (Wilkinson, 2011). Despite the beef sector accounting for 12% of total English agricultural output (EBLEX, 2013), there have been no national published studies to date on its carbon footprint (CF). Furthermore, most beef LCA research utilises national herd statistics, with few studies using detailed farm level data (Wiedemann et al., 2015).

Motivated by these criticisms levelled at the existing literature, this paper presents preliminary results from a primary data-driven CF analysis of pasture-based beef production systems at the Rothamsted Research North Wyke Farm Platform (NWFP) in the UK. The study is part of a wider research project aiming to identify optimal pasture-based beef systems based on newly designed sustainability metrics, which take economics, environment and human nutrition into consideration.

2. Methods

The NWFP is comprised of three hydrologically isolated “farmlets” with different pasture-based livestock systems (Figure 1). The “green” system is a permanent pasture (approximately 60% *Lolium perenne*) that has not been ploughed for > 15 years. The “red” system is high sugar perennial ryegrass (*Lolium perenne* cv. Abermagic), which was reseeded from 2013–2015 with a view to be improved again in 4-5 years. The “blue” system is a high sugar perennial ryegrass (*Lolium perenne* cv. Abermagic)/white clover (*Trifolium repens* cv. Aberherald) mix, also reseeded from 2013–2015. Conventional strategies for N fertiliser application are used for the green and red farmlets, while the

blue pasture receives a reduced amount of N fertiliser predominately in the form of farmyard manure. Each system has 30 Charolaise/Hereford-Friesian cattle per year which enter the farm post-weaning (Orr et al., 2016).

The system boundary for the present analysis was set from the production of fertilisers and supplementary feeds to the farm gate (Figure 2). The CF was considered for cattle kept on the NWFP. The cow/calf operation, which is maintained at a nearby but separate location outside of the NWFP, was not included in the model. Life cycle inventory data are provided in Table 1. Based on data obtained from frequent on-site weighing (usually fortnightly), pasture equivalent to 2.5% of the cattle's bodyweight was assumed to be consumed each day (Herring, 2014). Estimates on pasture yields were obtained from the NWFP field staff. Greenhouse gas losses from production of winter supplements were considered for soybean meals purchased; however, emissions attributable to the production of silage, which is believed to contribute a very small percentage of the overall CF, were excluded from the model. Foreground GHG emissions were calculated according to Tier 2 IPCC (2006) guidelines. The system was modelled in SimaPro 8.1.1. (PRé Consultants, 2016) with the functional unit of 1 kg liveweight gain (LWG).

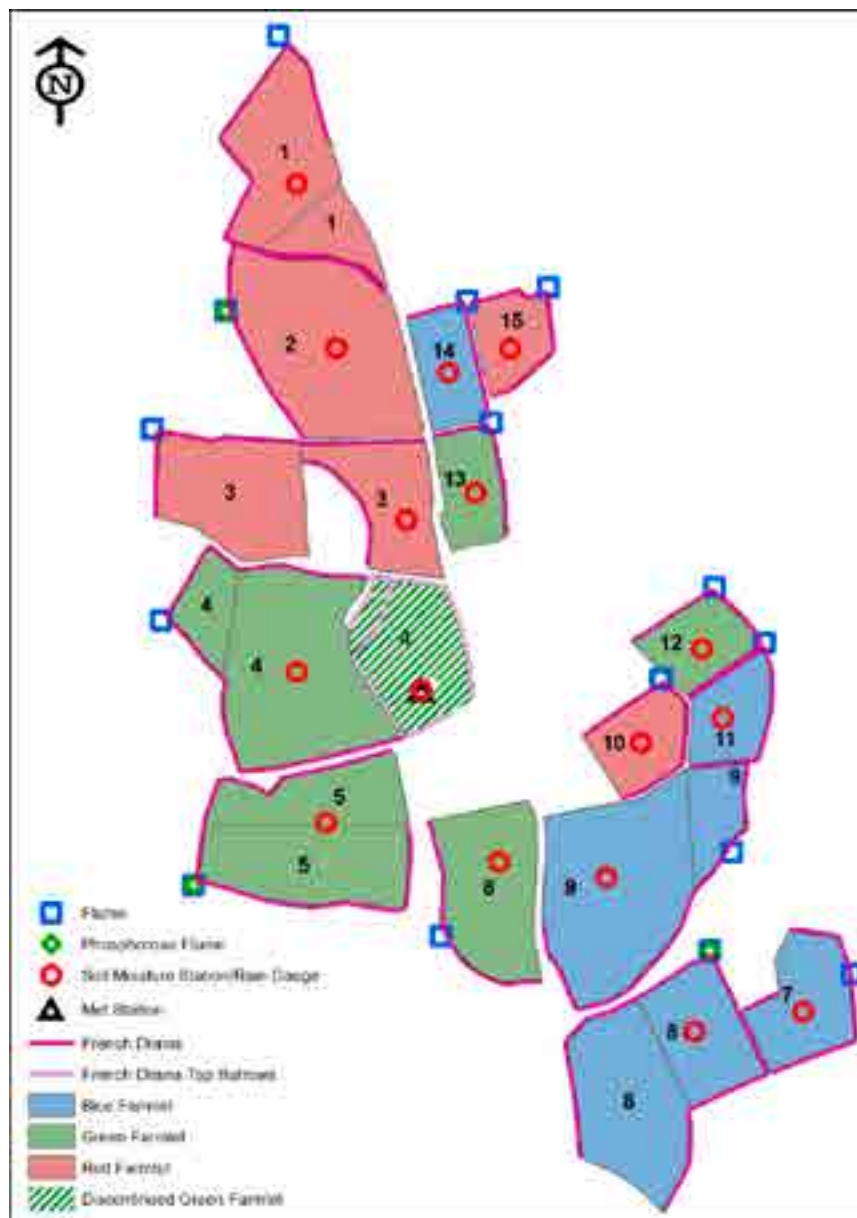


Figure 1: Map of the North Wyke Farm Platform, Devon, UK.

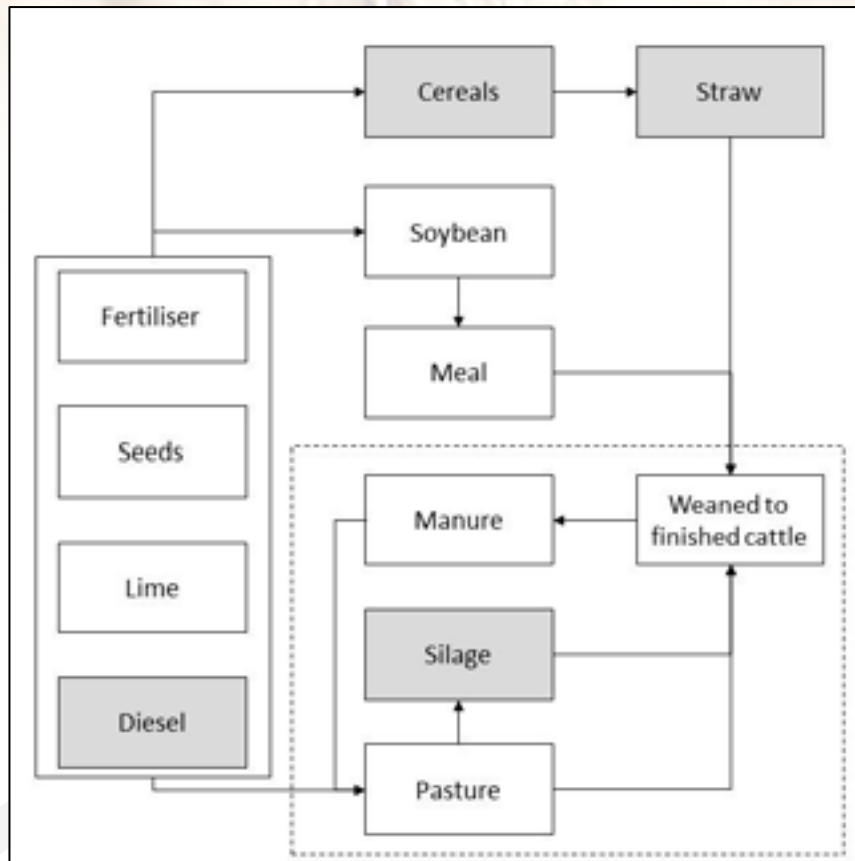


Figure 2: System boundary of the study. The dashed line represents the NWF boundary. Processes in grey are currently excluded but are to be incorporated in future studies.

Table 1: Overview of LCI data used in the study

Parameter	Unit	Green	Red	Blue
<i>Herd performance</i>				
Animal numbers	n	30	30	30
Average weight post weaning	kg	283	296	298
Average weight at sale	kg	605	599	597
Time on NWF	d	451	479	466
<i>Feed intake</i>				
Pasture	kg head ⁻¹	4520	4716	4608
Soybean meal	kg total	651	672	651
<i>Pasture</i>				
Area	ha	21.26	21.03	20.52
Yield	kg ha ⁻¹	11000	12500	10000
N fertiliser applied	kg	4135	3826	681
Lime applied	kg	0	4363	2804

3. Results

The primary results are presented in Table 2. In line with preceding studies on cattle production systems elsewhere, CH₄ emissions from enteric fermentation was the greatest contributor to GWP in all three systems. The red system generated the highest CF due largely to slower growth rates of animals. The “blue” farmlet generated the lowest CF, a result mainly attributable to the lower rate of fertiliser application. The supplementation of soybean meal had minimal impacts across the systems.

Although the research presented here is on-going, there are already some notable findings that seem to warrant further investigations. The blue treatment, which has a white clover mixed sward, generates less GWP than the red and green treatments by 17% and 14%, respectively. However, the green system, a permanent pasture representative of typical UK pasture-based beef production, finishes cattle at heavier weights in less time. The lower CF generated by “green” compared with “red” agrees with previous reports that faster LWG results in lower CH₄ emissions from enteric fermentation and N₂O emissions from manure management (Casey and Holden, 2006). From a farmer’s perspective, it could be concluded that the green system is most economically rewarding due to improved throughput, although the degree of the increased profitability depends on financial savings from lower fertiliser usage. On the other hand, the added cost of tillage and sowing also needs to be accounted for, both financially and environmentally. It is worth mentioning that the blue and red treatments were reseeded from 2013 to 2015, and have established better in some fields than others. White clover is known to require a year before nitrogen fixation becomes available to grasses (Andrae et al., 2016). Consequently, firmer conclusions should be drawn over the next two years when the blue sward in particular is expected to develop uniformly.

Table 2: Preliminary results from the CF study at NWFP. All values are presented as kg CO₂-eq kg LWG⁻¹. MM = manure management; ATD = atmospheric deposition.

Primary sources	Green	Red	Blue
<i>Emissions directly attributable to cattle</i>			
Enteric fermentation (CH ₄)	8.24	8.91	8.89
MM (CH ₄)	1.38	1.5	1.49
Direct MM (N ₂ O)	0.58	0.59	0.6
Indirect MM volatilisation (N ₂ O)	0.1	0.1	0.1
Indirect MM leaching (N ₂ O)	0.01	0.01	0.01
Soybean meal	0.048	0.05	0.048
<i>Emissions directly attributable to pasture</i>			
Direct soil manure (N ₂ O)	0.5	0.47	0.6
Direct soil fertiliser (N ₂ O)	1.29	1.13	0.22
Indirect soil leaching manure (N ₂ O)	0.09	0.09	0.11
Indirect soil leaching fertiliser (N ₂ O)	0.08	0.07	0.01
Indirect soil ATD manure (N ₂ O)	0.03	0.03	0.03
Indirect soil ATD fertiliser (N ₂ O)	0.02	0.02	0.004
Lime application (CO ₂)	N/A	0.11	0.09
Fertiliser N production	2.18	1.98	0.39
Seed production	N/A	0.003	0.0001
Total GWP	14.5	15.1	12.5

4. Discussion

de Vries et al. (2015) carried out a comprehensive review of beef LCA studies which examined various system boundaries, functional units and GWP results reported in previous research. From here, it is evident that the GWP values from the current study are largely incomparable with preceding studies due to the NWFP’s exclusion from the boundary of the cow/calf operation. This is so because NWFP is designed to enable rigorous comparisons between different farming systems in terms of nutrient cycling, and this objective can be better achieved by randomly allocating weaned calves to each farmlot (Orr et al., 2016). In other words, the CF obtained from this study is unconfounded with

the animal genetic effects passed over from cows to calves, although this benefit comes at the cost of inter-study comparability. Nevertheless, the average value of 14 kg CO₂-eq kg LWG⁻¹ falls within the range of previous work, even though the number would likely be higher if the cow-calf operation is also considered.

The results presented here are based on primary data in the form of detailed farm records and animal growth rates. All inputs to the NWFP are recorded, and a dedicated team of data analysts maintain and update selected datasets regularly (Harris et al., 2016). Silage grab samples are collected weekly during housing while snip samples from the three pastures are collected fortnightly during the grazing season. Ongoing chemical analysis is determining the modified acid detergent fibre (MADF) of forages, which will allow estimates of DMI based on metabolisable energy (ME). These estimates will, in the future, replace the current assumption of 2.5% DMI relative to LWG. Presently, IPCC calculations are used to estimate CH₄, N₂O and CO₂ emissions. However, the NWFP has automated systems located on each farmlet to constantly monitor N₂O and CO₂ emissions from soils, and eddy covariance towers are currently being installed to record lower atmospheric flows of N₂O and CH₄ originating from grazing cattle. These data combined with the aforementioned fibre analysis and environmental monitoring values will contribute to a highly detailed beef cattle LCA.

Utilisation of primary data has a number of novel potentialities with which to advance LCA methodologies. For example, detailed records of farm operations allow inventory modelling in a local context, reducing reliance on national level data and emission factors. Furthermore, individual data on crop and livestock performances allow the examination of heterogeneity both across livestock and field plots, enabling uncertainty analysis based on the true distribution of parameters rather than arbitrary assumptions. Finally, as experiments and measurements can be tailored flexibly to the purpose of each LCA study, scenarios set up for modelling analysis are not limited by data availability. In order to quantify these methodological benefits vis-à-vis the cost required for rich primary data collection, the results from the present study will be assessed against the new results once all the primary data have been incorporated into the modelling framework.

5. Conclusions

The three pasture-based beef production systems examined in this study resulted in different animal growth rates, and this difference was found to affect the CF. The “blue” legume-based system generated the lowest CF per kg LWG, a result that stemmed from lower N fertiliser application required. If animal growth efficiency can be improved on the “blue” treatment after the establishment of white clover, this system could be an economically *and* environmentally suitable method of beef production under the British soils and climate. However, economic and environmental costs of sowing and reseeding need to be accounted for before accurate conclusions can be drawn.

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250. A shrinking beef carbon footprint: Comparing greenhouse gas emissions of Canadian beef production between 1981 and 2011

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Recently, the environmental footprint of beef production has come under scrutiny and the public is looking for science-based information to provide an accurate assessment of the environmental impact of the industry. Total beef production in Canada has increased by 32% from 1981 to 2011 (Statistics Canada, 2015). Has environmental sustainability increased simultaneously with increased production? To determine the actual impact on sustainability indicators, greenhouse gas (GHG) emissions, breeding herd size, and land requirements of Canadian beef production were compared between 1981 and 2011.

Using a nation-wide industry level approach, temporal and regional differences in feed types and management systems, cattle categories, average daily gain and carcass weights were considered. Population data and land requirements were derived from Statistics Canada Census of Agriculture (Statistics Canada, 2014a, 2014b, 2014c). Estimates of marketed cattle were standardized on the basis of this data. Emissions were estimated using Holos, a whole-farm GHG emissions model (Little et al., 2013). The system boundary was at the farm gate and the functional unit was one kilogram (kg) of liveweight beef produced.

In 2011, beef production required 71% of the breeding herd, 74% of the slaughter cattle, and 76% of the land needed to produce the same amount of liveweight for slaughter as 1981. The estimated carbon footprint per kg of liveweight beef was 14.0 kg CO₂ equivalents (CO₂e) for 1982 and 12.0 kg CO₂e for 2011, a decline of 14%.

Enteric methane is the largest gas contributor (73% of emissions in both reference years) while the cow-calf component is the largest system contributor to enteric methane (approximately 80% of emissions in both reference years). The greatest opportunities for further mitigation lie within this component.

Improvements in livestock performance, including increases in average daily gain and slaughter weight, reduced time to slaughter, improved reproductive efficiency, and increased feed crop yields led to a reduction in the carbon footprint of Canadian beef over the three decades. Further studies will examine the impact of beef production on water use, air quality, biodiversity and provision of ecosystem services.

Keywords: beef cattle, environmental footprint, greenhouse gases

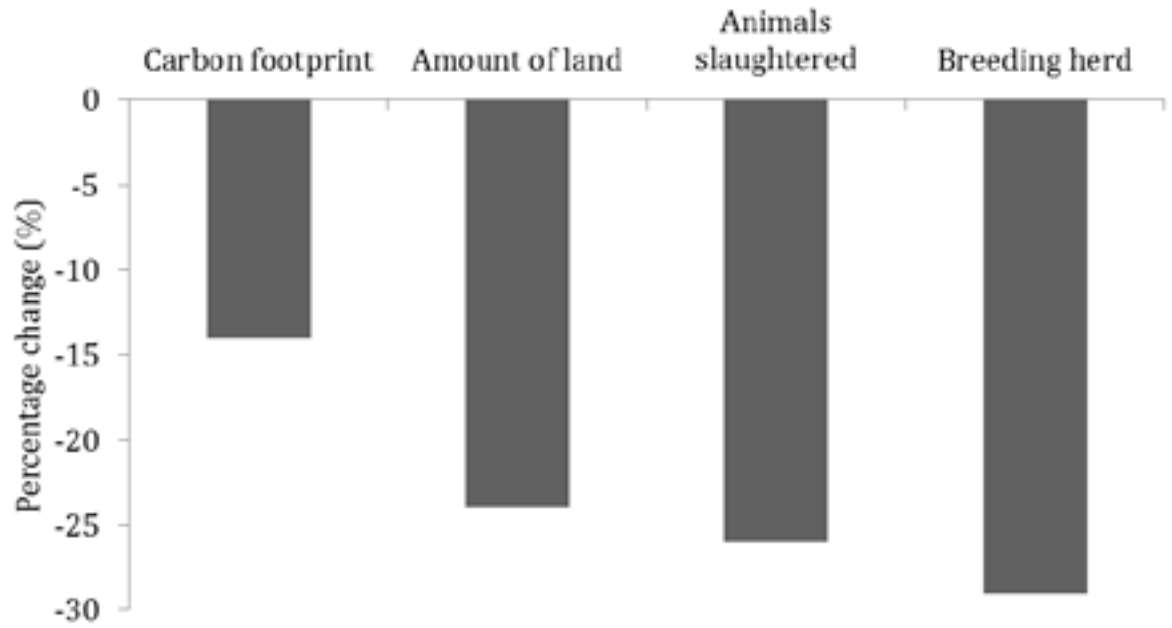


Figure 1. Percentage reduction in carbon footprint (CO₂e) and resource requirements to produce a given amount of Canadian beef in 2011 relative to 1981.

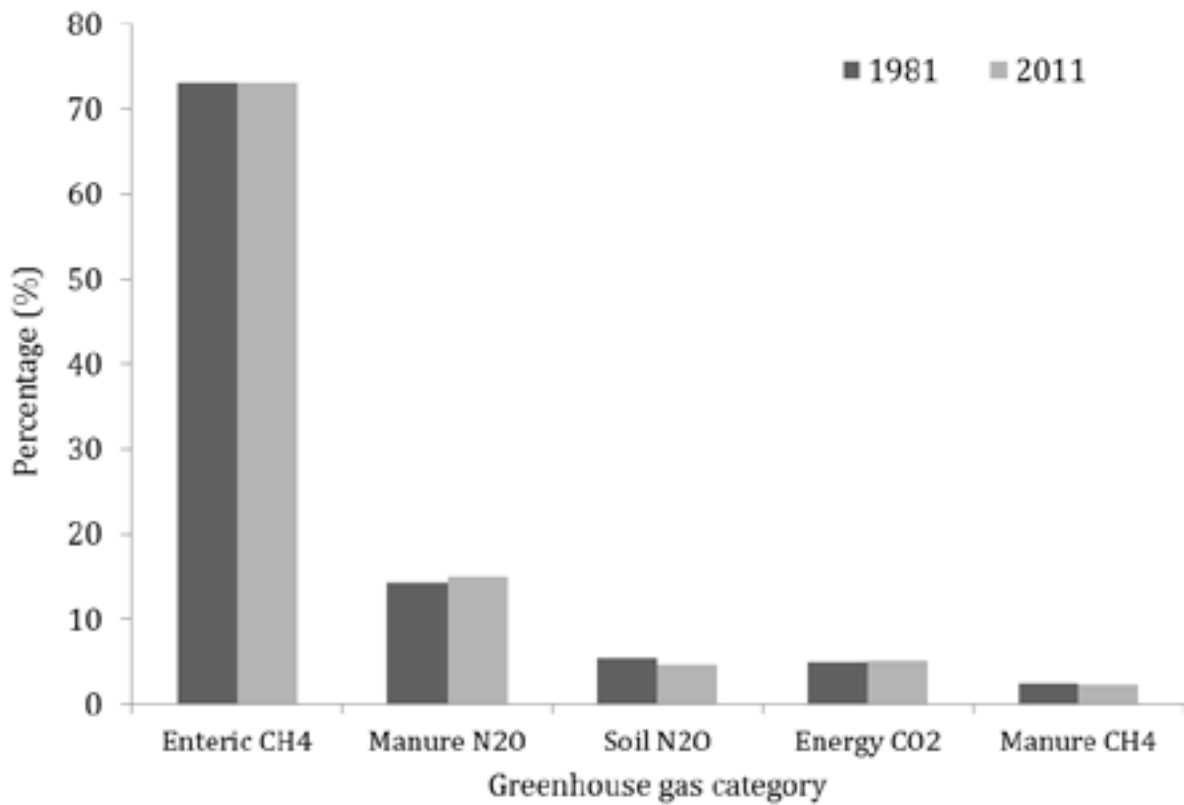


Figure 2. Contribution of different sources of GHG to the carbon footprint of Canadian beef in 1981 and 2011.

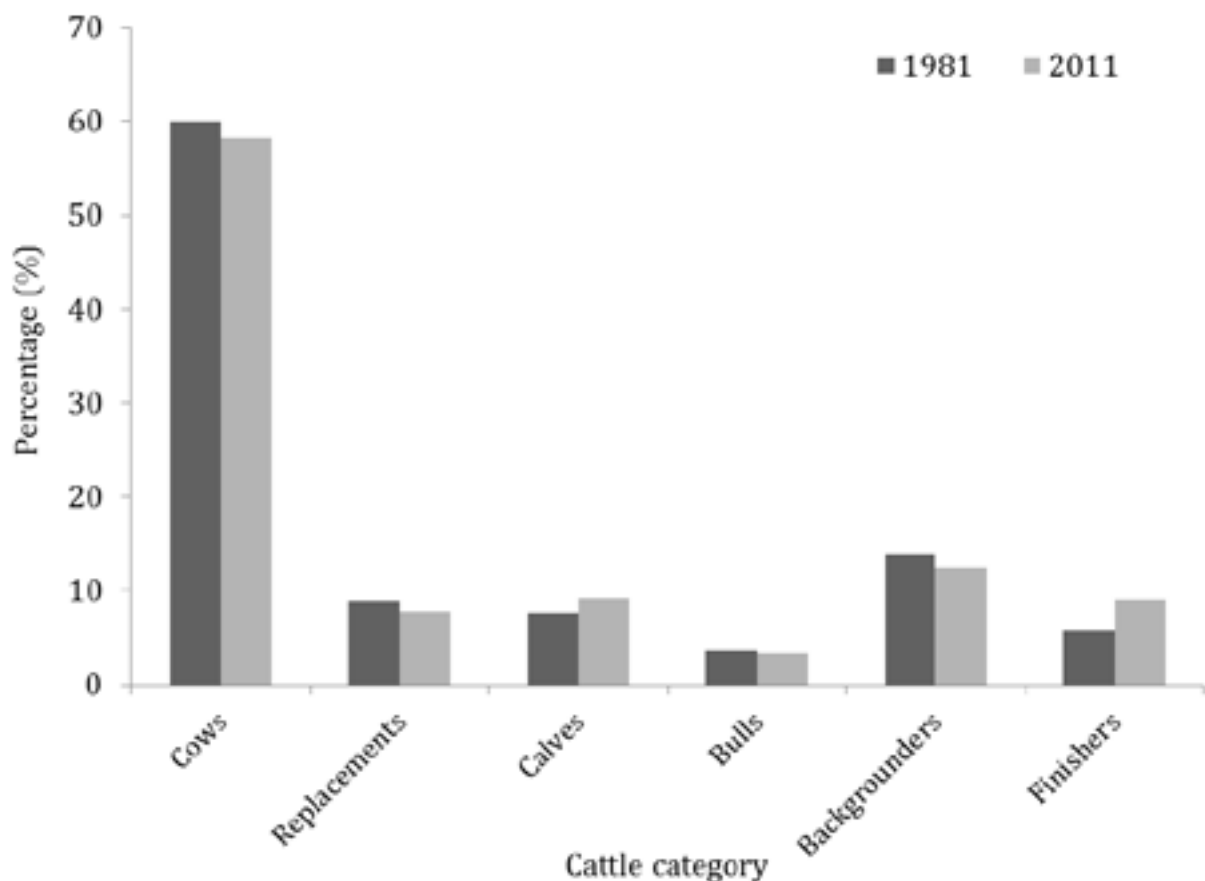


Figure 3. Breakdown of enteric methane emissions by cattle category in 1981 and 2011.

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309. Life cycle assessment of the environmental impact of extension supported commercial grass-based sheep farms

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ABSTRACT

Improving farm productivity is widely reported to mitigate the adverse environmental effects of livestock production and can potentially be achieved through greater adoption of agricultural advisory services. The goals of this work were to evaluate and compare several environmental impacts of 6 extension supported sheep farms and an economically averaged national farm using life cycle assessment (LCA). The mode of LCA used was descriptive and the system boundary was the cradle to farm-gate stage. The impact categories we assessed were acidification, freshwater eutrophication, marine eutrophication, climate change (carbon footprint), fossil fuel energy use and land occupation. The functional units related to environmental measures were kg of live weight sold and on-farm land area occupied. Generally, the results showed that the extension-supported farms had lower environmental impacts and resource use than the national average farm. The technical performance of the extension-supported farms generally increased over the 2-years of the study apart from P input use. This improvement in farm performance reduced most impacts, but the reduction in P-efficiency did increase freshwater eutrophication. Overall, our analysis indicates that sheep farms can mitigate environmental impacts by optimizing farm performance, but several farm efficiency measures need to be considered to avoid unacceptable declines in single aspects of farm performance such as the efficiency of P input use.

Keywords: carbon footprint, pollution swapping, farm optimization

1. Introduction

Consumers generally perceive sheep meat produced from grass as a healthy, natural and environmentally friendly product. Grassland sheep production has positive influences on the environment in terms of conserving landscapes and protecting biodiversity, but it also contributes to the substantial quantities of greenhouse gas (GHG) emissions and nutrients released by the livestock sector into our environment (Steinfeld et al. 2006; LEAP 2015). Several nations and regions have set targets to limit or reduce pollutants emitted by the livestock sector to prevent adverse environmental damage. However, globally meat production is projected to increase by 1-2% per annum until 2050 (Opio et al. 2013). The sheep sector, therefore increasingly needs to evaluate its environmental impact per unit of output or ecological footprint and develop strategies to improve it.

The effect of sheep farming systems on the environment are best assessed by evaluating emissions and resource use throughout the full life cycle of a product and across multiple indicators using a systems approach. Life cycle assessment (LCA) is typically the preferred method to use in this regard. In addition, the structure of the methodology is internationally standardized (ISO 2006 a, b) and specific guidelines have been developed for the livestock sectors, particularly the dairy sector (BSI 2011; IDF 2015; LEAP 2015). Researchers that have applied LCA to sheep farms, for example Ripoll-Bosch et al. (2013) and Wiedemann et al. (2015) usually assess on-farm and off-farm environmental impacts (e.g., mining of limestone) associated with livestock until the main product(s) is sold from the farm. This phase of LCA known as the cradle to farm-gate stage is a major source of pollutants from sheep production and key to mitigating environmental impacts.

Numerous strategies have been assessed to improve the environmental sustainability of livestock systems including modifying management practices, adopting new technologies or shifting to an alternative production system e.g. conventional to organic farming. Of the mitigation strategies assessed in this regard, optimizing farm performance is ubiquitously reported to reduce the ecological footprint of livestock production. The application of LCA to sheep farms however has largely been restricted to the evaluation of GHG emissions per unit of meat or carbon footprint. Consequently, it is not clear from this work how strategies to mitigate GHG emissions from primary sheep production affect other environmental impacts.

The objectives of our study were twofold. First, to quantify multiple environmental impacts of extension supported commercial Irish grass-based sheep farms and a national average farm using LCA. Second, to assess the influence of farm productivity on the environmental performance of sheep production by comparing the environmental impacts of commercial farms and a nationally averaged sheep farm. The aim of our latter goal was to determine if increasing farm productivity to mitigate one

environmental measure could have negative effects on other environmental indicators and thereby lead to “pollution swapping”.

2. Methods

2.1. Description of data collection and sheep production

The environmental impact of extension supported grass-based sheep farm was estimated using physical data from Irish farms that participated in the BETTER Sheep (BSP) farm program operated by Teagasc (2016). The sheep sector is a significant Irish employer and was worth €314 million to the national economy in 2014 (Teagasc 2015). The goal of the BSP program was to establish focal points for the on-farm implementation of technologies that will improve the sustainability of the sheep sector. The project provided 12 lowland sheep farmers spread across Ireland with specialist advisors that have access to the latest Teagasc research on grassland and sheep related technologies to increase farm productivity.

Seven of the 12 BSP farms participated in this study. Foreground information was obtained from these farms on a monthly basis in 2014 and 2015 using on-line or paper-based farm surveys. In addition, electricity and water use was metered on all farms. The surveys were typically completed in 1-2 hours and required farmers to provide data on land and crop areas, animal inventories, grazing dates, fertilizer and lime application, concentrate feeding, forage and manure imports/exports, manure storage and spreading and fuel purchases and water and electricity use. Sheep purchases and sales data were collected annually from farmers and meat processors. A trained auditor verified data throughout the study and carried out infrastructure surveys of the participating farms. Only 1 of the 7 farms surveyed did not provide sufficient data for the study.

The BSP group we evaluated aimed to maximize meat production from grazed grass. To achieve this objective sheep farmers bred ewes between late October and November and lambed ewes in spring (mid-March) to synchronize lamb growth with grass growth (Table 1). Generally, ewes were housed in the weeks coming up to lambing and fed conserved forages (grass silage, hay or both) and supplementary concentrate. After lambing, ewes and their progeny were turned out to pasture and supplemented with concentrate for a short period (< 2 weeks). Lambs were weaned at 12 weeks and sold for slaughter between 4-6 months at a live weight (LW) of approximately 45 kg.

The environmental performance of an average Irish lowland sheep farm was also estimated using the national farm survey (NFS) dataset of Hennessy et al. (2014). The NFS provides a representative sample of Irish farms and classifies farms as a sheep enterprise when at least 66% of the standardized gross margin of the farm comes from sheep meat and greasy wool. A wide spectrum of data are collected annually on 115 lowland sheep farms through the NFS, including financial, farm infrastructure data, animal husbandry data and production information. However, the husbandry dataset of the NFS was not sufficient to assess the average farm using LCA and was augmented with data from Teagasc (2015). The key farm inputs and outputs collected for the national average lowland farm and extension-supported BSP farms are summarized in Table 1.

Table 2 Technical summary of the mean inputs and outputs for the extension supported Better Sheep (BSP) farms and the Irish national lowland sheep system in 2014 and 2015.

Item	BSP 14	BSP 15	National 14
Ewes	361	351	107
Rams	7	7	2
Stocking rate (ewes/ha)	8.9	8.8	7.5
Lamb mortality (%)	10	8	12
Weaning rate (lambs/ewe)	1.56	1.55	1.30
Ewes culled (%)	22	22	25
Grass utilized (t dry matter/ha)	6.9	7.4	6.4
Concentrate (kg DM/ewe)	44	41	50
N fertilizer (kg/ha)	113	123	73.5
Electricity (kWh/livestock unit)	24	23	18
Farm fuel use (l/ha)	75	80	46
Sheep live weight (kg/ha)	702	746	419

2.2. LCA methodology

The LCA methodology was carried out according to the LEAP (2015) guidelines for small ruminants. The system boundary included foreground processes such as sheep rearing and background processes of transporting and manufacturing various farm imports e.g., pesticides and concentrate feedstuffs. Buildings and machinery were excluded from the analysis because of their small impact for grass-based livestock systems (Thomassen et al. 2008). Furthermore, some inputs, for example medicines were not included due to their lack of relevance.

The functional units we related environmental measures were kg of LW sold and on-farm land area occupied. Two functional units were used because agricultural systems have a local (e.g. eutrophication) and global effect (e.g. climate change) on the environment. The production of sheep yields more than one output (meat and wool). Additionally, concentrates that are fed to sheep can themselves be co-products of multifunctional systems (e.g., barley grain and straw). The products of multifunctional agricultural systems are difficult to separate. Therefore, allocation between products is usually required. Similar to several LCA studies of agricultural systems we allocated environmental impacts or resources to concentrate co-products and sheep outputs based on their market value.

The environmental impact categories and resource measures we chose to assess were acidification, freshwater eutrophication, marine eutrophication, climate change (carbon footprint), fossil fuel energy demand and land occupation. Other common categories included in LCA (e.g., ecotoxicity) were not taken into account, because detailed data on inputs such as pesticides was not available. The temporal coverage of our analysis was a period of 1 year and the mode of our LCA was attributional.

The resources used and emissions related to each process were quantified in the inventory analysis stage via a sheep farm model constructed in Microsoft Excel by Bohan et al. (2015). The quantity of resources used (e.g., lime) was directly measured or calculated for operations that farmers decided to contract to various service providers (e.g. silage contractors) based on the hours or area worked. These were estimated using the report of Teagasc (2011). For minor farm operations (e.g., hedge cutting) carried out by contractors, fuel use was estimated using data from Nemecek and Kägi (2007).

The Irish national GHG inventory (Duffy et al. 2014) was followed to estimate CH₄ emission from sheep. Enteric CH₄ loss for sheep was estimated as 6.5% of gross energy intake (GEI). For lambs, no enteric CH₄ emission were estimated for the first month post lambing, because milk was largely sufficient to sustain lamb growth. Post weaning enteric CH₄ emission was estimated as 6% of lambs GEI. The GEI and dry matter intake (DMI) of ruminants was estimated according to the net energy required for animal growth, milk production and maintenance (Jarrige 1989). The IPCC (2006) guidelines and the Irish GHG inventory were used to estimate CH₄ and CO₂ from manure and fertilizer. Based on Leip et al. (2010) we also tested the influence of including C removal or sequestration by grassland at a rate 0.89 t of CO₂/ha. Data from Nemecek and Kägi (2007) and the IPCC (2006) were used to quantify CO₂, SO₂, NH₃, NO_x and N₂O from fossil fuels.

Manure N emissions of NH₃, N₂O, NO_x and NO₃⁻ were quantified using a mass flow approach based on the annual quantity of N and total ammoniacal N (TAN) excreted. The TAN content of sheep manure was estimated as 60% of N excreted (Duffy et al. 2015). The total quantity of N and TAN in manure was partitioned between housing and grazing based on the length of the grazing season. Emissions of N were calculated for the different manure management stages (housing, storage, spreading) using emission factors from Duffy et al. (2014, 2015) and subtracted from the total pool to calculate the N or TAN available for the next stage. The same N emissions from the application of inorganic N fertilizer were estimated using data from Duffy et al. (2015).

Farm N and P balances were quantified as total N or P input in purchased feed, fertilizer, manure and livestock minus N or P exported in animals, feed and manure. The P lost to the environment was estimated as 0.5 kg P/ha when the P surplus/ha was between 1 and 5 kg, 1.5 kg P/ha when the surplus was between 5 and 10 kg, and 2.5 kg P/ha when the surplus/ha was > 10kg (Schulte et al. 2010).

Data for farms imports (e.g., electricity) were combined with emission factors from Howley et al. (2014) and the Carbon Trust (2013) to estimate off-farm CO₂, CH₄ and N₂O emissions. Soil CO₂ emissions from land use change were estimated for the production of some imported feedstuffs (e.g., South American soy). Land use change emissions were directly attributed to arable crops and estimated according to BSI (2011). For instance, average land use change emissions from South

American soy were estimated as 7.5 t CO₂/ha per annum (BSI, 2011). The Ecoinvent (2010) database was used to calculate NH₃, NO_x, NO₃⁻, SO₂ and P loss associated with the production and transport of farm imports. The land area occupied by imported feedstuffs was calculated using the feedprint database of Vellinga et al. (2013) and Ecoinvent (2010) was used for non-feed imports.

The resources used by sheep farms and emissions were grouped and converted into environmental impacts or measures using various characterization factors. P was considered the limiting nutrient for freshwater eutrophication and was calculated in P-equivalents (eq) using conversion factors P = 1 and PO₄³⁻ = 0.33 (Recipe 2008). N was considered the limiting nutrient for marine eutrophication and estimated in kg N-eq, N = 1, NH₃ = 0.092, NO₃⁻ = 0.23, NO_x = 0.039. Acidification potential was estimated in kg SO₂-eq, NH₃ = 2.45, NO_x = 0.56 and SO₂ = 1. The climate change impact of GHG emissions was calculated in terms of CO₂-eq using 100-year global warming potential (GWP) factors from the IPCC (2013). The GWP factors for key GHG emissions were 1 for CO₂, 28 for biogenic CH₄, 30 for fossil CH₄ and 265 for N₂O. Fossil fuel energy demand was quantified in MJ using the lower heating values of the cumulative energy demand method (Guinee et al. 2002).

3. Results

3.1. Environmental impacts and resource use

On average per kg of LW, the environmental impacts acidification, marine eutrophication and carbon footprint were between 10% and 44% lower for the BSP group in 2014 and 2015 than the national average farm in 2014 (Table 2). Similarly, the mean land occupation was between 24% and 30% lower for the BSP farms than the national average, but fossil fuel energy use was between 18% and 21% greater for the BSP farms than the national average and mean freshwater eutrophication was 3-4 times higher.

Table 3 Cradle to farm gate life cycle assessment results by impact category for the extension supported Better Sheep Farms (BSP) and the Irish national lowland sheep system in 2014 and 2015.

Impact category	Unit	BSP	BSP	BSP	BSP	National
		2014 Mean	2015 Mean	Min	Max	2014 Mean
Acidification	g SO ₂ -eq/kg LW	95.3	87.0	50.1	165.7	127.9
Marine eutrophication	g N-eq/kg LW	30.6	30.4	20.6	42.9	54.4
Fresh water eutrophication	g P-eq/kg LW	0.9	1.2	0.4	1.9	0.3
Carbon footprint	kg CO ₂ -eq/kg LW	8.4	8.2	5.8	9.6	10.4
Carbon footprint with seq ^a	kg CO ₂ -eq/kg LW	6.9	6.6	4.5	7.9	8.7
Fossil fuel energy use	MJ/kg LW	18.7	18.1	11.0	27.3	15.4
Land occupation	m ² /kg LW	15.2	14.0	9.8	21.8	19.9

^a Includes sequestration of carbon by grassland

There was a wide range in LCA results per kg of LW for the 6 BSP farms across the 2 years. The minimum and maximum environmental impacts of the 6 farms differed by a factor of 5 for freshwater eutrophication, a factor of over 3 for acidification and by between 64% and 76% for carbon footprint. Land and fossil fuel energy use varied by a factor of over 2 between the bottom and top farms.

Table 4 The on-farm local environmental impacts and nutrient use efficiency for the extension supported Better Sheep Farms (BSP) and the Irish national lowland sheep system in 2014 and 2015.

Impact category	Unit	BSP	BSP	National
		Mean 2014	Mean 2015	Mean 2014
Acidification	kg SO ₂ -eq/ha	57.5	57.7	127.9
Marine eutrophication	kg N-eq/ha	21.2	19.6	54.4
Fresh water eutrophication	kg P-eq/ha	0.8	0.3	0.3
On-farm N efficiency	%	16	18	15
On-farm P efficiency	%	55	47	98

The local impacts on-farm acidification and marine eutrophication were lower per ha for the BSP farms than the national average system and the mean N efficiency was greater for BSP farms. Mean fresh water eutrophication per ha was lower for the national average than the BSP group and mean P efficiency was greater for national average farm.

3.2. Environmental and resource profiles

Across the BSP and national sheep farms, the on-farm stage was the main source of acidification (89%). Figure 1 shows on-farm forage production (65%) and manure storage and housing (23%) were the main contributors to acidification. The key drivers of acidification for forage production were manure spreading (18%), grazing (50%) and fertilizer application (32%). N and P loss from on-farm forage production (71-92%) was the main contributor to marine and freshwater eutrophication for the BSP and national average farms. However, the contribution of imported feeds was also important for freshwater eutrophication (29%).

Enteric fermentation of feed by sheep was the main source of carbon footprint (50%) for the national average and BSP farms. The remainder of carbon footprint was largely generated from on-farm forage production (30%), and fertilizer and lime manufacture (10%).

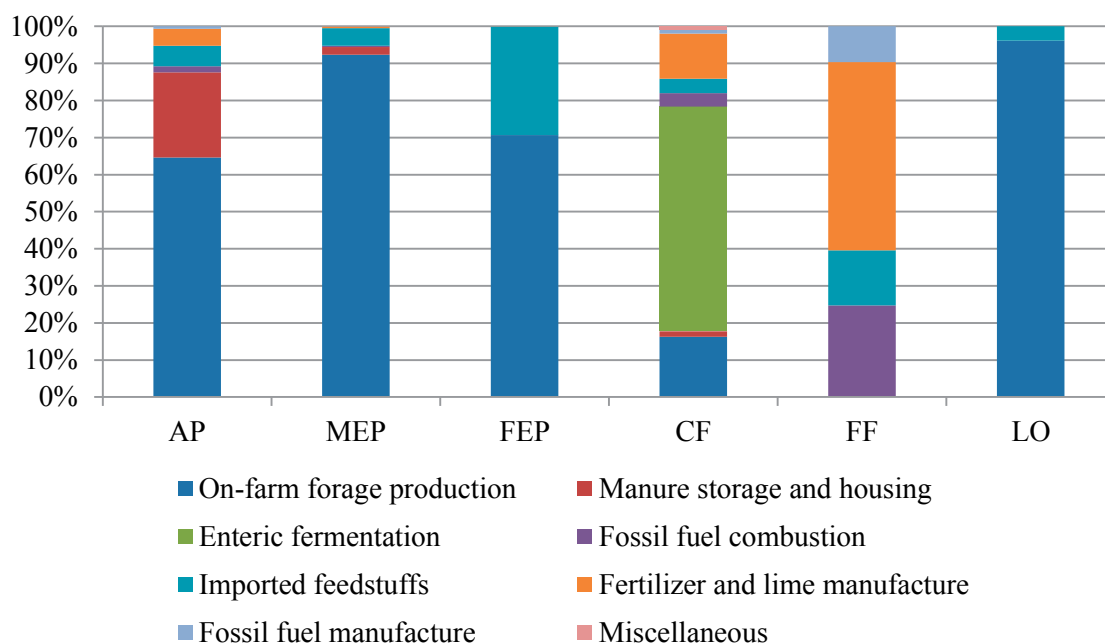


Figure 1 Mean contribution of inputs or sources to cradle-farm gate life cycle assessment impacts for the Better Sheep and national average farms. AP = Acidification, MEP = Marine eutrophication, FEP = Freshwater eutrophication, CF = Carbon footprint, FF = Fossil fuel use and LO = Land occupation.

The majority of fossil fuel energy was consumed off-farm (75%) for the national average and BSP farms. The largest consumer of fossil fuel energy was fertilizer manufacture (50%), followed by on-farm fossil fuel combustion (25%) and imported feedstuffs (15%). Land was mainly used to grow forage on-farm (96%) and the ingredients of concentrate feedstuffs off-farm (4%).

4. Discussion

The outcomes of our analysis were difficult to compare with other studies because most LCA studies of sheep farms only consider carbon footprint. In addition, LCA studies sometimes select different modeling methods and often use emission factors parameterized for specific regions (e.g., N leaching). Nevertheless, cautious comparisons between studies are useful to validate results. The outcomes of this study for carbon footprint were within the wide range of estimates for sheep LW (8-144 kg CO₂-eq/kg of LW; Ripoll-Bosch et al. 2013) and below the global average of 11.3 kg CO₂-

eq/kg of LW (Opio et al. 2013) for lowland farms. Our results for land use and fossil fuel energy demand, were generally higher than Williams et al. (2006) and Wiedemann et al. (2015) case study of Australian, UK and New Zealand sheep farms. Differences in land and energy use between studies were primarily explained by differences in crops yields and farm feeding practices. The UK results of Williams et al. (2006) for acidification and eutrophication were slightly higher than our results for these impacts largely because the UK sheep farms used more N and P inputs.

Similar to Opio et al. (2013) the key source of several global and local environmental impacts for the grass-based systems we assessed was forage production and it was a major consumer of resources. The BSP farms produced more forage than the economically average national farm and the efficiency of inputs used to produce forage by the BSP farm e.g., N fertilizer was generally greater than the national average farm. However, this was not always the case e.g., on-farm fuel consumption. The BSP group typically had lower environmental impacts than the national average farm, which implies similar to the conclusions of Gibbons et al. (2006) that primary producers can mitigate adverse environmental impacts by optimizing the technical efficiency of their farming enterprise.

Over the 2-years of the BSP study, members of the group adopted new tools to accurately measure grass supply and demand. These were the main changes that occurred on farm and led to increases in grass yield and quality, and greater LW production. Consequently, there was an improvement in several local and global environmental impacts including carbon footprint. However, there was a minor decrease in P-efficiency. The reduction in P-efficiency had little or no impact for most environmental impacts because P fertilizer was a minor contributor, but the reduction in P-efficiency was projected to increase freshwater eutrophication. This example of pollution swapping demonstrates that several measures of farm efficiency need to be considered simultaneously when aiming to improve environmental performance to avoid unacceptable declines in single farm aspects such as P efficiency. Currently, area-based thresholds for the local effects of farms on water and air are set in terms of total N and P inputs by the EU (European council 1991). For the sheep farms we assessed their total nutrient inputs were well below these area limits, which means that the local effect of farms on water and air quality measures was legislatively acceptable. This means there is scope to grow national sheep production, but the effects of nutrient surpluses on the environment should be regularly measured to verify pollution is not taking place.

The improvements we generally observed in environmental performance and resource use for the BSP farms support the hypothesis of Murphy et al. (2013) that environmental improvements can be achieved through greater adoption of agricultural advisory services. The service provides important advice to farmers regarding adopting better farm practices such as rotational grazing systems, online tools to monitor grass supply and demand, planning breeding of ewes, and cleaning and feeding systems to reduce lamb mortality. These changes have the largest potential to improve sheep farms environmental performance. Additionally, advisory services can be used to facilitate discussion groups for peer-to-peer learning. The latter is an important component of the success of extension, which Lapple et al. (2012) showed was a key contributing factor to the successful adoption of profitable new technologies by grass-based dairy farmers. There is however typically an economic cost to advisory support, which may be an impediment to adoption. The expense of this service should be self-financing, but agricultural supports could be used to initially reduce advisory costs to achieve greater participation. This would aid increasing sheep farms technical efficiency and improve farms environmental performance and resource use.

5. Conclusions

Our research shows that providing agricultural advisory support can aid sheep farmers to improve technical performance and thereby reduce multiple environmental impacts including carbon footprint. However, several farm performance measures need to be considered in tandem to avoid unacceptable declines in single aspects of farm performance such as the efficiency of P use, which can cause pollution swapping. This research therefore highlights the need for LCA studies of farming systems, particularly sheep, to consider multiple environmental and resource indicators rather than focusing on carbon footprint in isolation.

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ORAL PAPERS

FEED AND FERTILIZER

7. Feed And Fertilizer

229. Carbon footprint analysis of mineral fertilizer production in Europe and other world regions

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ABSTRACT

The production and use of mineral fertilizers contributes significantly to the carbon footprint of agricultural crops and crop-based food products. In arable crops such as winter wheat the share of nitrogen (N) fertilizer-related GHG emissions can be as high as 80%. The contribution of emissions from the production of mineral fertilizers is often as important as fertilizer-induced emissions from agricultural soils.

It is therefore important to use appropriate and up-to-date emission data for fertilizer production, which represent the actual technology and efficiency of manufacturing of specific fertilizer grades. Objective of this study is to provide up-to-date carbon footprint data for the main fertilizer products produced in selected world regions.

The association of the European fertilizer producers "Fertilizers Europe" has developed a carbon footprint calculator (CFC) for fertilizer production. This tool has been employed to derive reference values for the main mineral fertilizers produced in Europe and other relevant fertilizer-producing regions of the world. The European data are reported by all members in a regular survey to Fertilizers Europe and are representative for the year 2011. Reported data include energy consumption in ammonia synthesis (Haber-Bosch) and N₂O emissions from nitric acid production, as well as expert-validated data for other sources of CO₂ (e.g. energy consumption for urea synthesis and granulation). The non-European figures are based on an expert evaluation by Integer Research Ltd. of ammonia and nitric acid production in 2011, while for all other emission sources the European values were used.

The Fertilizers Europe CFC follows general LCA and carbon footprint rules. It covers all main sources of GHG emissions and has been reviewed by DNV GL to verify its completeness and correct calculations.

This paper explains the methodology applied in the calculation of the carbon footprint values. In addition the results will be presented per fertilizer product and production region. We suggest that these data should be used in carbon footprint studies as reference values for fertilizer production in different world regions with the technology baseline 2011.

Keywords: fertilizer production, product carbon footprint, GHG emissions

1. Introduction

Agriculture is responsible for 10 to 12% of the total global greenhouse gas (GHG) emissions (Smith et al., 2007) and the overall level of GHG emissions from agriculture is expected to grow further as agricultural production needs to expand in order to keep pace with increasing demand for food, feed, fiber and bioenergy. The production and use of mineral fertilizers is required to provide sufficient plant nutrients for sustainable food production. At the same time it also contributes significantly to the carbon footprint of agricultural crops and crop-based food products. The share of the global GHG emissions directly related to the production, distribution and use of fertilizers is estimated at between 2 and 3% (IFA, 2009). In arable crop production such as winter wheat the share of nitrogen (N) fertilizer-related GHG emissions can be as high as 80% (Brentrup et al., 2004; Skowrońska & Filipek, 2014). The contribution of emissions released during the production of mineral fertilizers is in most studies as important as the fertilizer-induced emissions from agricultural soils.

Information about production of mineral fertilizers used in major global life-cycle assessment (LCA) databases (e.g. Ecoinvent) is mostly outdated and relates to studies from 1990s (Patyk & Reinhardt, 1997; Kongshaug, 1998; Davis & Haglund, 2000). Since then production technologies have improved substantially mainly in terms of nitrous oxide (N₂O) emission control during nitric acid production, which is an intermediate product of nitrate-containing nitrogen fertilizers (EFMA, 2000a; Brentrup & Palliere, 2008). But also energy efficiency in particular in ammonia synthesis has improved over time (EFMA, 2000b; Jenssen & Kongshaug, 2003; Brentrup & Palliere, 2008).

It is therefore important to use appropriate and up-to-date emission factors for fertilizer production. The objective of this study is to provide up-to-date carbon footprint data for the main fertilizer products produced in important fertilizer-producing regions.

2. Methods

The association of the European fertilizer producers “Fertilizers Europe” has developed a carbon footprint calculator (CFC) for fertilizer production. The CFC is available for free use to anyone on simple request to Fertilizers Europe (www.fertilizerseurope.com). This tool has been employed to derive reference values for the main mineral fertilizers produced in Europe and other relevant fertilizer-producing regions of the world. The European data are based on primary data reported by all members in a regular survey to Fertilizers Europe. The data are representative for the year 2011. Reported data include energy consumption in ammonia synthesis (Haber-Bosch) and N₂O emissions from nitric acid production, as well as expert-validated data for other sources of CO₂ (e.g. energy consumption for urea synthesis and granulation of final products). The non-European figures are based on an expert evaluation by Integer Research Ltd. (2014) of ammonia and nitric acid production in 2011, while for all other emission sources the European default values were used. The reference data for Europe and other world regions will be regularly updated and published.

The CFC is a cradle-to-factory-gate calculator based on the principles developed by Kongshaug (1998). This means that the emission factors of the final products (expressed as kg CO₂-equivalent/kg fertilizer product) are calculated stepwise in building blocks that represent the actual steps in the production process. The building blocks include importation of raw materials, production of intermediates and the finishing process combining the materials into a final product (Fig. 1). All building blocks are characterized by emission factors and energy consumption values. The CFC takes into account all emissions with global warming potential (GWP), i.e. N₂O, CO₂ and CH₄. Using the GWP conversion factors (IPCC, 2007) N₂O and CH₄ emissions are converted to CO₂-equivalents (CO₂e). The CFC contains built-in default values for important fertilizer-producing world regions (EU, Russia, China and US) for the reference year 2011, but the user can also insert own individual values in order to calculate the carbon footprint of specific own-produced fertilizer products (Christensen et al., 2014).

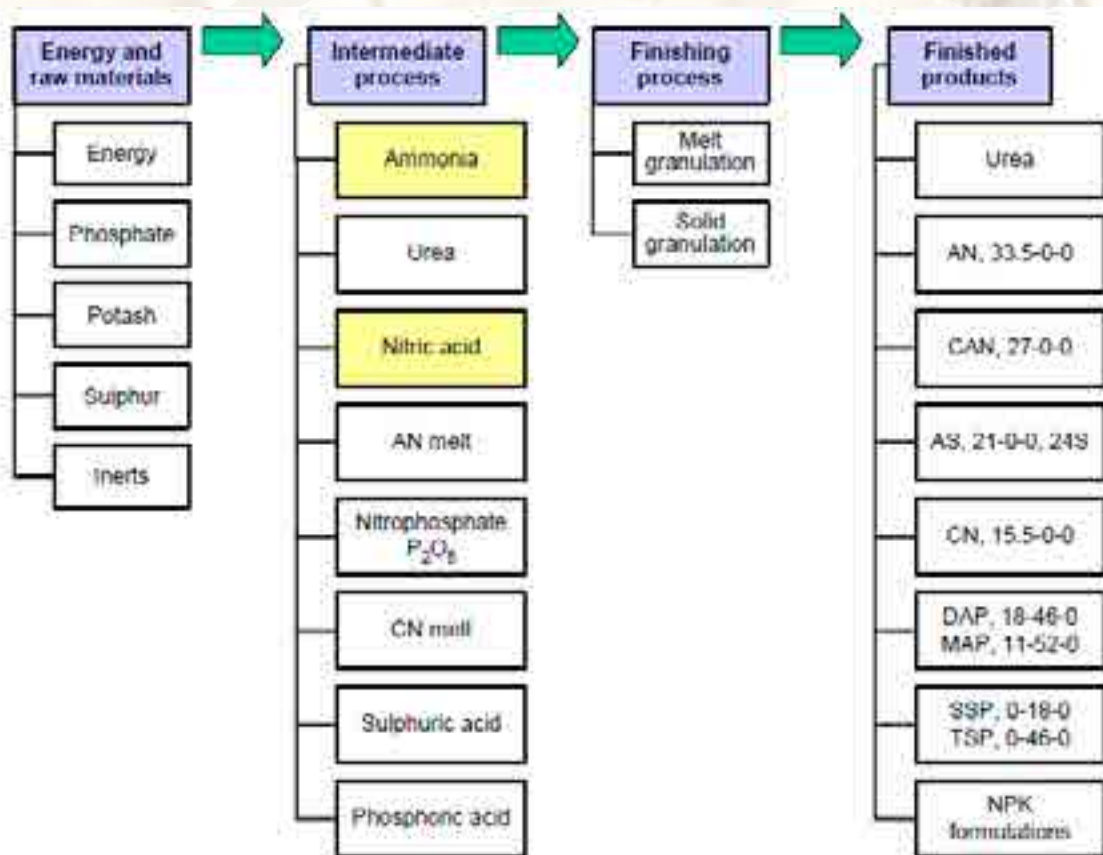


Figure 1: The building blocks used in the CFC for fertilizer production. Intermediates contributing to the majority of emissions are marked yellow (Christensen et al., 2014).

The Fertilizers Europe CFC follows the lines of the general LCA and carbon footprinting rules (ISO 14040/14067), but is not completely compliant with established standards such as PAS 2050 or Carbon Trust. The calculation covers all main sources of GHG emissions and has been externally reviewed by DNV GL to verify its completeness and correct calculations. Tables 1 and 2 summarize selected key background information used to calculate the default reference carbon footprint values for the different world regions.

Table 1: Background information on energy, raw materials and transportation used to calculate the reference carbon footprint values

Energy data		kg CO₂e/GJ	
Energy source	Region	Supply ^a	Use ^b
Natural gas	Europe	10.6	56.1
Natural gas	Russia	13.1	56.1
Natural gas	USA	20.8	56.1
Natural gas	China	12.9	56.1
Liquid petroleum gas (LPG)	Europe	16.3	63.1
Heavy fuel oil	Europe	10.6	77.4
High viscosity residue	Europe	0.0	80.7
Coal (bituminous)	Europe	10.7	94.6
Coal (bituminous)	China	10.5	94.6
Electricity	Europe	34.1	97.8
Electricity	Russia	45.8	121.4
Electricity	USA	47.0	139.7
Electricity	China	54.5	212.2
Electricity (coal-based)	Europe	26.8	238.9
Steam from natural gas (93% efficiency)	Europe	11.4	60.3

Steam from natural gas (93% efficiency)	Russia	14.1	60.3
Steam from natural gas (93% efficiency)	USA	22.3	60.3
Steam from natural gas (93% efficiency)	China	13.9	60.3
Steam from LPG (93% efficiency)	Europe	17.5	67.8
Steam from oil (93% efficiency)	Europe	11.4	83.2
Steam from coal (90% efficiency)	Europe	11.9	105.1
Raw material data ^c		kg CO₂e/t	
Type of raw material		Supply	
Phosphate rock (sedimentary, 4% CO ₂)		47.7	
Phosphate rock (sedimentary, 6% CO ₂)		67.7	
Phosphate rock (igneous, 2% CO ₂)		89.7	
Phosphate rock (igneous, 4% CO ₂)		109.7	
Potassium chloride (Muriate of potash/MOP)		232.2	
Potassium sulphate (Sulphate of potash/SOP)		108.4	
Dolomite (ground)		61.9	
Limestone (ground)		61.9	
Transport data ^d		kg CO₂e/t*km	
Means of transport			
Deep sea vessel		5	
Coastal shipping		16	
Barge		31	
Rail		22	
Truck		62	

^a Data from GaBi database (PE International, 2013)

^b Data for fossil fuels and steam from IPCC (2006), for electricity from IEA (2012)

^c Data for raw materials based on Jenssen & Kongshaug (2003), validated by Fertilizers Europe Technical Committee (personal communication, 2014)

^d Data for transport from McKinnon & Piecyk (2011)

Table 2: Reference values for the energy input required for ammonia and nitric acid production and direct emissions of nitrous oxide (N₂O).

Product	Region	Energy input			Direct emissions	
		Feedstock & fuel	Electricity ^a	Steam ^a	N ₂ O	
		Type	GJ/t product	GJ/t product	GJ/t product	kg/t product
Ammonia	Europe	Natural gas	34.7	0.79	-1.37	0
Ammonia	Russia	Natural gas	40.5	0	0	0
Ammonia	USA	Natural gas	35.7	0	0	0
Ammonia	China	Natural gas	42.2	0	0	0
Ammonia	China	Coal	54.0	0	0	0
Nitric acid	Europe	-	0	0.3	-1.75	0.87
Nitric acid	Russia	-	0	0.3	-1.75	7.40
Nitric acid	USA	-	0	0.3	-1.75	6.00
Nitric acid	China	-	0	0.3	-1.75	5.70

^a Assumptions by Integer Research Ltd (2014):

For non-European ammonia no steam generation and zero electricity consumption were assumed.

For non-European nitric acid the steam and electricity data from Europe were assumed.

3. Results & discussion

Table 3 shows the carbon footprint (CFP) values for the main mineral fertilizer products. The new European reference values are also included in the on-farm GHG calculation tool “Cool Farm Tool” and the regional values will be added soon (www.coolfarmtool.org).

Table 3: Reference carbon footprint (CFP) values for main mineral fertilizer products from different regions (reference year 2011)

Fertilizer product	Nutrient content	CFP at plant gate (kg CO ₂ e/kg product)			
		Europe	Russia ^c	USA ^c	China ^c
Ammonium nitrate	AN 33.5% N	1.18	2.85	2.52	3.47
Calcium ammonium nitrate	CAN 27% N	1.00	2.35	2.08	2.86
Ammonium nitrosulphate	ANS 26% N, 14% S	0.82	1.58	1.44	2.22
Calcium nitrate ^a	CN 15.5% N	0.67	2.03	1.76	2.20
Ammonium sulphate	AS 21% N, 24% S	0.57	0.71	0.69	1.36
Di-ammonium phosphate	DAP 18% N, 46% P ₂ O ₅	0.64	0.81	0.73	1.33
Urea ^b	Urea 46% N	0.89	1.18	1.18	2.51
Urea ammonium nitrate ^b	UAN 30% N	0.81	1.65	1.50	2.37
NPK 15-15-15	NPK 15% N, 15% P ₂ O ₅ , 15% K ₂ O	0.73	1.40	1.27	1.73
Triple superphosphate	TSP 48% P ₂ O ₅	0.18	0.25	0.19	0.26
Muriate of potash	MOP 60% K ₂ O	0.23	0.23	0.23	0.23

^a CN is assumed to be produced as co-product from NPK production via nitro-phosphate route (EFMA, 2000c)

^b Urea and UAN contain CO₂, which will be released shortly upon application to soil (0.73 kg CO₂/kg urea and 0.25 kg CO₂/kg UAN). This amount is not included in the plant gate CFP.

^c For Russia, USA and China specific values were used for energy supply, energy consumption for ammonia production and N₂O emissions from nitric acid production. All other values are equal to Europe. Specific assumption for China: 80% of ammonia production is based on hard coal; remainder on natural gas (IFA, 2009).

Figure 2 documents the improvements in fertilizer production which are particularly obvious when comparing the CFP of nitrate-containing products produced in Europe as shown for the example of calcium ammonium nitrate (CAN), which contains 50% nitrogen as nitrate. Values such as that from Ecoinvent (2002) represent European production technology of the 1990ties or earlier (Patyk & Reinhardt, 1997; Kongshaug, 1998). At that time nitric acid, which is the precursor of nitrate-N in mineral fertilizer, was produced without any abatement technology for N₂O emissions occurring at significant rates during the nitric acid production process (see also Table 2). The first Fertilizers Europe reference value for CAN represents production technology in 2006 (Brentrup & Palliere, 2008) and shows already some improvement due to partly installation of N₂O abatement catalysts in European nitric acid plants. Today, practically all European nitric acid plants are equipped with this technology, which led to an average reduction of N₂O emissions by 80-90% as compared to the pre-abatement time.

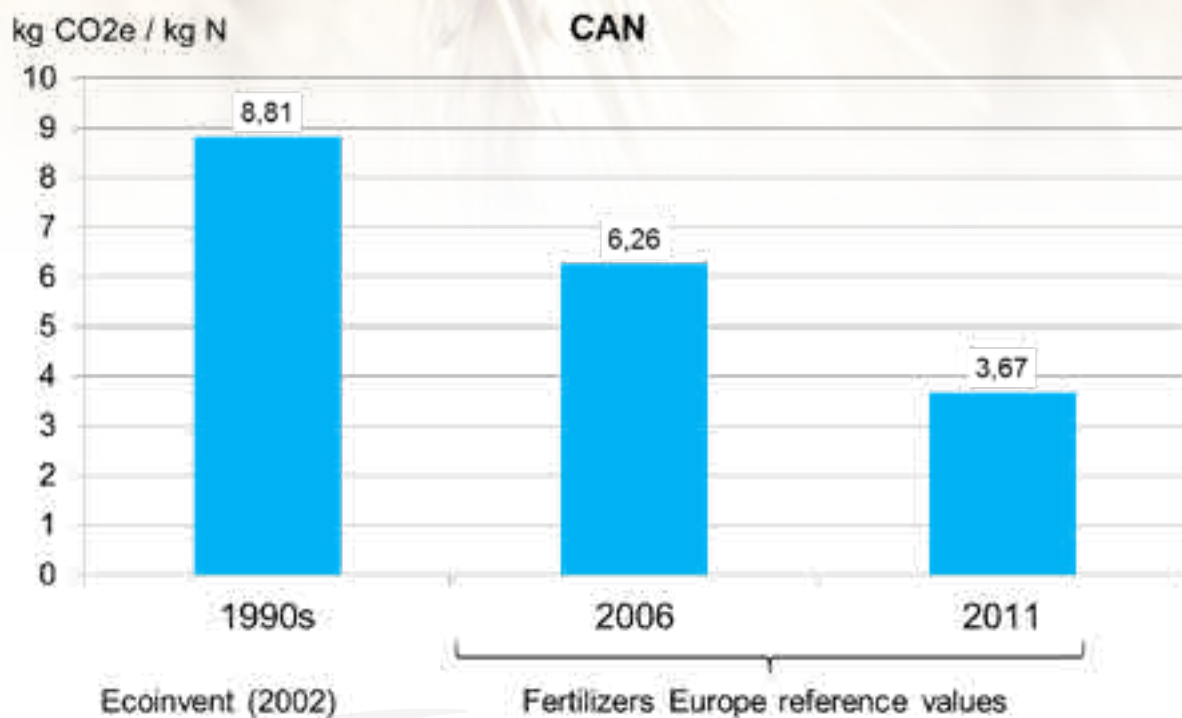


Figure 2: Development of carbon footprint values for Calcium ammonium nitrate (CAN) production in Europe from the 1990s until 2011

In order to compare the production carbon footprint of different fertilizer products the values need to be related to the same functional unit, which is in the case of nitrogen fertilizers one kg of N. Figure 3 compares three different products (AN, Urea, UAN) that contain only N as a plant nutrient and are therefore directly comparable without any allocation that would be required for multi-nutrient products. The graph shows that European production of all three products results in the lowest CFP among the production regions compared. The European values for AN, Urea and UAN are very close to each other at around 3.5 kg CO₂e/kg N. The CO₂ released during the hydrolysis of urea after application to soil is included in this comparison because it is in principle only a delayed emission of CO₂ that has been previously used in the factory to produce urea from ammonia. The same amount of CO₂ needed to synthesize urea will be emitted after application in the field. This is also valid for the urea part in UAN (50% urea, 50% ammonium nitrate).

The differences between the production regions are more obvious for nitrate-containing products than for urea. However, the differences are not only related to the existence and efficiency of N₂O abatement in nitric acid, but also strongly influenced by the source of fossil fuels used for the production of ammonia. China's ammonia production is still dominantly based on coal (IFA, 2009; Zhang, 2013) and this is the reason for the high CFP values for all three N fertilizers. N₂O emissions from nitric acid production in China is even lower than in the US and Russia (Table 2), but this does not compensate for the higher CO₂ emissions from coal-based ammonia production. The reason for lower N₂O emissions in China is the Clean Development Mechanism (CDM) under the Kyoto Protocol of the United Nations (IPCC, 2007), which supports certain emission reduction projects in particular in countries not included in the Annex I of the Kyoto Protocol (e.g. transition and developing countries). Russia and USA show low CFP of urea indicating high energy efficiency, but higher values for AN and UAN due to missing or very limited installation of N₂O abatement in nitric acid production.

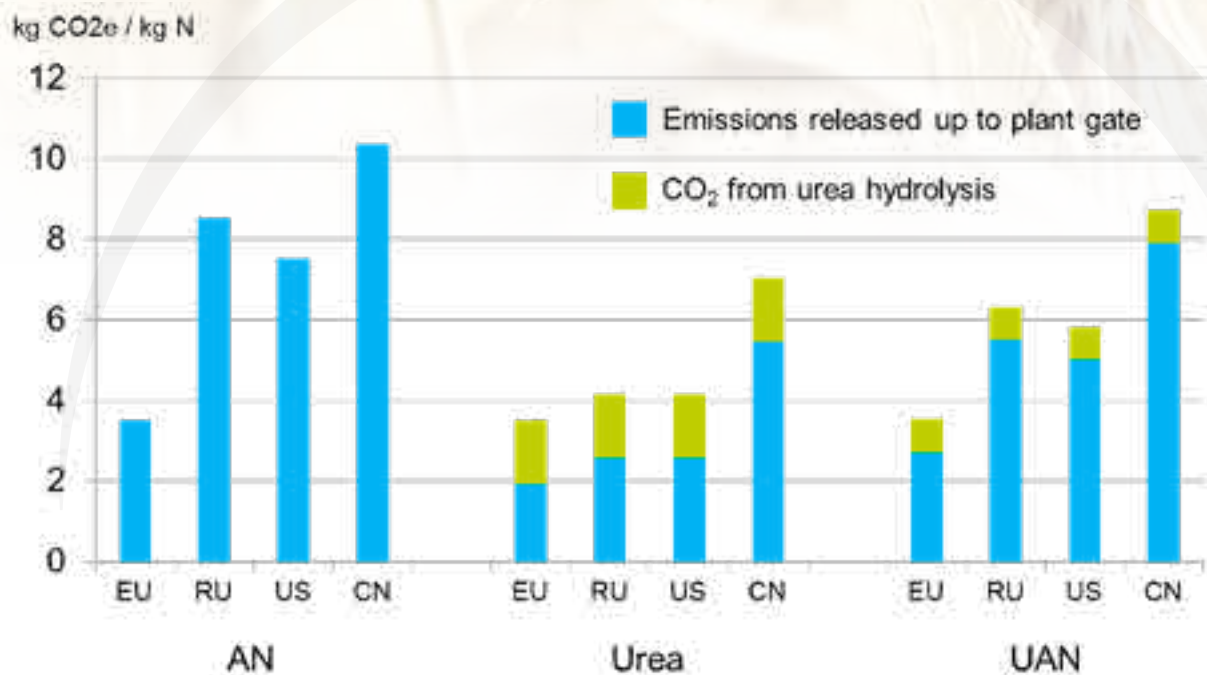


Figure 3: Carbon footprint of ammonium nitrate (AN), urea, and urea ammonium nitrate solution (UAN) expressed per kg of N and produced in different regions.

Fertilizer production is an important contributor of GHG emissions to the carbon footprint of agricultural products. When evaluating the CFP of fertilizers it is even more important to consider the complete life-cycle emissions in order to account for all additional sources of GHG emissions beyond the production step. This is mainly relevant for nitrogen fertilizers because their use in agriculture can lead to N₂O emissions via different pathways. In addition, the use of most N fertilizers acidifies the soil, which is usually compensated by application of lime releasing CO₂ after conversion in soil. Figure 4 summarizes the GHG emissions from production and application of different N fertilizers. The emissions from the use of the fertilizers were estimated using current default values for the different pathways:

- (1) IPCC Tier1 emission factor for direct N₂O emissions,
- (2) EMEP/EEA Tier 2 emission factors for NH₃ emissions (EEA, 2013) plus IPCC (2006) for indirect N₂O via NH₃,
- (3) IPCC (2006) for indirect N₂O via NO₃,
- (4) KTBL (2005) figures for lime demand together with IPCC (2006) for CO₂ from lime.

The resulting values are only rough estimates since for instance potential differences between the N products in their agronomic efficiency are not taken into account. This means that for example high losses as ammonia could lead to an additional need for N application in order to substitute for the lost nitrogen, which would then increase the CFP of this product. There is also evidence that different N forms behave differently in terms of direct N₂O emissions from soil. Applied to well drained soils without anaerobic conditions, the emission rates are often clearly lower than the default IPCC factor of 1% N₂O-N per unit N applied and the emissions usually decline with an increasing share of nitrate in the product. Poorly drained soils and high precipitation can lead to anaerobic conditions. This together with high organic soil carbon availability triggers N₂O emissions by denitrification. Under those conditions urea and ammonium based N fertilizers often show lower emissions than nitrate-containing products. However, using standard default emission factors for all pathways the overall CFP of urea is 8% higher than that of AN (see Fig. 4).

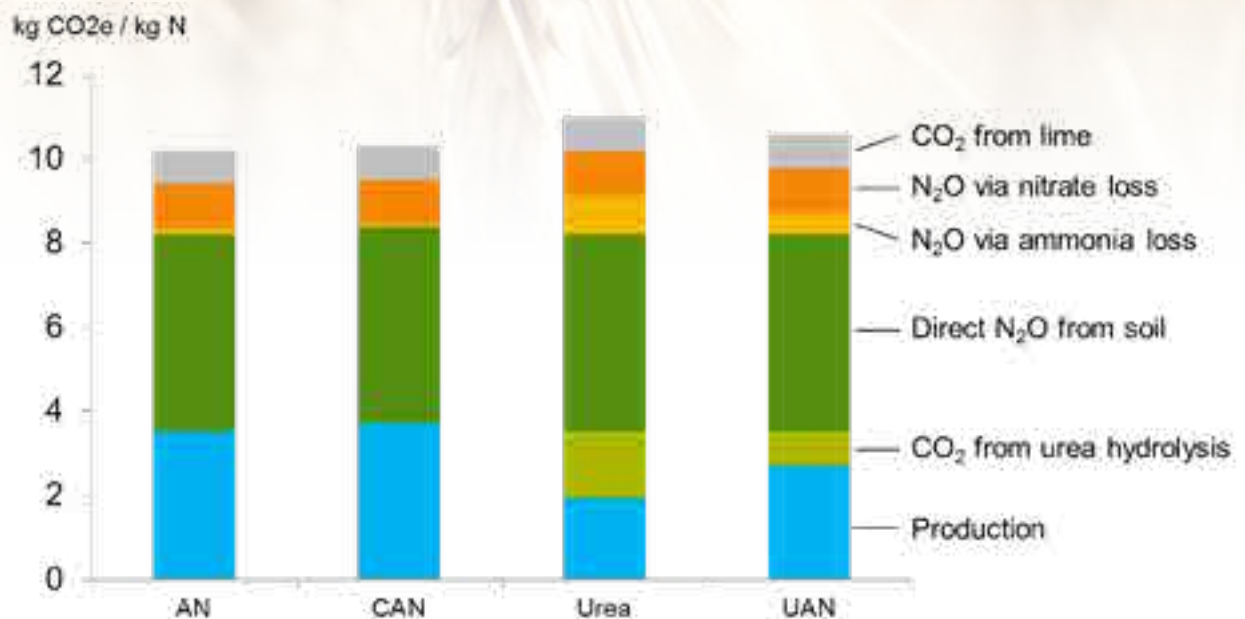


Figure 4: Carbon footprint of N fertilizer production and use based on default emission factors (for references see text).

4. Conclusions

The paper explains the methodology applied in the calculation of the carbon footprint values. The results show carbon footprint values per fertilizer product and production region (Europe, Russia, USA, and China). We suggest that these data should be used in carbon footprint studies as reference values for fertilizer production with the technology baseline 2011. The data clearly show the improvements made in Europe in particular in terms of N₂O emission reduction in nitric acid production, which is an intermediate in the production of all nitrate-containing N fertilizer products. The differences between the production regions are mainly due to two aspects, (1) absence or presence of N₂O emission control and (2) energy source (coal or gas) and efficiency in ammonia production.

For a valid conclusion about the carbon footprint of fertilizers it is necessary to include the use of the fertilizers into the analysis in order to have a complete picture along their life-cycle. Emissions from N fertilizer use on field can be even higher than of their production, in particular when improved production technology as in Europe is employed. Emissions from N fertilizer use are highly variable depending on soil and climate conditions and need to be assessed with care. The use of standard default emission factors suggests slightly higher life-cycle GHG emissions from urea as compared to ammonium nitrate.

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277. ECOALIM: a Dataset of the Environmental Impacts of Feed Ingredients Used for Animal Production in France

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ABSTRACT

Feeds contribute highly to the environmental impacts of livestock products. Therefore formulating low-impact feeds necessitates data of the environmental impacts of feed ingredients with consistent perimeters and methodology for life cycle assessment (LCA). We built the ECOALIM dataset of life cycle inventories (LCI) and associated impacts for feed ingredients utilised in animal production in France. It provides several perimeters for LCI (field gate, storage organization gate, plant gate and harbour gate) with homogeneous data source from R&D French institutes covering the period 2005-2012. The dataset of 149 environmental impacts is available as an Excel spreadsheet on the ECOALIM website and provides ILCD and CML climate change and acidification, eutrophication, CED non-renewable energy, phosphorus consumption, and CML land occupation. Life cycle inventories of the ECOALIM dataset are available in the Agribalyse® database in SimaPro® software. The ECOALIM dataset can be utilized by the feed manufacturer and the LCA practitioner to investigate the formulation of low-impact feeds. It also provides data for environmental evaluation of feeds and environmental evaluation of animal production systems.

Keywords: Life cycle inventory, raw material, livestock, database, formulation

1. Introduction

The Food and Agriculture Organization (FAO) argued that there is no technically or economically viable alternative to intensive animal production to feed the world (FAO, 2011). However, it is necessary to mitigate pollutant emissions related to livestock and, consequently, to estimate accurately them in order to identify mitigation options (Gerber et al., 2013).

In pig and poultry systems, the production of feeds contributes strongly to the environmental impacts of the animal product. In particular, it accounts for 50% to 85% of the climate change impact, 64% to 97% of eutrophication potential, 70% to 96% of the energy use and almost 100% of the land occupation (Basset-Mens and van der Werf, 2005; Boggia et al., 2010; Leinonen et al., 2012a, b; da Silva et al., 2014; Dourmad et al., 2014). In dairy production, enteric methane has the major contribution to GHG, but 27% to 38% of the nitrous oxide emissions are related to feed production (FAO, 2010). In beef production, the feed contribution to GHG emissions can reach up to 55% (Nguyen et al., 2010). Environmental impacts of feeds are highly determined by their composition in feed ingredients. Therefore, the investigation of mitigation strategies needs robust and accurate data on environmental impacts of feed ingredients, relying on consistent perimeters and methodologies for LCA.

Feed formulation in France involves mostly French feed ingredients that result from several cropping systems under various soil and climate conditions and cropping practices. A previous study (Nguyen et al., 2012) already highlighted the need for homogeneously developed data on environmental impacts of feed ingredients to formulate low-impact feeds and investigated mitigation options. The ECOALIM dataset is the result of a collaborative project between research and extension institutes, with the participation of animal nutrition firms. ECOALIM gathers the most accurate and representative data on life cycle inventories available to date for French feed ingredients. This

database can be utilized in France and in countries importing French feed ingredients as well. This paper presents the methodological choices and the perimeters implemented to develop the ECOALIM dataset, as well as its potential application.

2. Methods

For the ECOALIM database, several system boundaries have been defined (Figure 1): field gate, storage organization gate, plant gate, and harbour gate. Field gate is relevant for assessment of feed impacts in the case of on-farm feed production (e.g. crops produced on-farm and directly utilized by animal production unit). Feed formulation by feed companies necessitates some other boundaries: plant gate for coproducts of cereals, oilseeds and protein crops (e.g. meals) as well as for industrial products (e.g. amino acids) processed in France; storage organization gate for cereals, and harbour gate for imported feed ingredients.

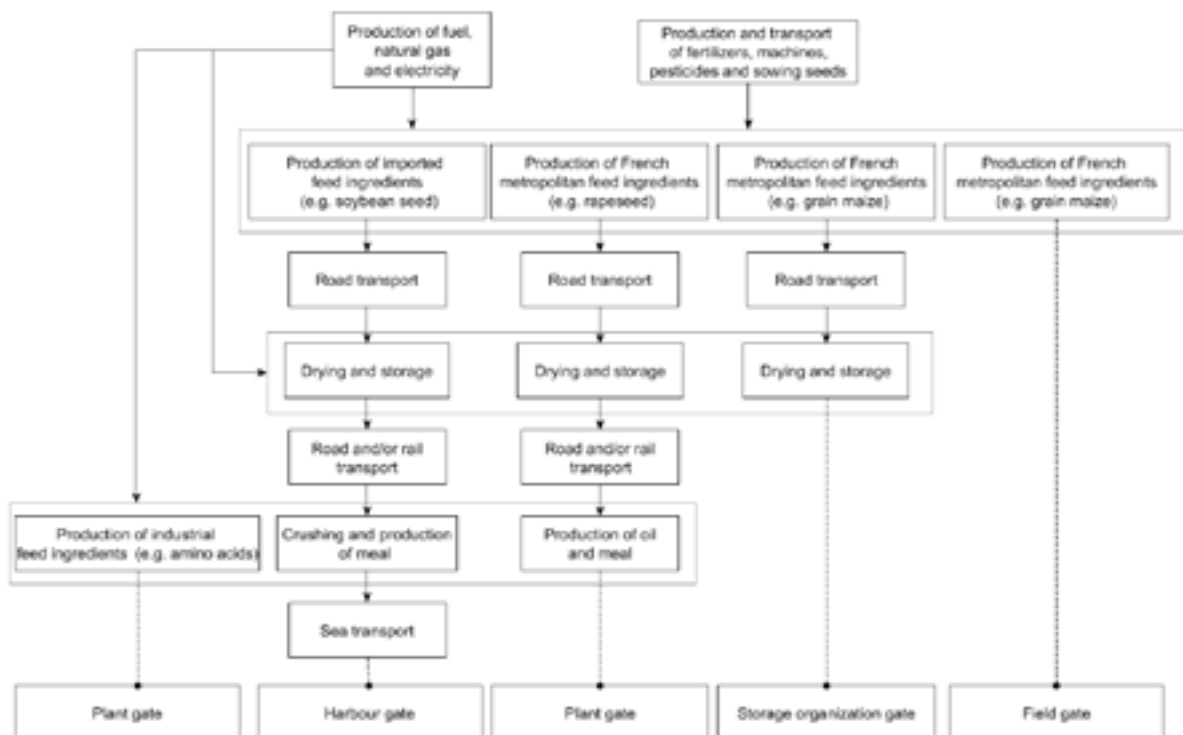


Figure 1. Flow diagram for production of the feed ingredients included in the ECOALIM dataset, with main processes for the production of crop inputs, crop production and feed ingredient production. System boundaries include all sub-processes.

In the ECOALIM dataset, the impacts of the coproducts of cereals and maize as well as oils and meals were calculated homogeneously with an economic allocation using the olympic five-year (2008-2012) average price of each coproduct. The functional unit considered is the kilogram of feed ingredient at the reference humidity rate, which is the usual functional unit for feed ingredients.

Life cycle inventories for French crops were based on the inventories of the French national database of the main agricultural products Agribalyse® (Koch and Salou, 2015), with updating emission factor for ammonia (EMEP/EEA, 2013) and also the references for agricultural practices when available. One major improvement relatively to Agribalyse® is the distribution of the impacts associated to phosphorus fertilization and nitrate leaching among the various crops involved in the same crop rotation (according to crop requirements and removal for phosphorus, and equally among crops for nitrate leaching).

For all the French crops, the average inventories (yields, amounts of fertilisers, pesticides and seeds) were obtained from French agricultural data (UNIP, 2005-2009; Agreste, 2005-2012, 2006). For the main agricultural ingredient (maize, wheat, barley, rapeseed and sunflower), additional specific inventories were constructed for systematic covercropping, systematic organic fertilization, and introduction of protein crop in the crop rotation. For these specific inventories, data came from the experimental farm network of the French agricultural institute dedicated to cereals (Arvalis-Institut du Végétal) in order to assess the variability of the results in function of the crop management.

For European and non-European crops, data for resources used came from scientific publications (Nemecek and Kägi, 2007; Prudêncio da Silva et al., 2010; Roeder et al., 2014) and statistics from national databases (Service, 2010; Ukraine, 2012; Service, 2014b, a) or FAOStat (FAO, 2013). Resources and emissions inventories were calculated according to previously published methodologies (Prudêncio da Silva et al., 2010; Boissy et al., 2011). Concerning industrial products, the dataset includes products directly issued from crops and their coproducts associated to processes of extrusion, crushing, milling, distillation...such as oils, meals, Dried Distilled Grains with Solubles (DDGS), molasses, pulps, flours. Products secondary issued from crops are starch and gluten feeds. Transformation plants have been surveyed to collect the relevant underlying data in 2013-2014, excepted for French oils and meals and non-French feed ingredients (Weidema et al., 2013). Inventories for the other industrial products come from Garcia-Launay et al. (2014) for feed-use amino acids, industrial confidential data for minerals, French Technical Centre for Meat (ADIV) data for animal byproducts, and from Ecoinvent data adapted to the French context for vitamins (Weidema et al., 2013).

The ECOALIM dataset uses the ILCD characterization method recommended by the Joint Research Centre (JRC, 2012) as well as the CML IA characterization method (PréConsultants, 2015) which is the most popular in agricultural LCA. Energy demand is calculated according to the CED 1.8 method (PréConsultants, 2015).

Therefore, climate change with (CCLUC) or without land use change (CC) are expressed in kg CO₂-eq, acidification (AC) in both molc H⁺-eq (AC_{ILCD}) and kg SO₂-eq (AC_{CML}), terrestrial eutrophication (EU) in molc N-eq, freshwater EU in kg P-eq, Marine EU in kg N-eq, EU_{CML} in kg PO₄³⁻-eq, land occupation (LO) in m².y, and non-renewable (CEDNR) and total energy demand (CEDTOT) in MJ. Phosphorus demand (PD) is expressed in kg P and sums up all the phosphorus and phosphate inputs along the life cycle.

Calculations were made using SimaPro software v8.0.5.13 (PRé Consultants, Amersfoort, The Netherlands) and the Ecoinvent v3.1 database attributional for background data (Weidema et al., 2013).

3. Dataset description

Table 1 contains the number of inventories and impacts available in the dataset for each type of feedstuffs. The whole dataset contains 149 average feed ingredients and 16 feed ingredients from specific crop itineraries. It offers a wide range of feedstuffs for utilization in various livestock productions (cattle, pig and poultry) and different perimeters for LCA, for feed formulation by feed manufacturers, farm cooperatives as well as for on-farm feed production.

Table 1: Number of data points available in the ECOALIM dataset according to the type of feed ingredients and the perimeter.

Types of feedstuffs	At field	At french harbour	At mill or transformation plant	At storage agency
Cereals	8	2	0	17
Coproduct from food industry	0	0	2	0
Coproduct of maize	0	1	5	0
Coproduct of wheat	0	0	5	0

Fats	0	6	22	0
Industrial amino acids	0	0	5	0
Minerals	0	0	10	0
Oil seeds and protein crops	6	0	4	13
Oilmeals	0	7	22	0
Other coproduct of animal origin	0	0	2	0
Other coproduct of plant origin	1	1	4	0
Silage	5	0	0	0
Vitamin	0	0	1	0

Table 2 provides an extract of the dataset for the main cereals, oil and protein crops produced in France. It highlights the relative homogeneity of impacts among cereals and the higher variability for oil and protein crops.

Table 2: Impact values of the main French feed ingredients at storage organization gate, for 1 kg of feedstuff.

Feedstuff	CCLUC, kg CO ₂ eq ILCD	CEDNR, MJ	EU, g PO ₄ ³⁻ eq CML	AC, g molc H ⁺ eq ILCD	LO, m ² .an CML	PD, g P
Oat	0.52	3.07	5.1	12.6	2.09	3.6
Soft wheat	0.43	2.85	3.7	10.7	1.34	4.1
Barley	0.39	2.71	3.7	9.5	1.48	4.1
Maize grain	0.46	4.47	3.6	13.4	1.23	3.5
Sorghum	0.36	2.53	3.6	4.8	2.12	5.2
Triticale	0.50	2.95	5.3	8.7	1.84	2.6
Pea	0.19	2.21	3.7	2.5	2.32	2.9
Faba bean	0.19	1.70	3.2	2.0	1.99	5.2
Rapeseed	0.94	5.49	7.6	21.0	3.12	7.3
Flaxseed	0.93	6.27	10.4	19.1	5.53	11.0
Soya bean	0.30	5.08	6.0	3.4	3.81	4.6
Sunflower seed	0.56	4.21	8.8	10.8	4.76	6.6

4. Discussion

The ECOALIM dataset can be mobilized for the environmental assessment of feeds, animal production systems and mitigation options (Figure 2). The dataset for impacts is available in an Excel file (available online at <http://www6.inra.fr/ecoalim>) and also in the Agribalyse® V1.3. database in SimpaPro®.

Both the feed manufacturer and the LCA practitioner can formulate feeds through classical least-cost formulation and calculate the environmental impacts of the obtained feeds through the ECOALIM Excel dataset. To investigate mitigation options they can also formulate eco-feeds through simultaneous economic and environmental formulation, using the ECOALIM Excel dataset in formulation software. The ECOALIM dataset in the Agribalyse® database in SimaPro® can be mobilised by the LCA practitioner in a more detailed farm inventory for the environmental evaluation of the animal product at farm gate using either the formulas of least-cost formulated feeds or the formulas of the eco-feeds. For these applications, the ECOALIM dataset offers harmonized, updated and reviewed data. It covers a range of feed ingredients that was not previously available in Agribalyse® and is to date the only dataset that relies on reliable foreground data representative for

France, collected with various surveys gathered in a common framework. To our knowledge, ECOALIM is the first dataset available for feed ingredients inventories and impacts that provides the whole set of the feed ingredients utilized in livestock production in France and specific inventories for systematic covercropping, systematic organic fertilization, and introduction of protein crop. It also applies to other European countries for several imported feed ingredients as well as for French feed ingredients exported. The incorporation of the ECOALIM dataset into the Agribalyse® database will allow further updating of the life cycle inventories and the addition of new feed ingredients such as fish meal and oil for aquaculture production. Furthermore, the integration of Agribalyse® database into Simapro® software ensures appropriate maintenance of the data and a wider dispersion among LCA practitioners.

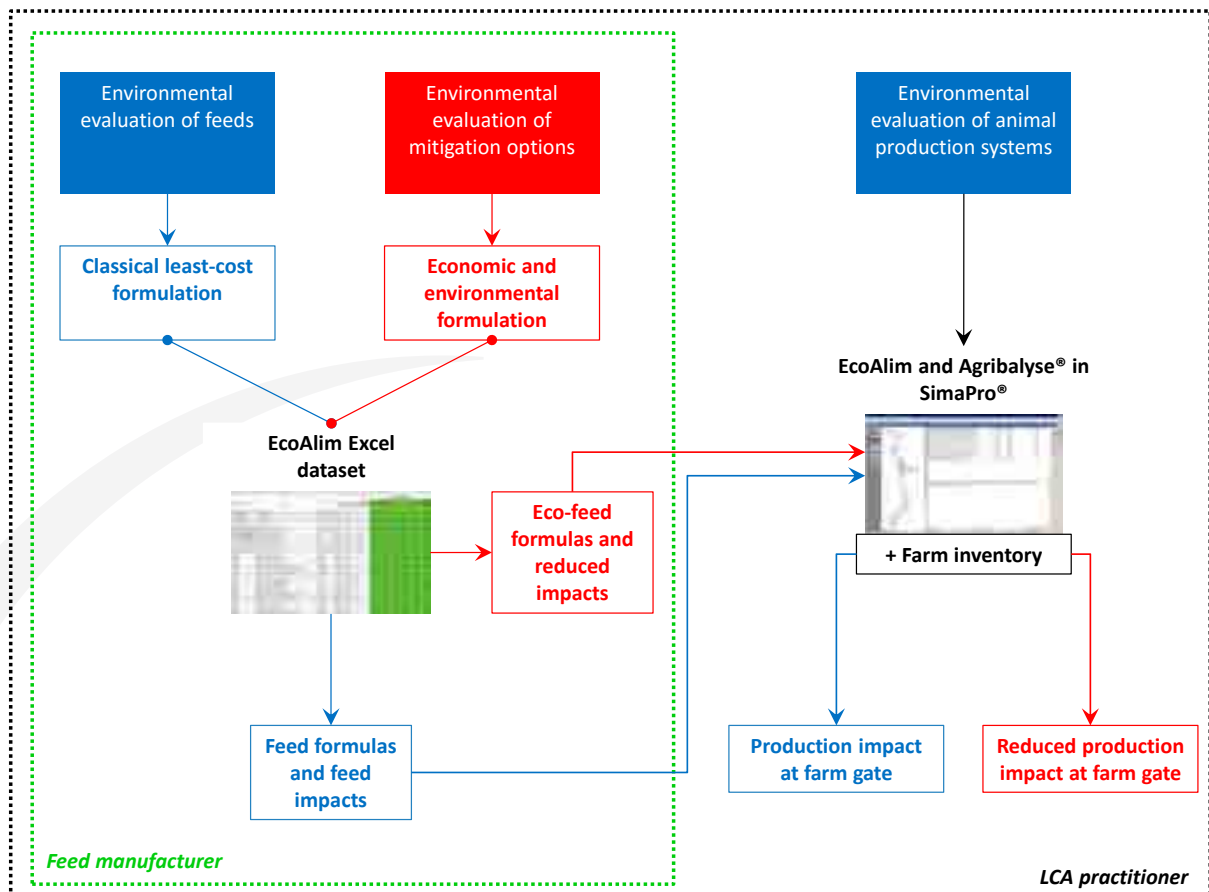


Figure 2: Diagram of the potential utilizations of the ECOALIM dataset. The blue pathway corresponds to the methodology utilized when evaluating classically the environmental impacts of feeds and animal products at farm gate. The red pathway illustrated the methodology utilized when evaluating mitigation strategies of the environmental impacts of feeds and animal products at farm gate.

5. Conclusions

The ECOALIM dataset provides life cycle impacts of feed ingredients produced with harmonized methodology, homogeneous data source and background data. This approach avoids double-counting of the environmental burden. The whole approach was developed in coherence with the Agribalyse® database that will allow further utilization of the ECOALIM dataset for environmental labelling of animal products. One opportunity will be to propose these data to the European initiative “Product Environmental Footprint” for environmental labelling of market products.

The ECOALIM dataset relies on representative and recent data that cover a wide range of production itineraries. Built with organizations including feed manufacturers, raw material producers and R&D institutes, this dataset contains feed ingredients necessary to formulate composed feeds and some French feed ingredients for which no data was available before (oat, sorghum, flaxseed, faba bean, soybean). The ECOALIM dataset will allow proposing feed formulas incorporating nutritional, economic and environmental constraints. This new approach of diet conception needs further research to evaluate its potential as a mitigation option for livestock-related impacts.

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182. Approach to model future protein supply strategies for animal feed in LCA

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ABSTRACT

With an aim of reducing land use in the production of feed for animals in the future, this consequential Life Cycle Assessment (LCA) was carried out to evaluate the environmental consequences of different protein supply strategies.

A case study was performed for piglet feed manufactured in Northern Europe. The functional unit was the production of 1 tonne of piglet feed. Three main scenarios of supplying protein for compound piglet feed were assessed: Business As Usual (BAU) reflecting current practices where soybean meal is a key protein ingredient, a scenario involving a maximum of local protein ingredients such as peas and fava beans (Local); and a scenario involving the incorporation of a maximum of single cell protein (SCP). These scenarios were applied to four piglet categories (piglets weighing, 20-35 kg, 10-20 kg, 6-10 kg and 4-6 kg), for a total of 12 scenarios. Background data were taken from the Ecoinvent v3.2 database, and foreground data for the SCP production were obtained directly from a SCP producer. For each scenario, the amount of each crop and non-crop ingredients were determined. The impacts of land use changes (LUC) and changed manure compositions were also included.

Preliminary results show that, though the inclusion of SCP did allow reducing the use of soybean meal and the associated land use changes, it did not lead to an overall reduction of the global warming potential, as compared to the BAU scenario, specially in the larger piglet groups. This was essentially due to the high LUC values and the high CO₂ emissions from the electricity and Oxygen usage in SCP production. In conclusion, the study showed that under current framework conditions, the BAU protein supply scenario led to a better environmental performance than the SCP and local supply strategies in the larger piglet groups whereas the SCP protein supply scenario performed better in the smaller piglet groups.

Future work will study the potential of further optimization of the SCP scenario, for example through using more efficient crop ingredients and cleaner energy sources.

Keywords: Compound pig feed formulation, consequential life cycle assessment, amino acids, protein supply strategies, single cell protein.

1. Introduction

The demand for protein has rapidly increased due to global dietary changes favoring more meat, resulting in a sharp increase of livestock feed which again constitutes mainly of protein. At present the main protein source in animal feed consists of plant proteins, majority of which is soybean. Since mostly soybeans are imported to Denmark from as far as South America it carries a huge carbon footprint which is affecting the environment in a negative manner. Therefore other options of protein sources, i.e. the use of a local plant protein such as Fava Bean could be considered as a short term alternative.

However, increased demand on plant proteins, increases land use changes which in turn results in deforestation and increased amounts of global carbon emissions, anywhere in the world. To overcome this global issue and to practice a more sustainable system, different strategies can be considered. One such strategy is use of Single Cell Proteins (SCP) in animal feed production.

In this research the aim was to uncover the environmental consequences of different protein supply strategies. A case study was made with piglet feed, as it uses a large amount of soybean meal and other crop based protein sources. This research also includes a full account of the impacts of land use changes in each strategy.

Three different protein strategies were compared: i) soybean meal as a main protein source (“business as usual”; BAU) ii) a strategy using a local protein crop as a main protein source (“local” strategy; here fava bean); iii) a SCP as the main protein source. The latter consists of Uniprotein produced by the company Unibio A/S. These protein strategies are applied to four key segments of piglet feed for: piglets weighing 20-35kg, 10-20kg, 6-10kg and 4-6kg.

This study focuses on piglet feed since this is a type of feed incorporating a high share of soybean meal.

A consequential Life Cycle Assessment (LCA) was performed and simulations were carried out with the use of Simapro 8.1.0.60 faculty software to measure the environmental consequences of each of the 3 protein supply strategies.

2. Methods (or Goal and Scope)

2.1 Life Cycle Assessment Model

Life Cycle Assessment (LCA) is a standardized comparative environmental assessment methodology (ISO, 2006) which consists of assessing and comparing the environmental impacts of selected products (UNEP, Life cycle assessment). The method used to conduct the LCA for this research was Consequential LCA. The functional unit (FU), which is a quantitative measure of the functions that the goods (or service) provide, related to all input and output flows in this study are calculated as the production of 1 tonne of piglet feed in 1 year, which also covers the temporal system boundary. The Goal and Scope were to make LCAs to uncover the environmental consequences of different protein supply strategies (BAU protein, local crop based protein - fava bean and Uniprotein – a SCP produced by the company Unibio A/S) in producing piglet feed. The geographical system boundary considered was Denmark, in terms of certain inputs and regulations pertaining to animal feed manufacturing which are applicable to Denmark as the final feed is produced in Denmark. In making the Life Cycle Impact Assessment the method used was ILCD V1.01. Out of the many impacts in this method Climate Change (IPCC 2014) – Global Warming Potential was selected for this research. Land use changes were also considered together with this impact category in making the assessments. To generate the results, background data were obtained from Ecoinvent 3 database of Simapro 8.1.0.60 faculty software. Foreground data were obtained from Unibio A/S Company as well as Vestjyllands Andel feed manufacturing company in Ringkøbing.

2.2 Scenario 1: Business as usual

Based on current optimization where all the amino acids are supplied at best price this scenario was modelled. As expected this included mostly crop based proteins in particularly soy.

2.3 Scenario 2: Local protein

The rationale behind selecting this scenario was to check the protein source which was the cheapest, easily available and costed the lowest to transport. Examples were fava bean and peas.

2.4 Scenario 3: SCP

The SCP is a microbial protein produced by fermentation; through the availability of Oxygen and a Nitrogen source, the microorganisms convert a carbon source (here natural gas) to protein. The microorganism *Methylococcus capsulatus* can grow on cheap carbon sources such as methane and methanol. *M. capsulatus* has a protein content of approximately 70% and can be used for production of so-called SCP. Static simulations of SCP production using *M. capsulatus* in an U-loop reactor have been used to determine the optimal operation. The optimal operating point is located close to both washout and oxygen limitation. With oxygen being the most expensive reactant, the U-Loop reactor is operated in the oxygen limited mode and substrate feed is controlled according to the oxygen feed. The maximum oxygen feed is determined by the maximum biomass concentration that can be

tolerated in the reactor. Higher biomass concentration is believed to give a more viscous fluid that requires more pumping energy for circulation in the U-loop and has a lower gas-liquid oxygen transfer rate. The optimal dilution rate is relative constant around 0.2 1/hr. This operating strategy gives the highest SCP productivity.

Unlike many other bio-chemicals, SCP is a commodity. Therefore, it is essential that the process equipment for manufacturing of SCP is energy efficient and able to utilize the raw materials with a high yield. The principle problems facing the manufacture of SCP is transfer of oxygen to the liquid phase and removal of the high amount of heat produced by the exothermic process. Conventional stirred tank fermenters cannot provide sufficient mass transfer of oxygen nor sufficient area for removal of the heat produced. The U-loop reactor is a reactor designed to have high degrees of gas-liquid mixing and heat removal. The legs (the u-loop) of the reactor are equipped with static mixers to have high gas-liquid mass transfer rates. The u-loop is also equipped with a heat exchanger for removal of the extensive heat produced by the fermentation. The tank on the top of the legs is essentially a degassing unit. It is used to separate the produced CO₂ from the liquid. Furthermore, as long as the substrates (methanol, oxygen, nitric acid, and minerals) are present, the microorganism will also grow in the tank at the top.

2.5 Market-based approach

The formulae were carefully optimized to supply the best possible nutrition, protein (amino acids) and price. Modelling was done in Agrosoft which contains a database with detailed biochemical composition of all feed ingredients (e.g. minerals, amino acids, etc.). The amino acid profile for the SCP was supplied by Danish SCP producing company.

2.6 Modelling changes in the composition of compound pig feed

Table 1 gives a better understanding of the implemented approach with details on the ingredients used as well as the country of origin and percentages of ingredient used in each scenario. The AgroSoft® software was used to calculate the compound pig feed recipes for each scenario.

2.7 Manure Management

The manure from the piglets fed with the FU was taken through the process of calculations to find out the specific amounts in In-house manure management, outdoor-manure management and application on land. Manure management is one consequence of changing the protein content in each feed with reference to the Nitrogen and Phosphorous content of it. By calculating the amount of manure excreted and emissions related to them the figures were obtained for each scenario (Hanne Damgaard, 2015, Hamelin et. al., 2011 and Tonini et. al., 2015).

Table 1: composition of ingredients in piglet feed in 12 scenarios (data from Vestjylland, Andel, Ringkøbing) in %¹

Component/ ingredient	Country of origin	20-35 kg group (% FM in feed)			10-20 kg group (% FM in feed)			6-10 kg group (% FM in feed)			4-6 kg group (% FM in feed)		
		BAU	Local Protein	SCP	BAU	Local Protein	SCP	BAU	Local Protein	SCP	BAU	Local Protein	SCP
Palm oil mix	Asia	1,17E+00	2,00E+00	1,00E+00	2,48E+00	3,50E+00	1,42E+00	3,00E+00	3,61E+00	3,00E+00	3,81E+00	3,81E+00	3,28E+00
Barley	DK & EU	2,50E+01	2,50E+01	4,00E+01	2,50E+01	2,50E+01	2,50E+01	3,77E+01	2,00E+01	2,00E+01	1,00E+01	1,00E+01	2,00E+01
Wheat	DK & EU	4,65E+01	2,94E+01	3,00E+01	4,40E+01	3,04E+01	4,75E+01	2,81E+01	3,43E+01	4,51E+01	4,73E+01	4,73E+01	3,87E+01
Peas	DK	-	5,00E+00	-	-	9,35E-01	-	-	-	-	-	-	-
Fava bean	DK & EU	-	2,00E+01	-	-	2,00E+01	-	-	1,20E+01	-	-	-	-
Soy	SA & USA	2,10E+01	1,51E+01	1,53E+01	1,60E+01	8,00E+00	1,60E+01	3,00E+00	-	-	-	-	-
Monocalcium phosphate	EU-Finland	6,76E-01	6,33E-01	4,84E-01	1,04E+00	1,02E+00	7,35E-01	7,55E-01	7,82E-01	4,66E-01	7,62E-01	7,62E-01	3,98E-01
Salt	DK	5,06E-01	5,06E-01	4,68E-01	5,26E-01	5,23E-01	4,14E-01	2,64E-01	3,38E-01	2,92E-01	1,33E-01	1,33E-01	1,08E-01
Calcium carbonate	DK	1,45E+00	1,43E+00	1,70E+00	1,44E+00	1,40E+00	1,64E+00	9,27E-01	9,55E-01	1,23E+00	8,56E-01	8,56E-01	1,02E+00
Lysin	EU & WW	4,78E-01	2,79E-01	4,84E-01	6,13E-01	5,61E-01	6,79E-01	4,91E-01	5,38E-01	6,06E-01	2,30E-01	2,30E-01	3,29E-01
Methionin	EU & WW	8,30E-02	1,20E-01	5,10E-02	1,49E-01	1,82E-01	9,50E-02	8,40E-02	1,32E-01	5,80E-02	6,30E-02	6,30E-02	2,60E-02
Treonin	EU & WW	1,15E-01	9,90E-02	9,40E-02	1,56E-01	1,66E-01	1,31E-01	9,50E-02	1,22E-01	9,60E-02	-	-	3,00E-03
Tryptophan	EU & WW	-	5,50E-02	-	5,10E-02	1,22E-01	-	9,30E-02	1,38E-01	-	4,10E-02	4,10E-02	-
Multi Vitamin	DK & WW	2,00E-01	2,00E-01	2,00E-01	2,00E-01	2,00E-01	2,00E-01	-	-	-	-	-	-
Vitamin E	EU & WW	1,16E-01	1,16E-01	1,16E-01	1,60E-01	1,60E-01	1,60E-01	-	-	-	-	-	-
Ronozyme	EU & WW	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02	1,50E-02
Xylanase	EU & WW	5,00E-02	5,00E-02	5,00E-02	5,00E-02	0,050	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02	5,00E-02
Sunflower meal	Ukraine	2,65E+00	-	-	-	-	-	-	-	-	-	-	-
Valine	EU & WW	3,00E-03	-	-	2,50E-02	5,50E-02	-	-	2,50E-02	-	-	-	-
SCP/ Uniprotein	DK	-	-	5,00E+00	-	-	5,93E+00	-	-	6,87E+00	-	-	6,00E+00
Oats	DK	-	-	5,00E+00	-	-	-	-	-	-	-	-	-
Soy protein	SA & DK	-	-	-	8,13E+00	7,58E+00	-	-	-	-	-	-	-
Fishmeal	DK	-	-	-	-	9,80E-02	-	5,57E+00	3,18E+00	4,38E+00	6,46E+00	6,46E+00	6,60E+00
Potato protein 1	DK	-	-	-	-	-	-	-	2,00E+00	-	-	-	-
Hamlet Protein	DK	-	-	-	-	-	-	1,00E+01	1,00E+01	1,00E+01	1,00E+01	1,00E+01	5,69E+00
Potato protein 2	Netherlands	-	-	-	-	-	-	2,08E+00	-	-	2,50E+00	2,50E+00	-
Lactose powder 1	Poland	-	-	-	-	-	-	7,50E+00	7,50E+00	7,50E+00	1,00E+01	1,00E+01	1,00E+01
Vitamin and microminerals	DK, EU, WW	-	-	-	-	-	-	3,30E-01	3,30E-01	3,30E-01	3,30E-01	3,30E-01	3,30E-01
Rapeseed cake	DK	-	-	-	-	-	-	-	4,00E+00	-	-	-	-
Lactose Powder 2	DK	-	-	-	-	-	-	-	-	-	7,50E+00	7,50E+00	7,50E+00

DK= Denmark, EU=European Union, SA=South America, WW=World Wide, FM= Fresh Matter

¹ The number of digits does not reflect the precision; shown merely to facilitate the comparison between the different feed ingredients.

2.8 Assessing land use changes

Land use changes (LUC) were calculated for every crop product used in the piglet feed. Direct LUC was calculated using data on yield, fresh matter: dry matter ratio, cultivated: harvested: processed ratio etc. The avoided LUC were calculated based on the changing supply of a relevant marginal crop / product with the future increase of a certain other crop/ product. E.g. the extra crop ingredients needed for the preparation of the compound feed will be taken from actual Danish cropland, thereby resulting one other crop grown today to be displaced. Such a displaced crop is, in consequential LCA, referred to as the marginal crop. In this study the following were regarded as marginal to the respective crop or crop co product (Tonini et. al., 2012).

- For palm meal which is a co product of palm oil – soy as the marginal protein and maize as the marginal carbohydrate.
- For soy oil which is a co product of soy meal – palm oil as the marginal oil and soy as the marginal protein and maize as the marginal carbohydrate for palm meal.
- For sunflower oil which is a co product of sunflower meal – palm oil as the marginal oil and soy as the marginal protein and maize as the marginal carbohydrate for palm meal.

These were based on Schmidt, 2007 and Dalgaard et. al., 2008 and are examples for environmental consequences of direct land use changes (dLUC). 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, Saxe et. al., 2014.). 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 (Saxe et. al., 2014). 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 (Saxe et. al., 2014). 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 (Saxe et. al., 2014).

3. Results (or LCI)

Table 2 presents the global warming potential results in kg of CO₂ eq. per FU, for all 12 scenarios of piglet feed.

Table 2: Global warming potential results for piglet feed in 12 scenarios (data from Simapro simulations) in kg of CO₂ eq. per FU¹

kg CO ₂ Equivalent per FU	20-35kg			10-20kg			6-10kg			4-6kg		
	BAU	Local	SCP	BAU	Local	SCP	BAU	Local	SCP	BAU	Local	SCP
Barley	1,18E+02	1,18E+02	1,89E+02	1,18E+02	1,18E+02	1,18E+02	1,78E+02	9,46E+01	9,46E+01	4,73E+01	4,73E+01	9,46E+01
Wheat	2,07E+02	1,30E+02	1,34E+02	1,95E+02	1,35E+02	2,11E+02	1,25E+02	1,52E+02	2,01E+02	2,10E+02	2,10E+02	1,72E+02
Oat	-	-	3,11E+01	-	-	-	-	-	-	-	-	-
Soy bean meal	1,85E+02	1,33E+02	1,34E+02	2,12E+02	1,37E+02	1,41E+02	1,14E+02	8,79E+01	8,79E+01	8,79E+01	8,79E+01	5,00E+01
Sunflower meal	1,30E+01	-	-	-	-	-	-	-	-	-	-	-
Palm oil	8,96E+01	1,53E+02	7,66E+01	1,90E+02	2,68E+02	1,09E+02	2,30E+02	2,76E+02	2,30E+02	2,91E+02	2,91E+02	2,51E+02
Peas	-	5,77E+01	-	-	1,08E+01	-	-	-	-	-	-	-
Fava bean	-	2,37E+02	-	-	2,37E+02	-	-	1,42E+02	-	-	-	-
Uniprotein	-	-	1,73E+02	-	-	2,06E+02	-	-	2,38E+02	-	-	2,08E+02
Monocalcium phosphate	1,04E+01	9,70E+00	7,41E+00	1,59E+01	1,56E+01	1,13E+01	1,16E+01	1,20E+01	7,14E+00	1,17E+01	1,17E+01	6,10E+00
Salt	6,10E-01	6,10E-01	5,65E-01	6,34E-01	6,31E-01	4,99E-01	3,18E-01	4,08E-01	3,52E-01	1,60E-01	1,60E-01	1,30E-01
Calcium carbonate	6,19E-01	6,10E-01	7,28E-01	6,17E-01	5,99E-01	7,01E-01	3,96E-01	4,08E-01	5,28E-01	3,66E-01	3,66E-01	4,34E-01
Multi Vitamin	1,01E+01	1,01E+01	1,01E+01	1,01E+01	1,01E+01	1,01E+01	-	-	-	-	-	-
Vitamin E	5,83E+00	5,83E+00	5,83E+00	8,04E+00	8,04E+00	8,04E+00	-	-	-	-	-	-
Ronozyme & Xylanase	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00	3,38E+00
Vitamin and microminerals	-	-	-	-	-	-	1,66E+01	1,66E+01	1,66E+01	1,66E+01	1,66E+01	1,66E+01
Lysin	2,01E+01	1,17E+01	2,03E+01	2,57E+01	2,36E+01	2,85E+01	2,06E+01	2,26E+01	2,54E+01	9,66E+00	9,66E+00	1,38E+01
Treonin	4,83E+00	4,16E+00	3,95E+00	6,55E+00	6,97E+00	5,50E+00	3,99E+00	5,12E+00	4,03E+00	1,72E+00	1,72E+00	1,26E-01
Tryptophan	-	2,31E+00	-	2,14E+00	5,12E+00	-	3,90E+00	5,79E+00	-	-	-	-
Valine	1,26E-01	-	-	1,05E+00	2,31E+00	-	-	1,05E+00	-	-	-	-
Methionine	2,59E+00	3,75E+00	1,59E+00	4,66E+00	5,69E+00	2,97E+00	2,63E+00	4,13E+00	1,81E+00	1,97E+00	1,97E+00	8,13E-01
Fishmeal	-	-	-	-	1,34E+00	-	7,62E+01	4,34E+01	5,99E+01	8,84E+01	8,84E+01	9,02E+01
Lactose powder	-	-	-	-	-	-	3,27E+01	3,27E+01	3,27E+01	7,63E+01	7,63E+01	7,63E+01
Rape seed meal	-	-	-	-	-	-	-	1,85E+01	-	-	-	-
Potato protein	-	-	-	-	-	-	6,05E+02	5,81E+02	-	7,26E+02	7,26E+02	-
iLUC	8,34E+02	8,13E+02	6,87E+02	7,01E+02	7,51E+02	6,20E+02	2,71E+02	3,02E+02	5,36E+02	1,14E+02	1,14E+02	4,49E+02
Avoided barley	-3,48E+01	-2,53E+01	-3,37E+01	-4,93E+01	-5,28E+01	-4,73E+01	-2,36E+02	-2,24E+02	-3,46E+01	-2,52E+02	-2,52E+02	-3,46E+01
Avoided soy	-7,56E-01	-1,29E+00	-6,46E-01	-1,60E+00	-2,26E+00	-9,18E-01	-1,94E+00	-2,33E+00	-1,94E+00	-2,46E+00	-2,46E+00	-2,12E+00
Avoided palm	-1,33E+02	-7,32E+01	-7,39E+01	-1,17E+02	-7,53E+01	-7,73E+01	-6,28E+01	-1,26E+02	-4,83E+01	-4,83E+01	-4,83E+01	-2,75E+01
Manure management	3,49E+02	3,69E+02	3,47E+02	3,68E+02	3,80E+02	3,74E+02	3,96E+02	4,04E+02	4,16E+02	4,24E+02	4,24E+02	4,31E+02
Net value	1,68E+03	1,68E+03	1,72E+03	1,70E+03	1,99E+03	1,72E+03	1,79E+03	1,85E+03	1,45E+03	1,81E+03	1,81E+03	1,37E+03

¹ The number of digits does not reflect the precision; shown merely to facilitate the comparison between the different feed ingredients.

4. Discussion (or LCIA)

It is evident from the results from Table 2 that the scenario with the highest global warming potential varies from each piglet group. Although all the scenarios in general have very large amounts of CO₂ emissions and have negative impacts on the environment, in each weight category there are changes in the emissions of the three main scenarios.

However all 12 scenarios have negative emissions or positive impacts on the environment when considering the avoided palm meal, avoided barley meal and avoided soy meal.

When considering the 20-35 kg group, the best scenario based on the net values, is the BAU, next is the SCP and the last or the worst scenario is the local crop. The difference between the best and the worst scenarios (between BAU and local crop) is 280 kg of CO₂ eq. per FU.

In the 10-20 kg group, the best scenario based on the net values is the BAU, next is the SCP and the last or the worst scenario is the local crop. The difference between the best and the worst scenarios (between BAU and local crop) is 290 kg of CO₂ eq. per FU.

Considering the 6-10 kg group, the best scenario based on the net values is the SCP, next is the BAU and the last or the worst scenario is the local crop. The difference between the best and the worst scenarios (between SCP and local crop) is 400 kg of CO₂ eq. per FU.

In the 4-6 kg group, the best scenario based on the net values is the SCP and the worst scenario is the BAU and the local crop. The difference between the best and the worst scenarios (between SCP and BAU or local crop) is 440 kg of CO₂ eq. per FU.

The components having the largest influences within the categories are iLUC, raw manure spreading on land, Uniprotein, fava bean and potato protein. The high value for iLUC would be due to the reason that some crop materials are imported to Denmark from as far as South America. Uniprotein gives a considerable amount of emissions due to the fact that it uses a large amount of electricity in production of Oxygen (in liquid form which is imported) which is not used in the other categories. Fava bean too has a high amount of emissions and it is quite significant as it is used only in the local crop scenario. It is the same with potato protein since production of protein from potato is a by-process of the regular production of potato starch from potato and costs a large amount of electricity for the extraction of proteins.

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) (FAO, 2007 and Saxe et. al., 2014).

5. Conclusions (or Interpretation)

LCA was applied to three protein supply strategies in producing feed for piglets of four different piglet categories and thereby Environmental Impacts were calculated for a total of 12 scenarios.

The following can be concluded from the results of this research.

- Introducing a new protein source affects the whole feed and not just the protein ingredient. This cascading effect is the key to assess the environmental performance of a certain process or product, in this case the SCP.

- In the global warming potential impact category, BAU scenario is the best for larger categories of piglets in terms of having the lowest CO₂ emissions as well as a certain amount of avoided emissions which would be offset against the positive values.

- In the two larger weight categories local protein scenario has the worst environmental impact.

- SCP shows promising results specially in the two larger weight categories and it could be further improved with changing the electricity usage to 100% renewable energy sources and also by improving the efficiency of the SCP production by using biogas instead of natural gas in the production process.
- The combination of each piglet category with the protein sources gives variant results in terms of environmental impact and they should be considered case by case to find out the best scenario.

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285. LCA results of French cereal and oilseed feedstuffs with effect of different agricultural practices

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ABSTRACT

The French project ECOALIM aims to improve the environmental impacts of husbandries by optimizing their feed. This project defines the environmental impacts of the production of raw materials for animal feeding and optimizes the formulation of feed compounds with environmental constraints in order to improve environmental footprint of animal products. The project covers different farming systems and production areas in France and is based on life cycle assessment (LCA). The objective of this study was to develop a dataset of environmental impacts of feed ingredients, while taking into account agricultural practices and processing.

The LCA results obtained for rapeseed grown in France, with different agricultural practices, at the field gate, are discussed here, while similar results are also available for other crops (wheat, barley, maize, sunflower, pea...). Several life cycle inventories (LCI) were carried out: national average data representative of France (average French rapeseed) and LCIs with different crop managements based on case studies, in different regions of France, with different crop rotations (rapeseed in crop rotation with introduction of intermediate crops, or with introduction of legumes, or with organic fertilization, or rapeseed cultivated with an associated crop). The LCA methodology was applied, considering environmental burdens at the rotation system scale. Focus is made this paper on five impact indicators (Climate Change ILCD, Cumulative Energy Demand non renewable fossil+nuclear, ACidification ILCD, EUtrophication CML, Consumption of Phosphorus).

For the rapeseed LCIs presented, the main contributor to selected environmental impacts was field emissions. The assessment of practices such as organic fertilization, the introduction of intermediate crops, the introduction of legumes in the rotation or an associated crop within rapeseed crop showed some improvements on impacts. But results were very variable between the different case studies.

Keywords: rotation, rapeseed, organic fertilization, intermediate crop, associated crop

1. Introduction

Animal feeding contributes very significantly (up to 80 %) to the overall environmental impact of animal products (meat, milk) assessed by Life Cycle Assessment (LCA). But, the current feed formulation only takes into account economical and nutritional constraints. The French project ECOALIM aims to improve the environmental impacts of husbandries by optimizing their feed. This will lead to select different feedstuffs in the formulation, or to change their ways of production (crop managements, transformation processes). To perform such optimization, LCA data concerning the feedstuffs are needed. It should be then integrated to advice tools for stakeholders to help them in reducing the environmental impact of feed while still taking into account economic and social aspects.

A previous study highlighted the need for homogeneously developed data on environmental impacts of feed ingredients to formulate low-impact feeds and investigate mitigation options (Nguyen et al., 2012). The database constructed within the project ECOALIM gathers the most accurate and representative data on Life Cycle Inventories (LCI) to date for French feed ingredients, with 149 average feed ingredients (non-processed and processed ingredients, several perimeters: field gate, storage agency gate, plant gate and harbor gate) and 16 feed ingredients from specific crop itineraries (Wilfart et al., 2016).

In this paper, we focused on five LCIs concerning rapeseed at the field gate: the inventory for average French rapeseed obtained from statistical data, and four inventories constructed with different agricultural practices in order to evaluate their ability to reduce the environmental impacts of that crop. The studied agricultural practices were implemented at two different levels:

- at the level of the rapeseed crop : “associated crop” which means that winter oilseed rape is grown with an intercropping frost-sensitive crop (preferentially legume crop). This agroecological solution impacts directly the rapeseed crop management,
- at the level of the crop rotation: rapeseed crop is included in a cropping system and some agricultural practices implemented at this level may have an effect on the environmental

impact of each crop in that system. The three studied agricultural practices at this level were: organic fertilization (OF), systematic introduction of cover crop (SCC), introduction of protein crops in the rotation (legume crops as pea or soybean or faba bean, PC).

Similar LCA results, with the effect of agricultural practices, are available for other crops (wheat, barley, maize, sunflower ...) in the ECOALIM database. Other LCA perimeter where also, used, at the mill gate for instance, for rapeseed meal, with five different rapeseed meal LCIs coming from the five different rapeseed LCIs at the field gate (national average + 4 agricultural practices). Yet the LCA results obtained for rapeseed grown in France, at the field gate, with different agricultural practices, are discussed here.

2. Methods

The Life cycle inventory for French rapeseed at national level was based on the similar inventory included in the Agribalyse[®] database (French national database of the main agricultural products; Koch and Salou, 2015), with updated emission factors for ammonia (EMEP/EEA, 2013). Within this ECOALIM project, one major methodological difference compared to Agribalyse[®] was the distribution of the phosphorus fertilizer inputs and nitrate emissions among the different crops involved in the same crop rotation (according to crop requirements and removal for phosphorus, and equally among crops for nitrate leaching).

The LCA perimeter used for rapeseed at the field gate starts from “cradle” which takes into consideration the production of the inputs needed to produce the feed ingredient (seeds, fertilizers, energy, materials, ...), and it ends at field gate. The functional unit for data collection was 1 ha, while the functional unit for LCIA was 1 kg. Data were collected over the period 2006-2012. The background data (electricity, transport) came from the Ecoinvent V3.1 database. All produced data were checked and validated by an LCA expert committee.

Focus is made this paper on five impact indicators: Climate Change ILCD (kg CO₂ eq), Cumulative Energy Demand non renewable fossil+nuclear (MJ), ACidification ILCD (mol H⁺ eq), EUtrophication CML (kg PO₄³⁻ eq), and Consumption of Phosphorus (kg P).

3. Life Cycle Inventory

For the French rapeseed crop (**RS-Fr**), the national average inventory was obtained from statistical data adjusted with expert judgments. This inventory data were based on the period 2006-2010, but data were not collected each year (most data were calculated as the mean of three Terres Inovia’s surveys respectively conducted in 2008, 2009 and 2010).

For the “associated crop” rapeseed (**RS-AC**), the inventory was based on the RS-Fr inventory, with modifications related the intercropping frost-sensitive legume crop: additional seeds (faba bean seeds), decrease of the nitrogen fertilizer amount (- 30 kg N/ha), and less herbicide treatments (- 33% herbicide active substances and - 1 tractor with plant protection sprayer). These inventory data are presented in table 1, and were supported by experiments (Cadoux et al., 2015).

Table 1: Main inputs used and yield for average French rapeseed and “associated crop” rapeseed

Crop	Yield (9% H ₂ O)	N mineral	N manure	P ₂ O ₅	K ₂ O	Seeds	Pesticide active ingredient	Diesel	Agricultural machinery
	kg/ha	kg/ha	kg/ha	kg/ha	kg/ha	kg/ha	kg/ha	kg/ha	kg/ha
RS-Fr	3243	161.6	17	48	28	2.5	1.98	78.3	8.2
RS-AC	3243	131.6	17	48	28	2.5 rapeseed + 50 faba bean	1.43	77.4	8.1

For the other three rapeseed LCIs, with agricultural practices introduced at the crop rotation level, introduction of systematic cover crop (**RS-SCC**), organic fertilization (**RS-OF**), introduction of protein crop (**RS-PC**), the method was quite different, and not based on statistical data. Hence, different LCI for soft winter wheat, barley, grain maize, oilseed rape and sunflower were carried out

based on 12 case studies, representative of different farming systems (characterized by different pedoclimatic conditions, yield potential, choice of the crops in the rotation), distributed in different regions of France. These case studies have been defined at the rotation scale by regional agronomists to be representative of the main farming systems in each main production region. Among these case studies, seven ones included rapeseed and were located in the East of France (rotation rapeseed-wheat-barley), in the North of France (with one short rotation rapeseed-wheat-barley, and another longer rotation rapeseed-wheat-sugar beet-wheat-barley), in the Centre of France (rotation rapeseed-wheat-barley), in the West of France (rotation rapeseed-durum wheat- soft wheat-sunflower-soft wheat-barley), in North-West of France (rotation rapeseed-soft wheat-sugar beet-soft wheat) and in the South of France (rotation rapeseed- soft wheat-sunflower- durum wheat). The implementation of the scenarios (SCC, OF, PC) was consistently adapted to each case study, depending on the regional agronomic possibilities, with regional expert judgments. The choices and aftermath on the LCI, concerning each scenario introduced into the case studies were:

- Introduction of systematic cover crop: letting volunteer rapeseed plants after rapeseed harvest was considered as a cover crop; sowing a combination of cruciferous and legume seeds between a cereal harvest and a spring crop sowing was another intermediate cover crop. This agricultural practice led to a decrease of nitrate leaching (-50% nitrate leaching after rapeseed harvest when there are volunteer rapeseed plants) and to a reduction of the amount of nitrogen fertilizer applied on the following crop. Seeds for intermediate crops and destroying the cover crop residues (with shredder or herbicide) were included in the inventory. The reduction of nitrate leaching was shared between all the crops in the rotation.

- Organic fertilization: the amounts, the choice of the organic fertilizer types (manure, slurry...) and their transport depended on each case study, and were adapted to the regional context. A rule, limiting the organic fertilizer inputs, was set up in order to avoid excessive phosphorus inputs at the rotation scale. This agricultural practice led to a reduction in mineral nitrogen fertilizer inputs, and to different results in the calculation of ammonia and nitrogen oxides emissions.

- Introduction of legume crop in the rotation: the choice of protein crop was adapted to the regional context. We considered spring pea in the case studies with rapeseed (other cases without rapeseed introduced soybean crop). This agricultural practice had a positive effect on the following crop (mainly cereal), as it allowed a lower nitrogen fertilizer input.

To produce for instance the **RS-SCC** inventory, in each regional case study, we calculated the effect of this agricultural practice (systematic introduction of cover crop in the rotation) by creating an LCI with +1 kg of rapeseed with systematic cover crop in one case study and -1 kg of rapeseed from basic scenario (rapeseed in the same case study without any of the three studied agricultural practices). Then, in order to produce an average inventory of the effect of SCC at the level of France, a ponderation of the effect of SCC in individual case studies was made, according to the yield of rapeseed crop in each case, and to the surface that each case study represents in each region (Jouy and Wissocq, 2011). Finally, the RS-SCC inventory was composed by the RS-Fr inventory for 1 kg of product + 1 kg of the average effect of SCC. So, the RS-SCC, RS-OF and RS-PC inventories had the same yield 3243 kg /ha at 9% moisture than RS-Fr and RS-AC.

4. Life Cycle Impact Assessment

The LCIA results are presented in table 2, while a comparison is showed in figure 1.

Table 2: LCIA results for 5 rapeseed crop at field gate, according to various agricultural practices

Impact	unit		RS-Fr	RS-SCC	RS-OF	RS-PC	RS-AC
Climate change ILCD	kg CO ₂ eq	CC	0,927	0,919	0,883	0,926	0,820
CED 1.8 non renewable fossil+nuclear	MJ	CED	5,537	5,537	4,786	5,512	4,944
Acidification ILCD	molc H ⁺ eq	AC	0,0210	0,0210	0,0194	0,0210	0,0190
Eutrophication CML baseline	kg PO ₄ ³⁻ eq	EU	0,00755	0,00651	0,00746	0,00772	0,00729
Phosphore consumption	kg P	P	0,00735	0,00735	0,00143	0,00679	0,00730

The assessment of practices such as organic fertilization, the introduction of intermediate cover crops, the introduction of protein crops in the rotation and an associated crop within rapeseed crop showed some improvements on impacts (figure 1). First, Climate change (CC) and Acidification (AC) were mostly reduced with RS-AC (-12% and -10%), as it allowed a direct reduction of the amount of mineral nitrogen fertilizer in the rapeseed crop. Then, Cumulative Energy Demand non renewable fossil+nuclear (CED) was mainly reduced with RS-OF (-14%) and RS-AC (-11%) for the same reason. Eutrophication (EU) was mostly reduced with RS-SCC (-14%) as nitrate leaching was reduced in those rotations with intermediate cover crops. Finally, the consumption of phosphorus resources (kg P) was dramatically reduced down to 19% with RS-OF (-81%), as most of phosphorus supply necessary in the rotation for crop growth was brought by manure. RS-PC had a weak effect on indicators for rapeseed crop, whereas this agricultural practice mainly benefited the following crop after protein crop (always a cereal) with reduced nitrogen fertilization. So the introduction of legume crop in the rotation did not benefit directly rapeseed crop but proved to be a beneficial practice at a rotation scale (Nemecek, 2015).

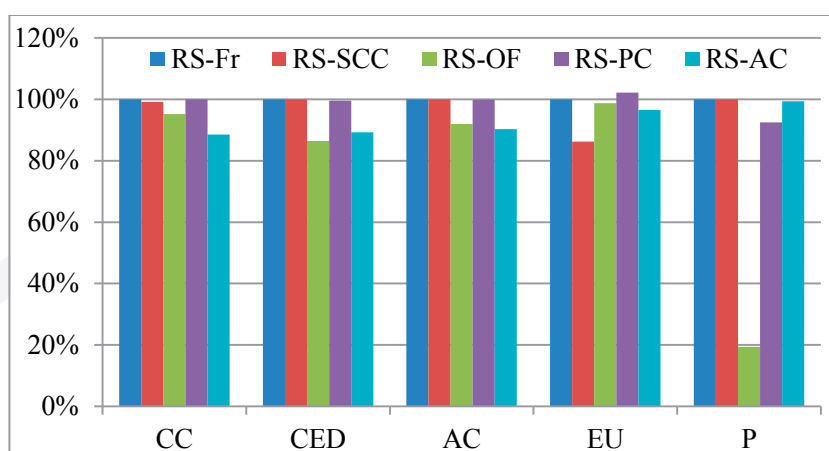


Figure 1: comparison of LCA results of rapeseed with various agricultural practices.

RS-Fr= Rapeseed, conventional, average France. RS-SCC= Rapeseed with systematic cover crop in the crop rotation. RS-OF= Rapeseed with high level of organic fertilization in the crop rotation. RS-PC= Rapeseed with insertion of a protein crop in the crop rotation. RS-AC= Rapeseed with an associated crop (frost-sensitive legume crop) with the rapeseed crop

In figure 2, only RS-Fr and RS-AC LCA results were presented, to evaluate the main contributors to the five studied impacts.

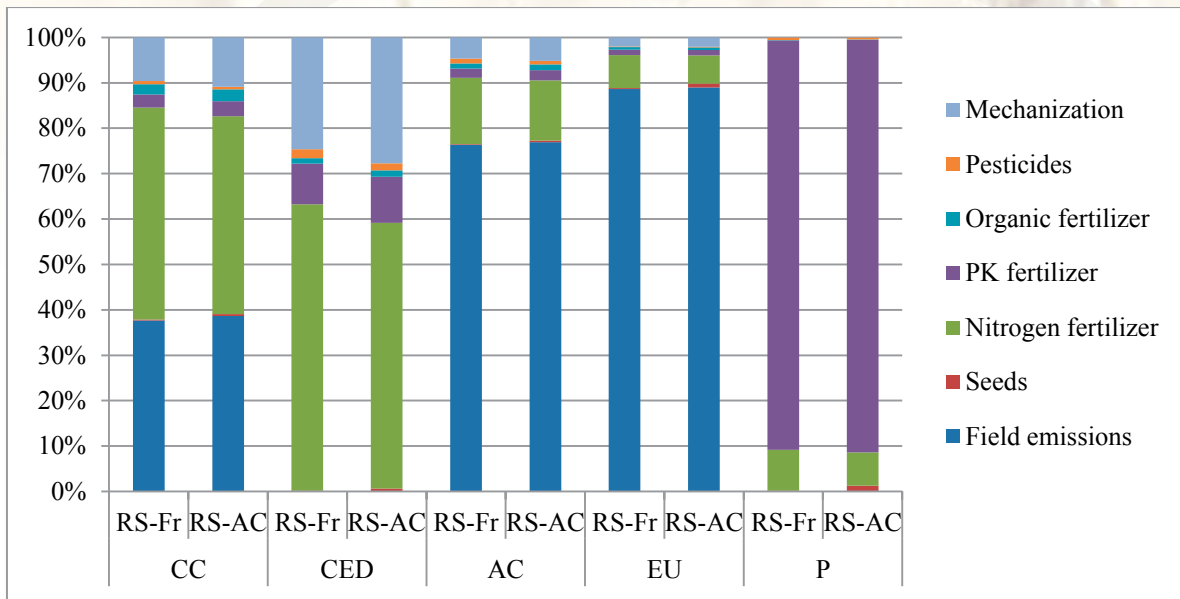


Figure 2: distribution of impacts for Average French rapeseed, and “associated crop” rapeseed

For rapeseed crop, the main contributor to acidification impact was field emissions (of which 91% ammonia and 9% nitric oxides) and to eutrophication impact too (of which 56% nitrate, 26% ammonia, 8% phosphorus, 5% nitrous oxide, 4% nitric oxides). The field emissions were also an important contributor to climate change impact, mainly due to nitrous oxide emissions, which are linked to nitrogen fertilization. For climate change, the other important contributor was the production of nitrogen fertilizer supplied to rapeseed crop. This nitrogen fertilizer is a main contributor to CED impact, as its production is energy consuming, while mechanization (including agricultural machinery, diesel production and diesel combustion) was also an important contributor. The consumption of phosphorus, as a non renewable natural resource, was due obviously to the input of mineral phosphorus fertilizer on the crop. The RS-Fr and RS-AC repartitions were almost identical.

But the results were very variable between the different case studies (figure 3), as the introduction of agricultural practices was adapted to each regional context. For instance, the organic fertilization was a stronger lever to reduce the climate change impact, in the case studies in the Centre of France and in the East. This was because, in these two regions, manure is rarely used on arable crops, so the effect is higher than for a case study like West of France where organic fertilization is already present in the basic scenario. So, the inventories for agricultural practices, based on seven case studies for rapeseed, gave highly variable LCIA results, which made comparison difficult. This is to be added to the variability of primary data.

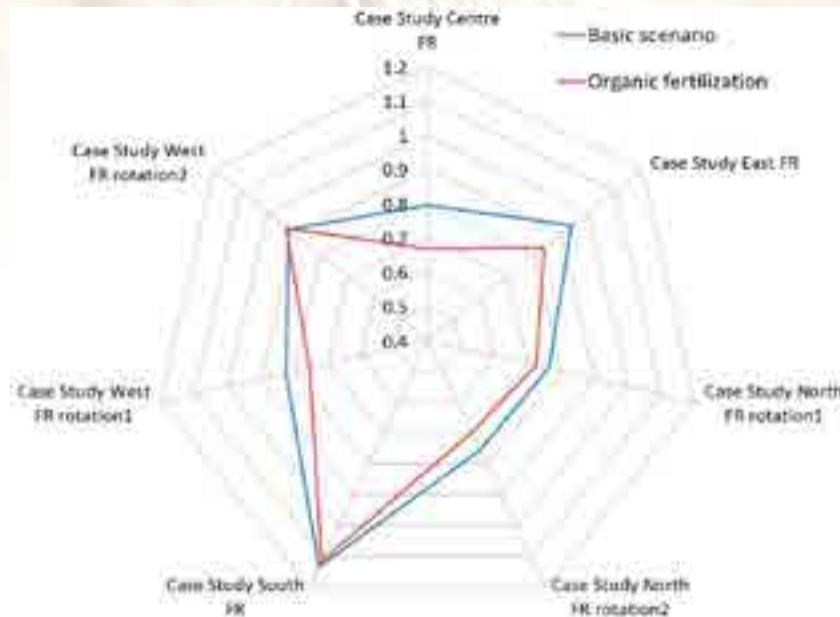


Figure 3: Climate change results of rapeseed at the field gate (g CO₂-e/kg) for seven case studies (in different French regions), and two different agricultural practices (Basic scenario versus High level of organic fertilization in the crop rotation scenario).

4. Conclusions

The assessment of practices such as organic fertilization, the introduction of intermediate crops, the introduction of legumes in the rotation or an associated crop within rapeseed crop showed some improvements on impacts. The agricultural practices studied here had beneficial effects by lowering mineral nitrogen or phosphorus fertilizers, or by lowering emissions such as nitrate or nitrous oxide. A combination of several agronomic levers, adapted to each context could be a way to lower environmental impacts more efficiently. Indeed, the introduction of intermediate crop reduced the eutrophication impact, while the associated crop (intercropping frost-sensitive legume crop within rapeseed crop), and organic fertilization both reduced Energy consumption, Climate change and Acidification impact for rapeseed crop. Organic fertilization had a strong effect on consumption of natural phosphorous resources. The introduction of legumes in the rotation didn't benefit rapeseed crop directly, but proved to be an interesting practice at a rotation level. Other agricultural practices could be further studied, with a beneficial effect on impact like ecotoxicity or human toxicity.

Finally, these results represent a step forward to share with the agricultural sector in order to promote good farming practices and environmental assessment. Thus, this should encourage the first processing and the feeding industries to promote better agricultural practices.

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59. Towards the global reference for Feed LCA data: the Global Feed LCA Institute

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ABSTRACT

The feed and livestock sector need to measure its impact on the environment in order to identify mitigation options. Harmonised methodology and robust data are required for that purpose. The Global Feed LCA Institute (GFLI) is a feed industry initiative with the vision to develop a freely and publicly available feed LCA database and tool. This is seen as a key input to support environmental assessment of livestock products. This paper provides an overview of how the Global Feed LCA Institute intends to achieve its objectives and the challenges it has to face. It describes the governance mechanism which has been put in place by the GFLI in order to achieve global geographical coverage but also focuses on the methodological aspects associated with the development of the GFLI database.

Keywords: animal feed, environmental footprint, database, global consistency, free access,

1. Introduction

Environmental footprinting of livestock products is a challenging but essential task to improve the accuracy of reporting on the real impacts of livestock products. This includes both understanding where the livestock chains in terms of impact and encouraging the benchmarking and measurement of both individual and collective reduction efforts. According to the Food and Agriculture Organisation of the United Nations (FAO), feed production represents 45% of the carbon footprint of livestock products globally. This share is even higher for monogastric animals. This shows that animal feed is an important part of the chain and it is essential that feed operators are able to understand their impact, not only from a business efficiency perspective but also to meet the expectations of their customers and public bodies, at national and international level. This also shows that high quality secondary data for feed is a critical requirement to achieve high quality environmental assessment of animal products. Considering the complexity of the feed supply chains it is indeed almost impossible to assess the environmental impact of feed products by relying only on primary data. The nature of the feed supply chains is global, which means that the methods and data need to be global as well.

To this end, the Global Feed Life Cycle Assessment Institute (GFLI) is a feed industry initiative with the vision to develop a freely and publicly available feed LCA database and tool. The GFLI's objective is to support meaningful LCAs of livestock products and enable to benchmark feed industry environmental impacts. The GFLI was officially launched in January 2016. The GFLI is seen as long term organisation, with a first set of deliverables scheduled for the end of 2017. The GFLI will implement the methodology developed by the Livestock Environmental Assessment and Performance partnership (LEAP) led by the FAO.

This paper provides an overview of how the Global Feed LCA Institute intends to achieve its objectives and the challenges it has to face. The first section of the paper describes the governance mechanism which has been put in place by the GFLI in order to achieve global geographical coverage. The second sections looks at the methodological aspects associated with the development of the GFLI database whereas the third section provides some insights on how the deliverables of the GFLI will be disseminated.

2. Governance approach

Developing a truly global database is an ambitious and challenging objective. This task cannot be performed by a single organisation, since it would not be able to provide all the necessary expertise and knowledge. The pressure to be active in the area of environmental footprinting, be it an institutional pressure or a market pressure varies also significantly across the different world regions. The GFLI should therefore enable pioneers and front runners to get involved as soon as possible but also stimulate stakeholders or world regions which are currently less interested or less concerned by the GFLI activities to join. It is indeed important to avoid creating a gap between the different regions of the world for the development of the GFLI database. In addition, the different stakeholders and the different regions of the world may have different expectations regarding the GFLI database, in particular relative to the level of detail it should provide. This element should also be factored in the GFLI governance mechanism.

To deal with these challenges, the GFLI set up an innovative governance mechanism based on regional projects, which will populate the common database. These regional projects are led by a consortium of feed companies and feed associations which constitute a Project Steering Group. It is necessary to join a Project Steering Group, i.e. to contribute to a regional project, in order to become a member of the GFLI. Joining a regional project implies adherence to the GFLI vision and objectives mentioned in the GFLI framework agreement signed by all members. Instead of a regional approach, a Project Steering Group can also be based on a sector, with particular needs in terms of data. This is for example the case for the fish feed sector, for which discussions have started to develop a specific project within the GFLI. The GFLI activities can therefore be split in two categories:

- The common activities, which benefit to all members: establishing the database infrastructure, developing the procedure for data collection and compiling the data collected by the regional projects in the common database
- The regional (or sectoral) activities, i.e. the collection of data for a given world region or sector

The different Project Steering Groups are responsible for financing the regional or sectoral activities, but each member of the different Project Steering Groups also contributes financially to the common activities of the GFLI. The Project Steering Groups are also autonomous to determine their internal governance rules.

Within the GFLI, all the Project Steering Groups follow the same methodology and the same procedure for data collection (see section 3) which ensures global consistency. There are currently two operational Project Steering Group: one for Europe and one for North America (USA and Canada). They are composed of the following members, which are either feed companies or feed associations:

- Europe Project Steering Group: AB AGRI, Agrifirm, AIC, Bemefa, Cargill Animal Nutrition, Evonik, FEDIOL, FEFAC, For Farmers, Nevedi, SNIA, Nutreco, USSEC
- USA/Canada Project Steering Group: AFIA, ANAC, ADM Alliance Nutrition, Alltech, Cargill Animal Nutrition, CHS Nutrition, Diamond V, Elanco Animal Health, Evonik, Hi-Pro Feeds, LandO'Lakes Purina, Mosaic Feed Ingredients, National Soybean Board, Potashcorp

Discussions are currently taking place to launch regional activities in China later in 2016, but also in Brazil and Australia.

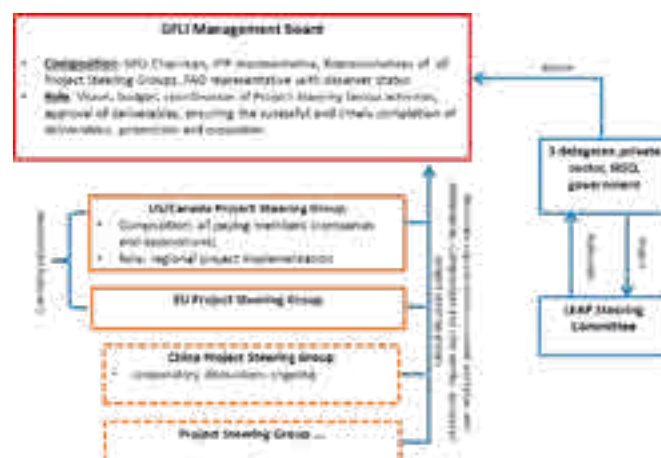
The executive body of the GFLI is the GFLI Management Board, which oversees and coordinates the activities of the Project Steering Groups.

It is up to the different project Steering Group to determine the list of feed ingredients for which data will be collected and brought to the GLFI. This has to be done from the perspective of compound feed production, which means that imported feed ingredients and the related logistics have also to be taken into consideration. (e.g. soybean meal for EU compound feed production). Dealing with imported feed ingredients may lead to an overlap between regional projects. To ensure synergy between the regional projects the following principles apply:

- Data collected by a regional project for processes within their regional scope, takes priority over data collected by other regional projects. For instance, a South American project is in the lead for deriving soy cultivation data in Brazil. Other regions that would like to use data of soybean from Brazil shall use the data that are brought in by the South American project.
- However, when no data is collected for a specific region (i.e. a region has not yet joined the GFLI project), other regions may collect data for that region. For example, the European project may include soybean cultivation from Brazil in lieu of data collected in a South American project.
- If a region starts collecting data of feed ingredients in scope for its consumption, it should first determine if data that are already collected by other regions overlap with the scope of their data collection.

Coordination is necessary between the activities of the regional projects. This role is facilitated by the GFLI Management Board, which is executive body of the GFLI. The main responsibilities of the GFLI Management Board are to implement the GFLI vision, to monitor the budget, to coordinate the Project Steering Groups activities, to ensure a successful and timely completion of the deliverables and to promote and expand the GFLI. The GFLI Management Board is composed of representatives of the regional projects, a chairperson which does not represent any project and a representative of the International Feed Industry Federation (IFIF). The LEAP Partnership is also sitting in the GFLI Management Board with an observer status, to facilitate alignment with the LEAP recommendations. Within the GFLI Management Board, decisions are based on consensus. The governance of the GFLI is summarized in the figure below.

Figure 1: GFLI governance mechanism



3. Methodological approach

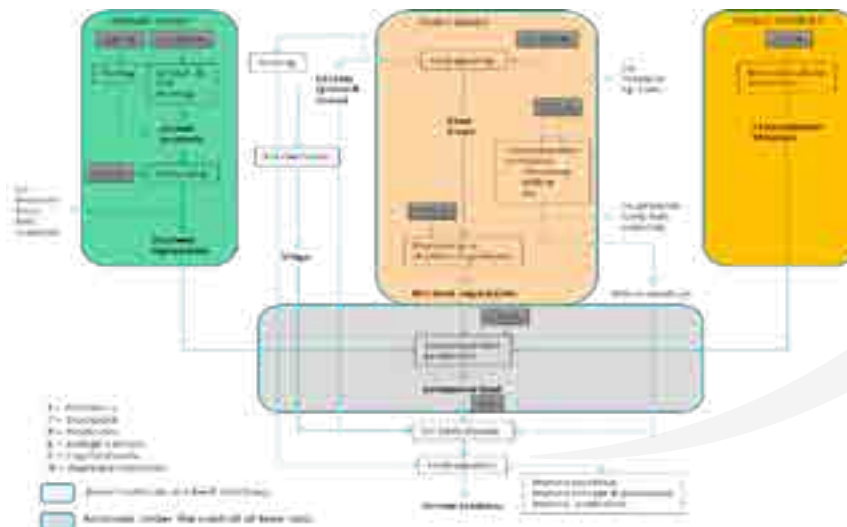
From a methodological perspective, developing a global and consistent database is another challenge, especially when the task of collecting data is split across different groups of stakeholders which also use different sources of information. The consistency of the data should be addressed at two levels: the consistency of the datasets themselves, but also the consistency for the definition of the scope of the activities of the regional projects, i.e. the list of datasets that the regional projects will work on.

a. Consistency of datasets

For the consistency of the datasets, the GFLI mainly relies on the methodological recommendations provided by LEAP in its guidelines for the assessment of the environmental performance of animal feeds supply chains.

The GFLI activities are conceived as a module to support the environmental assessment of animal products. The GFLI database will therefore follow a cradle to gate approach. The system boundaries considered for the GFLI activities are described below. It includes the production of the different types of feed ingredients (from plant origin, from animal origin and from non-agricultural sources) as well as the feed-mill related activities (processing of feed ingredients and feed delivery to the farm).

Figure 2: System boundaries of the GFLI database



The LEAP guidelines provide detailed information regarding the activity data which shall be collected as part of the life cycle inventory for the following life cycle stages: cultivation (including land use change and assimilating animal production and fisheries to cultivation); processing of feed materials, compound feed production and preparation of animals' ration. The decision tree provided in the LEAP Guidelines can also be used for allocation. Following the LEAP recommendations, the GFLI will consider economic allocation as the baseline option (especially for the processing of feed materials). The GFLI database will however include three types of allocation (economic, energy and mass) in order to facilitate sensitivity assessment and to ensure that the influence of allocation in the results is properly taken into account.

Being able to rely on the GFLI database in the context of the Environmental Footprint (EF) initiative launched in 2013 by the European Commission is another objective of the Global Feed LCA Institute. As far as the methodology is concerned, this means that the GFLI database will support the impact assessment methods currently listed in the Product Environmental Footprint (PEF) Guide, as a baseline (this objective triggers also consequences on the infrastructure of the database and the underlying documentation). It is

however possible for regional or sectoral projects to consider additional impact assessment methods which may be more relevant for a specific or regional context.

As already mentioned, the GFLI is an ongoing project. It means that several methodological aspects still need to be clarified. This is particularly the case for the identification of proxies, in case of data gaps but also to achieve a harmonised yet simple way of assessing the quality of the data brought by the regional projects to the GFLI. The GFLI is assessing the opportunity and feasibility to use the Data Quality Rating system currently employed in the EF pilot phase as starting point for data quality evaluation.

b. Consistency among regional projects

The GFLI governance mechanism offers some useful flexibility but is rather decentralized. The inherent risk with this approach is to end up with inconsistent choices made by the regional projects. To reduce this risk, the GFLI has defined a road map which shall be followed by the regional projects for their activities

Table 1: GFLI road map for data collection

Step 1	Define the list of feed ingredients which are relevant in order to perform a meaningful assessment of the environmental impact of compound feed for the region/sector at stake.
Step 2	Literature review and methodological compliance check
Step 3	Define data collection and update plan
Step 4	Identify synergies / overlap with other regional projects
Step 6	Start collection/update of LCI
Step 7	Organize an external review of the collected/updated data to check compliance with data collection procedure
Step 8	Submit data to the GFLI management Board

A consistent approach to define the scope of the regional projects is also extremely important for the objectives of the GFLI. The scope of the regional project are defined from the perspective of a compound feed producer in a given region, meaning that the list of feed ingredients for which data will be collected should enable meaningful environmental assessments. To achieve this consistency, the GFLI has come with another stepwise approach which supports the decisions to be taken by the regional project.

- Step 1: divide the list of ingredients into consistent groups of ingredients.

This grouping of ingredients is applicable for all regional projects. For each of the groups, a mass threshold relative to consumption by the compound feed industry has been defined and all the feed ingredients which contribute to this threshold (in decreasing order) shall be included in the scope of the regional projects. With this approach, it becomes mandatory for regional projects to collect data for a broad range of feed ingredients, which increases the possibility to conduct meaningful assessments. The groups and the respective thresholds are listed in table 2.

Table 2: list of groups of feed ingredients and mass threshold to be considered by regional projects to define the scope of their data collection activities

Group	Threshold (mass)	Group	Threshold (mass)
Cereals (unprocessed except drying)	95%	Products of animal origin	50%
Root products (tapioca, sugar beet, etc...): processed and unprocessed	90%	Dried forages	50%
Processed cereals products	90%	Pulses	80%
Vegetable meals	95%	Minerals	50%
Vegetable oils	95%	Additives (vitamins, enzymes, amino-acids)	50%
Other co-products from the food and fuel industries	80%	Other	Voluntary threshold

The example below illustrates how this grouping and threshold approach can be implemented in practice. Based on public statistics but also on expert judgement, the consumption of cereals by the EU compound feed industry is estimated as follows.

Table 3: Breakdown of cereals consumption by the EU compound feed industry (source: FEFAC)

Cereal type:	Share in feed sub-group cereals:	Cereal type:	Share in feed sub-group cereals:
Common wheat	31%	Rye	2%
Maize	30%	Sorghum	1%
Barley	25%	Durum wheat	1%
Triticale	5%	Other cereals	2%
Oats	3%	Total	100%

The threshold defined for cereals in table 2 is 95%. It means that life cycle inventory data shall be collected for wheat, maize, barley, triticale, oats and rye.

- Step 2: identify animal species that are important for the region/stakeholders in the regional project and identify feed ingredients specific to these species.

A regional project may decide that a certain animal species is particularly relevant. If it does not represent significant volume of compound feed production and if it requires specific feed ingredients; these ingredients will not be necessarily identified in step 1. The regional project can then decide to include these ingredients in its scope, based on expert judgement. An example is the veal production using feed ingredients of dairy origin which represents a small share of the ingredient consumed by the compound feed industry.

- Step 3: identify other missing ingredients.

If needed, the regional project board may decide to employ additional criteria to add even more ingredients to their list. These can for example be ingredients that currently do not have a large share in feed formulations but are foreseen to become more dominant in the near future. Also feed ingredients that have a small share but seem to be environmentally beneficial and could be used to improve current feeds may be included. Conversely, ingredients that have a very small share, but are regarded to be environmentally harmful (based on previous experience or information from other regions), may be included.

- Step 4: subdivide the list of ingredients into ingredient/origin combinations.

The region for which a dataset will be defined and implemented in the GFLI database can be large. In many situations this region consists of smaller units such as countries, states or provinces, etc. The appropriate geographic level of detail depends on the relevance of having detailed data available on the basis of expected contribution in overall LCA results and potential variability. The appropriate level of geographic granularity is not necessarily the same for all feed ingredients. Furthermore it is important to gain understanding of how the production of a feed ingredient is regionally distributed over its different life cycle stages. This is illustrated by the figure 3 which maps cultivation of corn and production of ethanol in the United States.

Figure 3: corn production and ethanol processing in the United States (source: USDA)



Such an analysis can be used to focus further on specific sub-regions but also to determine average transport scenarios (distances and means). The subdivision can be thus be based on for example ingredient-country combinations (soybean meal crushed in the Netherlands), but could also be on a state level (almond hulls, California), depending on what data resolution is available. For each identified ingredient, the specific regional coverage of the ingredient country combinations should be as large as possible (>90%). Origins contributing more than 5% shall also be included. The analysis can be done on the basis of import/export and consumption statistics (sources could be the UN COMTRADE database, FAOstat, PRODCOM EU statistics, trade.gov or other regional/national statistics) or other sources of information (reports, scientific literature for a specific ingredient).

- Step 5: remove ingredients for which it will practically infeasible to collect data.

At this stage, the list of feed ingredients may be quite long. The possibility is therefore given to regional projects to remove ingredients from the list to cope with constraints such as in case lack of access to data, or budget limitations.

4. Dissemination and use

The ambition of the GFLI is to become a reference for Feed LCI data. It means that in addition to the database development, the GFLI should also consider how to facilitate access to the data. One key element from that perspective is the GFLI decision to offer free access to its database. Another important element is LCA software neutrality. The GFLI objective is to develop its database in a format that can easily be picked up by the main LCA software developers. To this end, the GFLI is monitoring closely the UNEP/SETAC initiative on

interoperability of LCA databases. While waiting for the outcomes of this initiative, the GFLI will consider the ILCD format developed by the European Commission as an interim solution. With this approach, the GFLI database is an embedded database in the main LCA software, as described in figure 4.

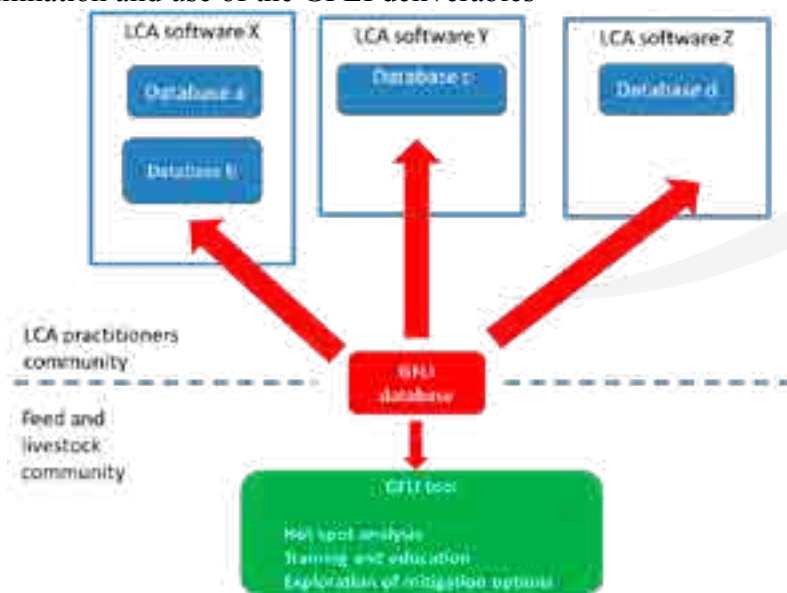
Besides facilitating uptake by practitioners, the GFLI policy is also to collaborate with existing initiatives and to establish synergies. This would reduce the costs associated with the development of the database and increase its acceptance at the same time. To illustrate this, discussions have already started between the GFLI and the French public authorities for enabling access to the French database Agribalyse through the GFLI. Considering the USA/Canada regional project, establishing a link with the LCA Digital Commons developed by the USDA would also be extremely useful.

The other aspect to consider for the dissemination of the GFLI deliverables is how to deal with the feed and livestock community. Despite the importance of environmental footprinting for the compound feed industry, very few feed companies can rely on in-house LCA expertise. The purpose of the GFLI tool is to provide access to the GFLI database for non-LCA experts. The GFLI tool is therefore targeted at the feed and livestock community and will support the following actions:

- Hot spot analysis
- Training and education
- Exploration of mitigation options

This twin-track approach aims at facilitating the use of the GFLI deliverables.

Figure 4: dissemination and use of the GFLI deliverables



5. Conclusion

The Global Feed LCA Institute is a rather new initiative which takes place in a changing landscape as far as the development of LCA databases is concerned. To deliver according to its objectives, the GFLI has set up an innovative governance mechanism which relies on regional project delivering data in a consistent and harmonised manner.

The GFLI is conceived as a long term organisation. The initial phase of the GFLI has started in a project mode, with the financial support of the feed companies and associations which have decided to become GFLI members. Relying on front runners to start an activity is a

quite common mechanism. The EU Project Steering Group will deliver its data to the GFLI by the end of 2016. The USA/Canada Project Steering will deliver its data to the GFLI by the end of 2017. In order to become a sustainable organisation, the GFLI needs however to identify the relevant mechanism to maintain and update the outcomes of the initial project phase, without relying permanently on front runners. From that perspective, being able to recruit new members and engage activities in other regions is a key factor of success.

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174. Environmental consequences of agricultural products and co-products diversion from animal feed to bioenergy production

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With recent bioenergy development throughout the world, many agricultural products, by-products and lands are being diverted from their initial use to serve as inputs for bioenergy production. In this context, the present paper aims at evaluating the environmental consequences of agricultural products and co-products diversion from animal feed to biogas and bioethanol production.

To meet this goal, this study uses consequential LCAs to identify feed ingredients required to replace bioenergy inputs in animal feed. Two case studies are considered: (1) a bioethanol facility fed with forage wheat grains and producing, aside from ethanol, distillers' grains with solubles (DGS), a co-product used in dairy and pig feeding, and (2) a biogas plant fed with industrial by-products (potentially) initially used as animal feed. In the case of the bioethanol facility, grains processed into bioethanol need to be replaced by other ingredients in animal feed while DGS displaces other ingredients in animal feed. In the case of the biogas plant, industrial by-products (silage maize, sugar beet tails, downgraded potatoes and cereal middlings) fed to the biogas plant need to be replaced by other ingredients in animal feed.

The originality of the present method lies in the fact that a given feed ingredient does not just replace another single feed ingredient, considering the huge number of parameters to account for in the design of balanced diets (energy, proteins, digestibility, fibers, etc.). Instead, whole diets are replaced by other diets, all of them being equivalent with respect to animal feeding requirements. Balanced diets for fattening beefs, dairy cows and pigs, representative from the Walloon market (Belgium), *i.e.* composed with the most commonly used ingredients, are used to calculate which feed ingredients are required or replaced as a consequence of bioenergy production.

Results show that induced impacts are mostly due to additional land required to produce crops replacing diverted by-products originally fed to animals, enlightening the urge to use as a priority bioenergy inputs not usable as animal feed.

138. Environmental consequences of using alternative protein sources in pig diets

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ABSTRACT

Feed production is responsible for the majority of the environmental impact of livestock production, especially for monogastric animals, such as pigs. Several studies demonstrated that replacing soybean meal (SBM) with alternative protein sources, such as locally produced peas or rapeseed meal, potentially reduces the environmental impact of pork production. These studies, however, used an attributional life cycle assessment (ALCA), which solely addresses the direct environmental impact of a product. A replacement of SBM with alternative protein sources, however, can also have indirect environmental consequences, e.g. impacts related to replacing the original function of the alternative protein source. Accounting for indirect environmental consequences in a consequential life cycle assessment (CLCA) might change environmental benefits of using alternative protein sources. This study aims to explore differences in results when performing an ALCA and a CLCA to reduce the environmental impact of pig production. We illustrated this for two case studies: replacing SBM with rapeseed meal (RSM), and replacing SBM with waste-fed larvae meal in diets of finishing-pigs. We used an ALCA and CLCA to assess global warming potential (GWP), energy use (EU) and land use (LU) of replacing SBM with RSM and waste-fed larvae meal, for finishing-pigs. The functional unit was one kg of body weight gain. Based on an ALCA, replacing SBM with RSM showed that GWP hardly changed (3%), EU hardly changed (1%), but LU was decreased (14%). ALCA results for replacing SBM with waste-fed larvae meal showed that EU hardly changed (1%), but GWP (10%) and LU (56%) were decreased. Based on a CLCA, replacing SBM with RSM showed an increased GWP (15%), EU (12%), and LU (10%). Replacing SBM with waste-fed larvae meal showed an increased GWP (60%) and EU (89%), but LU (70%) was decreased. The CLCA results were contradictory compared with the ALCA results. CLCA results for both case studies showed that using co-products and waste-fed larvae meal not reduces the net environmental impact of pork production. This would have been overlooked when results were only based on ALCA.

Keywords: consequential life cycle assessment, attributional LCA, pigs, insects, soybean meal, rapeseed meal, feed optimization

1. Introduction

The global demand for animal source food (ASF) is expected to increase. Simultaneously, livestock production causes severe environmental pressure via emissions to air, water, and soil (Steinfeld et al., 2006). The global livestock sector is responsible for about 15% of the total anthropogenic emissions of greenhouse gases (Gerber et al., 2013). The sector also increasingly competes for scarce resources, such as land, water, and fossil-energy.

Feed production is responsible for the largest part of the environmental impact of livestock production, especially for monogastric production systems (De Vries and De Boer 2010; Gerber et al., 2013). To reduce the environmental impact of feed production a widely applied mitigation is to replace feed ingredients with a high environmental impact for ingredients with a lower environmental impact. Studies, exploring the environmental impact of different feed ingredients, demonstrated that diets containing soybean meal (SBM) often result in a large environmental impact (Cederberg and Flysjo 2004; Eriksson et al., 2005; Van der Werf et al., 2005; Weightman et al., 2011). Currently, SBM is the main protein source in pig diets (Vellinga et al., 2009). Cultivation of SBM has a high environmental impact; due to large transport distances; due to a high economic allocation as SBM nowadays drives the production process (Cederberg and Flysjo 2004; Van der Werf et al., 2005; Vellinga et al., 2009); and due to emissions related to land use change (LUC), such as deforestation in South America (Foley et al., 2007; Prudêncio da Silva et al., 2010).

Several studies have assessed the environmental impact of replacing SBM with: 1) locally produced protein (e.g. peas), 2) co-products from the bio-diesel industry (e.g. rapeseed meal (RSM)), or 3) novel protein sources (e.g. waste-fed larvae meal) (Eriksson et al., 2005; Meul et al., 2012; Sasu-Boakye et al., 2014; Van Zanten et al., 2015a; Van Zanten et al., 2015b). Replacing SBM with locally produced protein sources resulted in a reduction of global warming potential (GWP) up to 13% and of land use (LU) up to 11% (Eriksson et al., 2005; Meul et al., 2012; Sasu-Boakye et al., 2014). Replacing SBM with RSM resulted in a reduction of LU up to 12%, but did not change GWP and energy use (EU) (Van Zanten et al., 2015b). Replacing SBM with waste-fed larvae meal in diets of

finishing-pigs is not analysed so far, but preliminary results of Van Zanten et al. (2015a) showed waste-fed larvae meal has potential to reduce the environmental impact of feed production.

Replacing SBM with alternative protein sources, therefore, might be a potential mitigation strategy. Those studies, however, were based on an attributional life cycle assessment (ALCA), implying they addressed the direct environmental impact of a product in a status-quo situation. The direct environmental impacts result from the use of resources and emissions of pollutants directly related to the production of one kg of pig meat, such as feed use, diesel for transport, and electricity for heating, at a specific moment in time. Although commonly used in pig production (McAuliffe et al., 2016), an ALCA does not include consequences of a change in diet composition, outside the production chain of pork.

These indirect environmental consequences are related to changes in use of farm inputs or its outputs. Van Zanten et al. (2015a), for example, explored the environmental impact of using waste-fed larvae meal as livestock feed. These larvae were partly fed on food-waste, which was originally used for production of bio-energy. Accounting for these indirect consequences implied including the environmental impact of production of energy needed to replace the original bio-energy function of food-waste in the analysis. Applying this indirect environmental impact assessment method, called consequential LCA (CLCA), might have different outcomes.

To our knowledge there are only a few studies that assessed the difference in results between an ALCA and a CLCA, when assessing the environmental impact of changing livestock diets (Mogensen et al., 2010; Nguyen et al., 2013), none of them relates to pig production. This study, therefore, aims to explore the differences in results when performing an ALCA and a CLCA to reduce the environmental impact of pig production. We illustrated this comparison with two case studies: replacing SBM with RSM, and replacing SBM with waste-fed larvae meal in diets of finishing-pigs. In this study, GWP, EU, and LU were assessed per kg of body weight gain.

2. Methods

An ALCA and a CLCA will be used to assess the environmental impact of replacing SBM with RSM or with waste-fed larvae meal. For this purpose, we first describe the diet formulation and growth performance of finishing-pigs (2.1), and subsequently explain the environmental impact assessment methods used (2.2).

2.1 Diet composition and growth performance

All diets were designed to meet the requirements of a Dutch average standard diet for finishing-pigs, and contained 9.50 MJ net energy (NE) and 7.59 g standard ileal digestible (SID) lysine per kg of feed, while pigs were fed ad libitum. The basic scenario (S1) was defined, using SBM as major protein source (see Table 1), based on Van Zanten et al. (2015b). In the second scenario (S2), SBM was replaced with RSM based on their crude protein (CP) content, as described by Van Zanten et al. (2015b). In the last diet (S3), 15% SBM was replaced with 15% waste-fed larvae meal also based on their CP content. In table 1 the diet compositions of the three scenarios are given. For a more detailed description of the diet composition please see the Appendix. As the nutrient content of the diet in each scenario was identical (9.50 MJ NE/kg feed and 7.59 g lysine/kg feed), and no adverse effect of pig performance were found by including RSM (McDonnell et al., 2010) or waste-fed larvae meal (Makkar et al., 2014) in finishing-pig diets, a similar growth performance was assumed between the three scenarios. Growth performance was based on Van Zanten et al. (2015b), who calculated the growth performance of finishing-pigs for S1 and S2. Scenarios started with 100 days, weight at start 45 kg, final age 180 days, and total feed use 183 kg. The final body weight of the growing-pigs was 116.4 kg (Van Zanten et al., 2015b).

Table 1: Diet composition of scenario 1 (S1) containing SBM, scenario 2 (S2) containing RSM, and scenario 3 (S3) containing larvae meal.

Ingredients	S1	S2	S3
Rapeseed meal, CP <380	-	23.00	-
Soybean meal, CP<480	15.00	-	-
Waste-fed larvae meal	-	-	15.00
Peas	9.36	10.00	-
Maize	30.00	30.00	30.00
Wheat	29.74	30.24	24.29
Wheat middlings	0.90	-	26.57
Barley	10.10	-	-
Sugarcane molasses	2.00	2.00	2.00
Vit. and min. premix	0.40	0.40	0.40
Phytase premix	0.65	0.65	0.65
Animal fat	-	2.09	-
Limestone	1.24	0.96	1.10
Salt	0.37	0.29	0.26
Monocalcium phosphate	0.11	0.01	-
Sodium bicarbonaat	-	0.09	0.15
L-Lysine HCL	0.10	0.22	0.03
L-Tryptophan	-	0.01	-
L-Threonine	-	0.02	-
DL-Methionine	0.03	0.01	-
Nutrient content g/kg			
Nett energy, MJ	9.5	9.5	9.5
Crude protein	162	160	166
Lysine (SID)	7.59	7.59	7.59
Crude fibre	30	47	45
Crude fat	27	50	60
Phosphorus (P)	3.75	4.65	5.31
Digestible P	2.27	2.27	2.27

2.2 Life cycle assessment

To assess the environmental impact of each scenario, a life cycle assessment (LCA) was used. LCA is an internationally accepted and standardized method (ISO14040, 1997; ISO14041, 1998; ISO14042, 2000; ISO14043, 2000) to evaluate the environmental impact of a product during its entire life cycle (Guinée et al., 2002; Bauman and Tillman, 2004). During the life cycle of a product two types of environmental impacts are considered: emissions of pollutants and use of resources, such as land or fossil-fuels (Guinée et al., 2002). We assessed GHG emissions, EU, and LU. These impacts were chosen because the livestock sector contributes significantly to both LU and climate change worldwide (Steinfeld et al., 2006a). Furthermore, EU was used as it influences GWP considerably. LU was expressed in m² per year, whereas EU was expressed in MJ. The following GHGs were included: carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O). These GHGs were summed up based on their equivalence factors in terms of CO₂ (100 years' time horizon): i.e. carbon dioxide (CO₂), biogenic methane (CH₄, bio): 28 kg CO₂-eq/kg, fossil methane (CH₄, fossil): 30 kg CO₂-eq/kg; and nitrous oxide (N₂O): 265 kg CO₂-eq/kg (Myhre et al., 2013). In this study only the environmental impact related to feed production is assessed because no changes are expected on related emissions of piglet production (rearing), enteric fermentation from pigs, and from pig housing. Changes from manure management can be expected but no data is available on related emissions of manure management when insects are used as feed.

As stated in the introduction two types of LCA exist: ALCA and CLCA. Both methods are explained below.

Attributional LCA

An ALCA describes the environmentally relevant physical flows to and from all processes, in the life cycle of a product, at one specific moment in time. During the life cycle of a product, like pork, multifunctional processes occur. A multifunctional process (also referred as product-packages) is an activity that fulfils more than one function (Ekvall and Weidema, 2004), yielding two or more products: the determining product, which determines the production volume of that process (e.g. rapeseed oil), and a co-product (e.g. rapeseed meal; Weidema et al., 2009). In case of a multifunctional process, most ALCA studies of livestock products partition the environmental impact of the process to the various products based on their relative economic values, a method called economic allocation (De Vries and De Boer, 2010). In our ALCA, we used economic allocation to divide the environmental impact between the determining product and the co-product.

To assess the environmental impact of the three scenarios, the environmental impact of each ingredient must be known. GWP, EU, and LU of most feed ingredient were based on Vellinga et al. (2013). Production of feed ingredients included impacts from cultivation (e.g. impacts related to the production and use of fertilizers, pesticides, machinery, and energy), impacts from drying/processing, and impacts from transport to the farm. GWP, EU, and LU related to waste-fed larvae meal were based on Van Zanten et al. (2015a). LU and EU values of feed additives (salt, chalk, vitamins and minerals, phytase, monocalcium phosphate, and amino acids) were based on Garcia-Launay et al. (2014) (GWP was based on Vellinga et al. (2013)). Appendix Table A.4 provides an overview of GWP, LU, and EU per kg of feed ingredient. To assess the average impact of one kg feed, the environmental impact per kg feed ingredient was multiplied by its relative use in the diet. Next, for each scenario, the average environmental impact per kg feed was multiplied with the total feed intake during the finishing period and divided by the growth performance during the finishing period ($116.4 \text{ kg} - 45 \text{ kg} = 71.4 \text{ kg}$). The functional unit was one kg weight gain.

Consequential LCA

A CLCA describes how environmental flows change in response to a change in the system (Ekvall and Weidema, 2004). Only those processes (within and outside the system) that respond to the change, are considered. Considering changes is especially important when a mitigation strategy includes the use of co-products or food-waste. This is because the production volume is restricted for co-products and food-waste. For co-products, for example, a change in demand of the determining product (e.g. sugar) directly affects the production volume of the co-product (e.g. beet pulp) (Weidema et al., 2009), whereas a change in demand of co-product does not. Due to this, co-products are limited available. Increasing the use of co-products in animal feed, therefore, results in a reduction of co-product use in another sector necessitating substitution (see Appendix for more information). We define the sector which was using the co-product before, the previous user.

Within CLCA, system expansion is generally used to handle multifunctional processes. System expansion implies that you include changes in the environmental impact of the alternative production process for which the co-product could be used, into your analysis (Ekvall and Finnveden, 2001). Van Zanten et al., (2014) developed a theoretical framework to assess the environmental consequences of using co-products in livestock feed. This framework provides assistance in how to assess the environmental impact of changing the application of a co-product. In this study, the theoretical framework of Van Zanten et al. (2014) was used to assess the environmental consequences of replacing SBM with RSM or with waste-fed larvae meal. Based on this framework the net environmental impact was calculated. The net environmental impact depends on the environmental benefits minus the environmental costs. The environmental benefits are determined by the decrease in environmental impact related to the product that was replaced with co-products or food-waste. The environmental costs are determined by the increased environmental impact related to the marginal product which is replacing the co-product used by the previous user (the product that replaces the 'old' application of the co-product or food-waste).

To assess the consequences of replacing SBM with RSM or waste-fed larvae meal in the diet of finishing-pigs, the following steps were needed.

First, the difference in feed ingredients between the diet in S2 and S3, and the reference scenario (S1) was determined, by subtracting all feed ingredients used in S1, from those in S2 and in S3 (see Table 3). Table 3 shows which feed ingredients changed compared with the basic scenario containing SBM, for example, replacing SMB with RSM, i.e. resulted in an increase in RSM of 23%, a decrease of 15% SBM etcetera. In case a feed ingredient is used in the same amount, such as maize, it is not considered as it does not result in an environmental change.

Second, the environmental impact of a change in each feed ingredient was determined (Table 3). The computation of this environmental impact differed depending on the feed ingredient being: a determining product without a co-product (so no product-package); or a determining product with a co-product; or a co-product. In case a feed ingredient is a determining product without a co-product, the environmental impact of cultivation can be fully ascribed to this single product. The environmental impact of the cultivation of peas, for instance, is fully allocated to the only product, namely peas. The environmental impact of determining products without a co-product, therefore, could be based on ALCA data of Vellinga et al. (2013). This is reasonable because when a product is not part of a product-package, the environmental impact related to cultivation does not have to be allocated between products. Similar to an ALCA, therefore, the environmental impact related to cultivation is fully ascribed to this single product. We, therefore, assumed no differences in environmental impact between the ALCA and the CLCA for a determining product without a co-product.

In case a feed ingredient is part of a product-package, the environmental consequences related to the ingredient had to be calculated. In case SBM was replaced with RSM, the following products were part of a product-package: SBM (i.e. a determining product with oil as co-product), and the co-products wheat middlings, RSM, and animal fat. In case SMB was replaced with waste-fed larvae meal, the following products were part of a product-package: SBM and the co-products wheat middlings, and waste-fed larvae meal. The indirect environmental consequences related to waste-fed larvae meal were based on Van Zanten et al. (2015a). For the other four products - RSM, animal fat, wheat middlings, and SBM - the indirect environmental impact was calculated. Below we explain the method used in detail for one ingredient, namely RSM. Please consult Appendix to see the description of the method to calculate the indirect environmental impact of the other three ingredients.

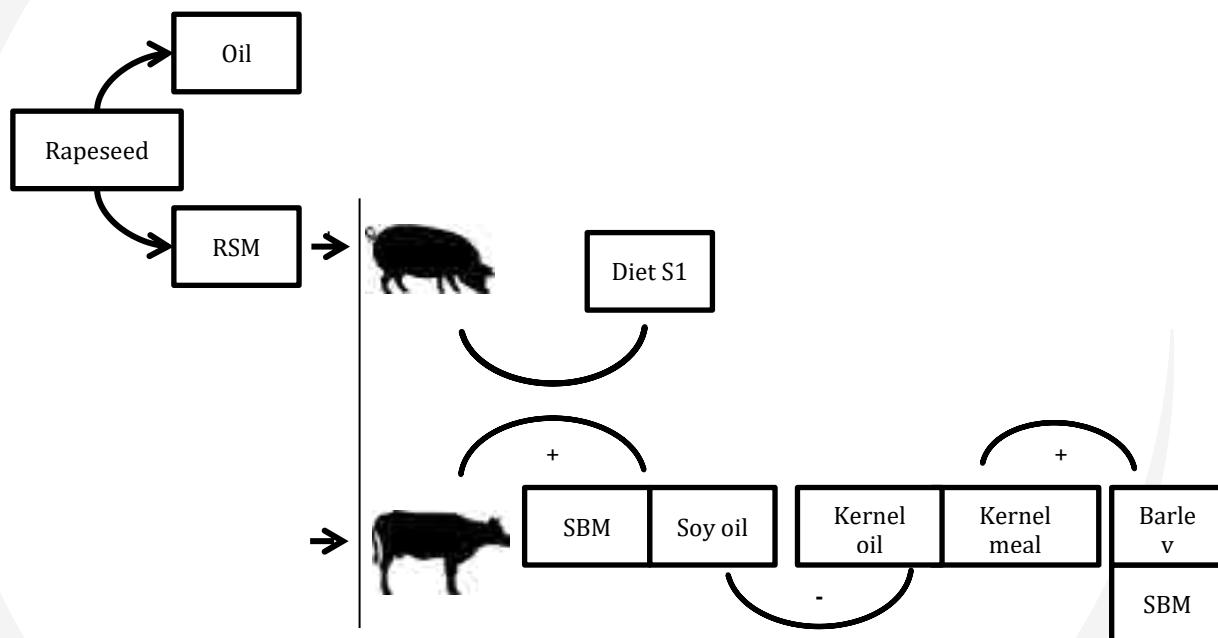


Figure 1: Principle to assess the environmental consequences of rapeseed meal (RSM) based on Van Zanten et al. (2014).

Figure 1 illustrates how to assess the environmental consequences of RSM. RSM is a co-product from the bio-diesel industry, and does not drive the production process. An increased use of RSM in diets of finishing-pigs, therefore, results in a reduction of the original applications of RSM. We assumed that RSM was originally used in diets of dairy cows (the previous user). Increasing the use of RSM in pig diets, therefore, resulted in a decreased use of RSM in diets of dairy cow. RSM in diets of dairy cows, therefore, was replaced (or also often called displaced) with the marginal product, which we assumed to be SBM (Weidema, 2003). Replacing RSM with SBM in diets of dairy cows was based on net energy for lactation, as this was the limiting nutritional factor of SBM. An increased production of SBM also results in an increased production of soy-oil, the depended co-product. The increased production of soy-oil was assumed to replace the marginal oil, being palm-oil (Dalgaard et al., 2008; Schmidt et al., 2015). A reduction in production of palm-oil, however, also implies a reduction in production of palm kernel meal. A reduction of palm kernel meal resulted in an increased use of the marginal meal SBM. Replacing palm kernel meal with SBM was based on their energy and protein content, as suggested by Dalgaard et al. (2008). The reduction of 19 g of palm kernel meal, therefore, was replaced with 3 g SBM and 15 g barley. Barley is assumed to be the marginal feed grain (Weidema, 2003). Thus, the amount of CP and energy in palm kernel meal is equal to the total amount of CP and energy in SBM and barley.

Table 2: Global Warming Potential (GWP) expressed in g CO₂-eq per kg of final diet, energy use (EU) expressed in MJ per kg of final diet, and land use (LU) expressed in m² per kg of final diet of replacing soybean meal (S1) with rapeseed meal (S2) and replacing soybean meal (S1) with waste-fed larvae meal (S3) in pig diets.

	<i>S1 - S2</i>	GWP	EU	LU	<i>S1 - S3</i>	GWP	EU	LU
	<i>g/kg</i>	CO ₂ -eq	MJ	m ²	<i>g/kg</i>	CO ₂ -eq	MJ	m ²
Rapeseed meal	230.0	289	2.4	4.83	0.0	-	-	-
Soybean meal	150.0	267	2.5	0.57	-150.0	267	2.5	0.57
Larvae meal	0.0	-	-	-	150.0	3068	36.5	0.07
Peas	6.4	741	6.6	5.71	-93.6	741	6.6	5.71
Maize	0.0	-	-	-	0.0	-	-	-
Wheat	5.0	378	3.0	1.14	-54.5	378	3.0	1.14
Wheat middlings	-9.0	387	3.4	1.07	256.7	387	3.4	1.07
Barley	101.0	379	2.9	1.28	-101.0	379	2.9	1.28
Sugarcane molasse	0.0	-	-	-	0.0	-	-	-
Premix	0.0	-	-	-	-2.0	4999	0.8	0.00
Phytase premix	0.0	-	-	-	-2.0	4999	26.0	0.15
Fat from animals	20.9	4828	21.53	1.76	0.0	-	-	-
Chalk	-2.8	19	0.00	0.00	-1.4	19	0.0	0.00
Salt	-0.8	180	3.50	0.00	-1.1	180	3.5	0.00
Monocalcium- Phosphate	-1.0	4999	18.4	0.32	-1.1	4999	18.4	0.32
Bicarbonaat	0.9	180	3.9	0.00	1.5	180	3.9	0.00
L-Lysine HCL	1.2	6030	119.9	2.27	-0.7	6030	119.9	2.27
L-Threonine	0.2	16978	119.9	2.27	0.0	-	-	-
DL-Methionine	-0.2	5490	89.3	0.01	-0.3	5490	89.3	0.01

3. Results

Using an ALCA approach, S1 resulted in 1.62 kg CO₂-eq, 14.01 MJ, and 4.81 m².yr per kg weight gain, S2 in 1.67 kg CO₂-eq, 14.11 MJ, and 4.12 m².yr per kg weight gain, and S3 in 1.45 kg CO₂-eq, 14.17 MJ, and 2.14 m².yr per kg weight gain.

Replacing SBM (S1) with RSM (S2) based on an ALCA, therefore, hardly changed GWP and EU but it decreased LU per kg weight gain, implying that this strategy has no potential to reduce GWP and EU but has potential to reduce LU of pork production. Using a CLCA approach, this strategy resulted in an increase of 0.25 kg CO₂-eq, 1.61 MJ, and 0.48 m².yr per kg weight gain, yielding even less unambiguous results. Relative differences between the ALCA and CLCA approach are presented in Figure 3.

Replacing SBM (S1) with waste-fed larvae meal (S2) based on an ALCA approach resulted in a decreased GWP and LU and hardly changed EU, implying this strategy has potential to reduce GWP and LU but has no potential to reduce EU of pork production. Using a CLCA approach, this strategy resulted in an increase of 0.97 kg CO₂-eq, 12.51 MJ, and a reduction of 3.38 m².yr per kg weight gain, yielding less unambiguous results. Relative differences between the ALCA and CLCA approach are presented in Figure 4.

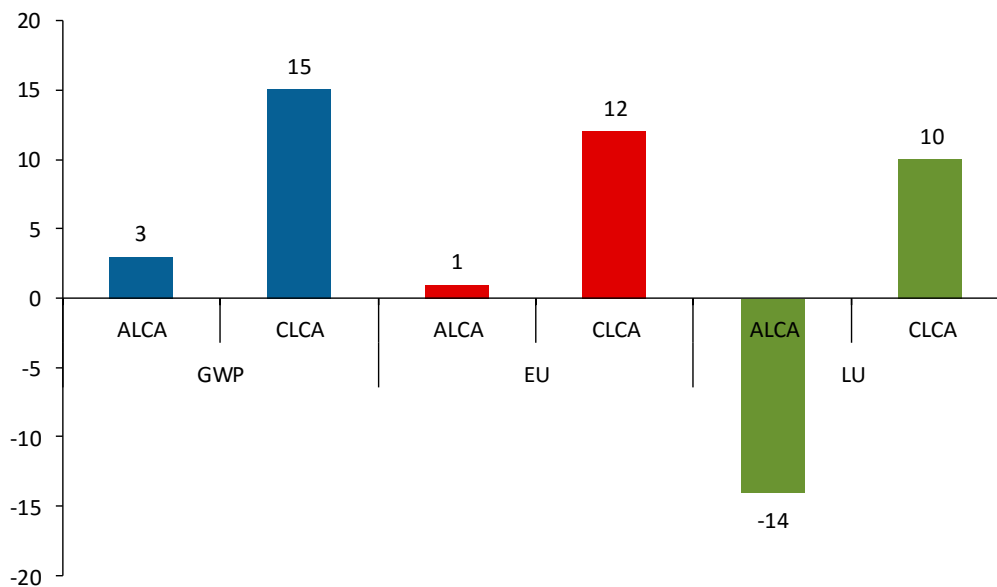


Figure 2: The environmental impact of replacing SBM with RSM in pig diets based the attributional LCA approach and the consequential LCA approach in %.

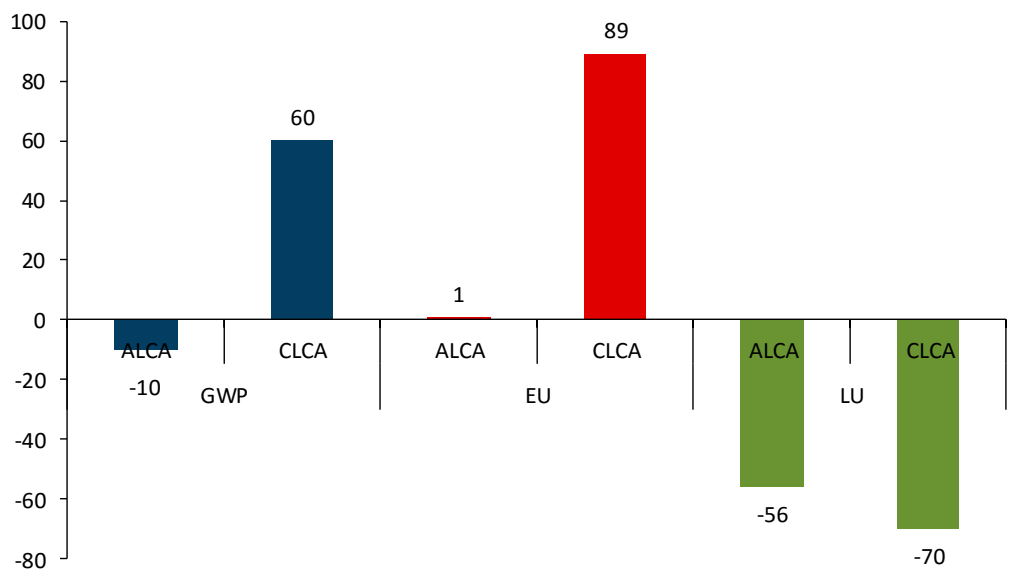


Figure 3: The environmental impact of replacing SBM with waste-fed larvae meal in pig diets based on the attributional LCA approach and the consequential LCA approach in %.

4. Discussion

Based on an ALCA, replacing SBM with RSM reduced LU, but hardly changed GWP and EU. Based on a CLCA, however, replacing SBM with RSM resulted in an increased GWP, EU, and LU. Differences in results between ALCA and CLCA were caused because the net environmental impact was increased. In S1 15% SBM and 8% barley was replaced with 23% RSM, 2% animal fat. As RSM and animal fat are both co-products, using them resulted in indirect environmental consequences. The increased impact (environmental costs) related to the consequences of using co-products (using RSM resulted in an increased use of SBM in diets of dairy cows, whereas using animal fat resulted in an increased use of palm oil in broiler diets) was higher compared to the reduction in impact (environmental benefits) due to decreasing SBM and barley in pig diets.

Based on an ALCA, replacing SBM with waste-fed larvae meal reduced GWP and LU, but hardly affected EU. Based on a CLCA replacing SBM with waste-fed larvae meal resulted in an increased GWP and EU but LU was still decreased. The difference in results between ALCA and CLCA were mainly caused by the high environmental impact of the waste-fed larvae meal. The environmental impact of waste-fed larvae meal was based on Van Zanten et al. (2015a). Larvae were fed partly on food-waste. Food-waste was originally used for anaerobic digestion in the Netherlands. Using food-waste for waste-fed larvae meal production decreased its availability for anaerobic digestion, as the amount of food-waste was limited by the amount of food spilled by humans. The decreased production of electricity, heat and digestate, therefore, was substituted by fossil-fuels and synthetic fertilizer, resulting in an increased environmental impact. Although waste-fed larvae meal can be a feed ingredient with a high nutritional value, changing the application of food-waste, from bio-fuel production to waste-fed larvae meal production, does not reduce the overall environmental impact.

Results of both case studies, therefore, showed that using co-products and food-waste not necessarily results in a reduction of the environmental impact.

Assessing the status quo of a pig system by performing an ALCA can create understanding about the environmental impact of the current situation, and can yield hotspots (i.e. processes with a major impact), and potential improvement options. By performing an ALCA, we identified that replacing SBM with RSM or waste-fed larvae meal can (partly) reduce the environmental impact. Results of this study, however, also showed that an ALCA does not grasp the full complexity of the consequences of implementing an innovation. Based on the results of an ALCA study, one can easily

conclude that feeding more co-products or waste products to livestock results in an improved environmental impact, while this is not necessarily the case.

To assess the environmental impact of implementing an innovation, a CLCA is suitable, especially in combination with scenario analysis to underpin uncertainty (Zamagni et al., 2012; Plevin et al., 2014; Meier et al., 2015). By performing a CLCA, information will be provided on the environmental change in comparison with the current situation. CLCA studies can be highly relevant especially in case we assess the environmental impact of a novel feed ingredient. Such studies provide information about interactions outside the production chain resulting in environmental consequences when the innovation will be implemented, providing support to policy makers during decision making (Zamagni et al., 2012; Plevin et al., 2014; Meier et al., 2015).

The difficulty of performing a CLCA related feed optimization, however, is that it requires insight into world food and feed markets. Especially in pig feed, where feed optimization is based on least cost optimization and a wide range of ingredients are available, diet formulations can change from day to day resulting each in different environmental impacts. Currently, we see developments as precision farming in which each finishing-pig is fed based on its individual needs. It is hard to fully grasp such a complex and dynamic system with an LCA (Plevin et al., 2014). Some researchers as in the report of Dalgaard et al. (2007), therefore, 'simplify' the assumptions and state that at the end an increased demand for pork always results in an increased demand for the marginal protein source, SBM, and the marginal energy source, barley, although different feed ingredients are used. We can, however, wonder if this is correct? The starting point of a CLCA is the point where the stone hits the water, resulting in waves, the so-called cause-and-effect chain (Ekvall and Weidema, 2004). The first waves, or first consequences have most impact. Ekvall and Weidema (2004), therefore, advise to include only the environmental relevant waves and not to estimate consequences far down the cause-and-effect chain. In this study we experienced that performing a CLCA of a complete diet resulted in many consequences on different levels (some far down the cause-and-effect chain). Using RSM in pig diets, for example, increased SBM in diets of dairy cows, resulting in a decrease of palm oil, resulting in a decrease of palm kernel meal, resulting eventually in an increase of SBM. Such consequences will also affect feed prices, resulting eventually in different feed optimization with again different environmental impacts. This complexity makes it difficult to get reliable results when a CLCA for a single diet is assessed, as uncertainties related to the cause-and-effect chain are high.

Results of this study showed, furthermore, that diet formulations are complex, and simplifying the assumptions does not provide e.g. feed companies or policy makers insight on how to reduce the environmental impact of their diets. For example, the study of Van Zanten et al. (2015a) assumed based on CP content, that waste-fed larvae meal will replace SBM or fishmeal. We found, however, based on feed optimization that waste-fed larvae meal replaces protein sources (SBM) and energy sources (barley) and consequently more co-products e.g. wheat middlings were used.

Related to feed optimization, we recommend farmers and animal nutritionists to use an ALCA to get insight in the environmental impact of their feed. In case, however, a policy maker or the feed industry wants to apply a mitigation strategy, it is recommended to perform a CLCA. Such a CLCA should be based on several scenarios (e.g. including different marginal products) to provide insight into different pathways.

5. Conclusions

Based on an ALCA, replacing SBM with RSM reduced LU, but did not affect GWP and EU. Whereas replacing SBM with waste-fed larvae meal decreased GWP and LU, but did not affect EU. Based on a CLCA, replacing SBM with RSM increased impacts on the environment. Replacing SBM with waste-fed larvae meal resulted in an increased GWP and EU but still reduced LU. The CLCA results were, therefore, contradictory with the standard ALCA results. Environmental benefits from an ALCA appeared more promising than from a CLCA. CLCA results for both case studies showed that using co-products and food-waste not necessarily reduces the environmental impact of pork production. For both cases, replacing SBM with RSM or waste-fed larvae meal resulted in an increased net environmental impact. This would have been overlooked when results were only based on ALCA.

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APPENDIX

Diet composition

Diets had to meet requirements for SID methionine and cystine 62%, SID threonine 65%, and SID tryptophan 20%, relative to SID lysine. Furthermore, because of nutritional reasons and taste, the following dietary restrictions were applied in all scenarios: a diet contained a maximum of 30% maize, 40% wheat, 40% barley, 10% peas, 2% molasses, contained 500 FTU phytase per kg, and 0.4% premix to provide minerals and vitamins.

The final diet was formulated by using a commercial linear programming tool (i.e. Bestmix®, Adifo, Maldegem, Belgium), with the nutritional value of feed ingredients from CVB database (CVB (Dutch feed tables), 2011). Linear programming was used to optimize the diet by minimizing the cost price of the diet. The same pricelist was used as in S1 and S2. The CVB database, however, does not contain information about the nutrient content and digestibility of waste-fed larvae meal. The digestibility coefficient is needed to assess the actual nutritional intake. Because the actual nutritional intake is based on the nutrient content multiplied with the digestibility coefficient. The nutrient content of waste-fed larvae meal (Table A.1) was adapted from Van Zanten et al. (2015a), but values were consistent with a literature review of Makkar et al. (2014). Information about the digestibility coefficient of waste-fed larvae meal for pigs is unknown. Information about the digestibility coefficient of waste-fed larvae meal for poultry is, however, available. As the digestibility coefficient for poultry and pigs is quite similar for other protein-rich ingredients, such as SBM and fishmeal, calculation on the digestibility coefficient of waste-fed larvae meal were based on the digestibility coefficient for poultry (Appendix Table A.2 and Table A.3). By using the following equation (CVB, 2011), the net energy (NE) value of waste-fed larvae meal was calculated resulting in 13.01 MJ per kg waste-fed larvae meal:

$$NE \text{ (kJ/kgDS)} = (10.8 \times 425 \text{ digestible crude protein}) + (36.1 \times 228 \text{ digestible crude fat}) + (13.7 \times 0 \text{ starch}) + (12.4 \times 0 \text{ sugar}) + (9.6 \times 20 \text{ remaining carbohydrates}).$$

Table A.1: Nutrient content (g/kg) of soybean meal (SBM) and rapeseed meal (RSM) based on CVB (2010), and waste-fed larvae meal (a) based on data of a laboratory plant (Van Zanten et al., 2015a) and waste-fed larvae meal (b) based on the average value found in Makkar et al. (2014).

	SBM	RSM	Larvae meal (a)	Larvae meal (b)
Dry matter	87.3	87.3	88.0	-
Crude protein	46.4	33.5	47.9	50.4 ± 5.3
Crude fat	1.9	2.6	24.2	-
Crude fibre	3.7	12.0	6.4	5.7 ± 2.4
Ash	6.5	6.7	6.2	10.1 ± 3.3
Phosphorus	0.6	1.1	9.5	16.0 ± 5.5
Calcium	0.3	0.7	8.5	4.7 ± 1.7
Lysine (g/16gN)	6.2	5.5	6.8	6.1 ± 0.9
Methionine (g/16gN)	1.4	2.0	2.4	2.2 ± 0.8

Table A.2: Apparent total tract digestibility coefficients of house fly larvae determined in studies with different types of poultry.

Source	Hwangbo et al. (2009)	Zuidhof et al. (2003)	Pretorius (2011)	Mean value
Animal	Broiler	Turkey	Broiler	
Components				
Crude protein	98.5	98.8	69.0	88.8
Crude fat	-	-	94.0	94.0
Crude fiber	-	-	62.0	62.0
Amino acids				
Arginine	95.5	91.7	-	93.6
Alanine	95.7	94.4	-	95.1
Aspartic acid	93.2	93.2	-	93.2
Cystine	92.7	78.1	-	85.4
Glutamic acid	95.1	93.9	-	94.5
Glycine	95.5	88.0	-	91.8
Histidine	93.7	94.3	87.0	91.7
Isoleucine	92.2	93.9	-	93.1
Leucine	94.7	93.5	-	94.1
Lysine	97.6	96.9	-	97.3
Methionine	95.6	97.7	-	96.7
Phenylalanine	96.8	96.5	-	96.7
Proline	93.4	89.7	-	91.6
Serine	95.6	91.0	-	93.3
Threonine	93.3	91.3	93.0	92.5
Tryptophan	93.9	93.1	95.0	94.0
Tyrosine	96.1	98.0	-	97.1
Valine	94.5	93.8	91.0	93.1

Table A.3: Comparison of the digestibility coefficient (in %) for crude protein, crude fat, and amino acids (AID) between pigs and broilers for soybean meal (SBM) and fishmeal (CVB, 2011).

Ingredient	SBM	SBM	fishmeal	Fishmeal
Animal	Pigs	Broiler	Pigs	Broiler
<i>Total tract</i>				
Crude protein	93	85	87	86
Crude fat	65	71	87	87
<i>Ileal for pigs and total tract for broilers</i>				
Arginine	93	89	91	92
Alanine	85	83	89	91
Aspartic acid	87	89	77	83
Cystine	82	82	70	89
Glutamic acid	90	91	89	89
Glycine	83	81	85	84
Histidine	89	89	85	84
Isoleucine	87	88	89	89
Leucine	87	88	89	91
Lysine	89	88	89	90
Methionine	90	88	88	91
Phenylalanine	88	89	86	89
Proline	89	89	94	84
Serine	87	88	87	84
Threonine	84	85	86	85
Tryptophan	86	89	86	85
Tyrosine	88	89	86	88
Valine	86	87	88	91

Table A.4: Global Warming Potential (GWP) expressed in g CO₂-eq per kg product, energy use (EU) expressed in MJ per kg product, and land use (LU) expressed in m².yr per kg product based on the attributional LCA approach.

Ingredients	GWP	EU	LU
Rapeseed, extruded	456	3.4	1.25
Soybean meal	694	5.9	3.11
Larvae meal	785	9.3	0.00
Peas	741	6.6	5.71
Maize	580	5.2	1.29
Wheat	378	3.0	1.14
Wheat middlings	243	2.2	0.58
Barley	379	2.9	1.30
Sugarcane molasses	319	3.7	0.22
Phytase premix	4999	26.0	0.15
Mervit starter 2220	4999	0.8	0.00
Animal fat	823	12.4	0.00
Limestone	19	0.0	0.00
Salt	180	3.5	0.02
Monocalcium phosphate	4999	18.4	0.32
Sodium bicarbonaat	180	3.9	0.00
L-Lysine HCL	6030	119.9	2.27
L-Threonine	16978	119.9	2.27
DL-Methionine	5490	89.3	0.01

Attributional LCA and consequential LCA related to co-products

Feeding livestock mainly co-products from arable production or the food processing industry offers potential to reduce the environmental impact of livestock products, such as pork, chicken, and eggs. The amount of co-products available, however, is limited and dependent on the production volume of the determining product. For example, the amount of wheat middlings depends on the production volume of wheat flour. This means that when company A decides to increase its use of co-products in livestock diets, fewer co-products are available for company B, which has to adapt his production plan. Based on an ALCA, which does not take the consequences for company B into account, increasing the amount of co-products is a promising strategy to reduce the environmental impact of company A. However, taking into account the consequences for company B, might give a different result: the environmental benefit of increasing the use of co-products in company A will depend on the current application of the co-product in company B. By performing a CLCA, information will be provided on the environmental consequences in comparison with the current situation. So, if the current application of a co-product is bio-energy, and the new application will be livestock feed, the consequences related to the decrease in bio-energy production will be taken into account.

Note: explanation is based on the book chapter 'Future of animal nutrition: the role of life cycle assessment' by C.E. van Middelaar, H.H.E. van Zanten, I.J.M. de Boer in 'Sustainable nutrition and feeding of pigs and poultry' which will be published soon.

Calculation the environmental impact of wheat middlings, animal fat, and SBM based on the theoretical framework of Van Zanten et al. (2014)

Figure 1 illustrates how the environmental consequences of animal fat, wheat middlings, and SBM is calculated. The same principle as for RSM, based on the theoretical framework of Van Zanten et al. (2014), was applied. Below the calculations related to wheat middlings, animal fat, and SBM are explained in more detail.

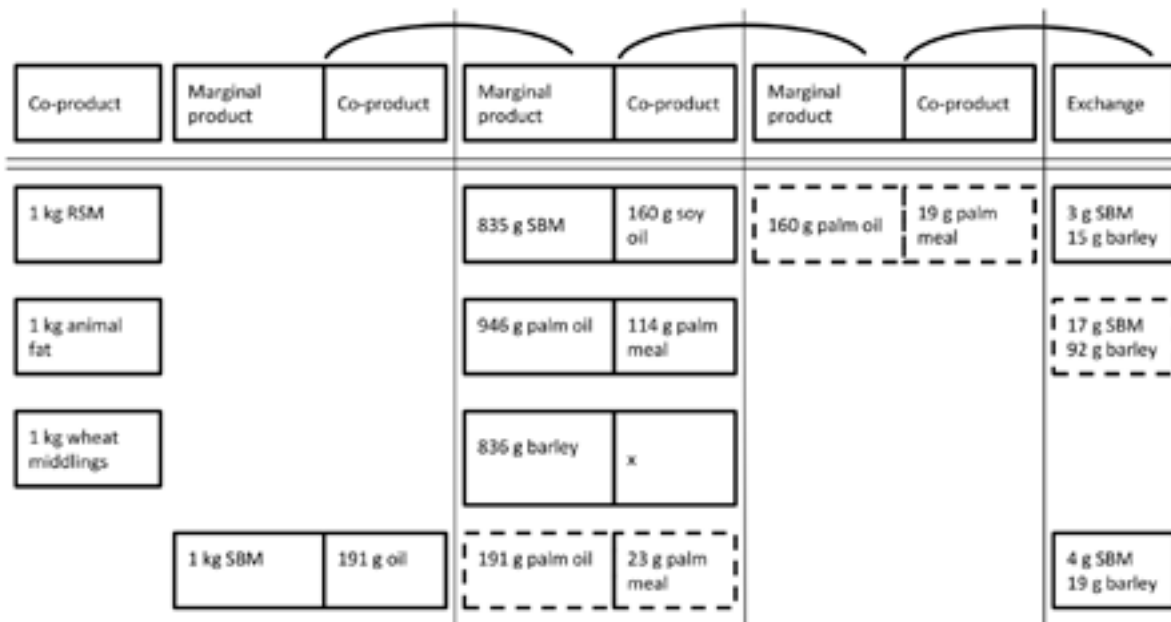


Figure A1: Description of the environmental consequences of increasing rapeseed meal (RSM), animal fat, and wheat middlings and decreasing the use of SBM in diets of finishing-pigs. The full-lines represent an increased production of a product while the dotted-lines represent a decreased production of a product.

Wheat middlings. An increased use of the co-product wheat middlings in diets of finishing-pigs resulted in a reduction of the original application. We assumed that wheat middlings were originally used in diets of dairy cows and that wheat middlings were replaced with barley (the marginal product). The replacement of wheat middlings with barley in diets of dairy cows was based on energy content of barley. An increased production of barley resulted also in an increased production of straw. Straw can be used as bedding material but eventually should be returned to the field to prevent depletion of soil organic matter. We, therefore, did not take straw into account.

Animal fat. An increased use of the co-product animal fat in diets of finishing-pigs resulted in a reduction of the original applications. We assumed that animal fat was originally used in broiler diets and that animal fat was replaced with palm oil (the marginal product). The replacement of animal fat with palm oil in broiler diets was based on energy content. An increased production of palm oil resulted also in an increased production of palm kernel meal, the depended co-product. Palm kernel meal displaces SBM, the marginal product. The displacement of the marginal product is again based on the energy and protein content and follows the same principles as described in the paper.

SBM. A decreased use of the determining product SBM in diets of finishing-pigs resulted in a reduction of soybean production. A reduced production of SBM resulted also in a reduced production of soybean-oil, the depended co-product. The decreased production of soy-oil increased palm-oil production, the marginal product (Dalgaard et al., 2008; Schmidt et al., 2015). The production of palm-oil yields, however, palm kernel meal as well. Palm kernel meal displaces SBM, the marginal product. The displacement of the marginal meal is again based on the energy and protein content and follows the same principles as described in the paper.

47. Multi-objective formulation is an efficient methodology to reduce environmental impacts of pig feeds

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ABSTRACT

The production of pig feeds has a major contribution to climate change, energy use and land occupation impacts of the animal product. Nonetheless, the traditional least-cost (LC) feed formulation methods minimize the cost of the feed mix, without consideration of its environmental impacts. The objective of this study was to estimate the potential mitigation of environmental impacts calculated by Life Cycle Assessment through a multi-objective formulation of pig feeds, in the French context. The linear programming problem built searches the best feed formula under nutritional constraints with a multi-objective function including an economic price index (price of the feed mix relative to LC formulation) and an environmental impacts index (environmental impacts relative to LC formulation). A weighting coefficient between price and environment (α) ranging from 0 to 1 was included. Growing and finishing feeds were formulated with two scenarios of feed ingredients availability (current limited LIM, increased NLIM) and 4 scenarios of feed ingredient prices. When increasing α from 0 to 0.5, the environmental indexes of the growing and finishing feeds dropped down to -10% in LIM and down to -17 to -20% in NLIM scenario, respectively. Concomitantly, the average feed price increased by 1.5% in LIM and 1.7% in NLIM. For α higher than 0.5, the environmental index was almost no further reduced. At $\alpha=0.5$, all the impacts considered were reduced relatively to LIM-LC, excepted for land occupation in NLIM. The low-impact feeds incorporated higher proportions of pea and wheat middlings and lower proportions of meals (rapeseed and sunflower) than LC formulated feeds. The multi-objective formulation of pig feeds is an efficient methodology to find low-impact feeds according to a given economic scenario. Improving the availability of some feed ingredients (pea, co-products of wheat...) at the territory level would allow (at same feed's nutritional composition) further reduction of pig feeds impacts relatively to the current French context. Multi-objective formulation can provide a decision support tool to the feed industry to produce low-impact feeds for the pig production chain.

Keywords: optimization, feed formulas, linear programming, pig feeds.

1. Introduction

Pig production systems (PPS) are facing societal, environmental and economic challenges all around the world. Animal production is expected to increase in the following years to feed the raising human demand for animal products (FAO, 2011). PPS should also reduce their environmental burden. They are associated various environmental impacts like climate change, land use, and eutrophication particularly in territories with high concentrations of livestock (North West France, Netherlands...). The rising of the feed ingredients prices (cereals and meals from oilseeds and protein crops) and the volatility of the animal products prices also reduce the stability and the average level of the gross margin of pig producers (EC, 2013).

In farrow-to-finish PPS, feeds account for 60% to 70% of the feeding cost and the production of feeds has a major contribution to climate change (55%-75%), energy use (70%-90%) and land occupation (85%-100%) impacts of the animal product (Basset-Mens and van der Werf, 2005; Dourmad *et al.*, 2014). Both feeds' cost and environmental impacts are highly determined by their composition in feed ingredients. Some of them, like soybean meal, account for more than 10% of the feed composition and are characterized by relatively high price and impacts (Wilfart *et al.*, 2016). Some other feed ingredients are incorporated into small amounts into feeds but have high environmental impacts per kilogram, e.g. feed-use amino acids and monocalcium phosphate (Garcia-Launay *et al.*, 2014). Therefore, there is possibly a great potential to reduce the environmental impacts of animal products through the formulation of low-impact feeds (Nguyen *et al.*, 2012).

Nonetheless, the traditional least-cost (LC) feed formulation method minimizes the cost of the feed mix, without consideration of its environmental impacts. LC formulation incorporates the feed ingredients to meet nutritional requirements according to production objectives, while minimizing the cost of the feed mix, using a linear programming model which calculates the feed cost as the objective function. However, the maximal technical performance does not necessarily correspond to the

economic and/or environmental optimum (Morel *et al.*, 2012; Pomar *et al.*, 2007). Therefore formulating low-impact feeds requires an alternative approach to LC. Castrodeza *et al.* (2005) developed a multiple goal programming model accounting for the feed cost and the excess of feed contents in amino acids and phosphorus, which does not consider the environmental impacts of the feed ingredients themselves. Nguyen *et al.* (2012) formulated low-impact feeds for poultry feeds under constraints of feed's climate change and eutrophication impacts with the cost being the objective function. They highlighted that accounting for only two impacts may lead to pollution transfer. Therefore, there is no reliable and simple feed formulation method available for feed manufacturers that aim at reducing both the feed cost and its environmental impacts. The objectives of this study were to develop a multi-objective formulation method of pig feeds relying on environmental impacts of feed ingredients calculated by Life Cycle Assessment (LCA), and to illustrate its potential to mitigate the environmental impacts of feeds for fattening pigs in the French context.

2. Methods

Table 1. List and description of the variables, vectors and matrixes inputs of the feed formulation problem.

	Inputs of the problem	Description
(1)	$\begin{bmatrix} Nut_{11} & \cdots & Nut_{p1} \\ \vdots & \ddots & \vdots \\ Nut_{1n} & \cdots & Nut_{pn} \end{bmatrix}$	Matrix of the nutrients composition of each feed ingredient, Nut_{ij} being the content of nutrient j in feed ingredient i .
(2)	$\begin{bmatrix} LCA_{11} & \cdots & LCA_{q1} \\ \vdots & \ddots & \vdots \\ LCA_{1n} & \cdots & LCA_{qn} \end{bmatrix}$	Matrix of the environmental impacts of each feed ingredient, LCA_{ik} being the k^{th} environmental impact of feed ingredient i .
(3)	$\begin{bmatrix} MinNut_1 \\ \vdots \\ MinNut_p \end{bmatrix} \text{ and } \begin{bmatrix} MaxNut_1 \\ \vdots \\ MaxNut_p \end{bmatrix}$	Vectors of the nutrient requirements and of the maximum nutrients contents of the feed, defined in accordance with the animal performance objective.
(4)	$\begin{bmatrix} MinRate_1 \\ \vdots \\ MinRate_n \end{bmatrix} \text{ and } \begin{bmatrix} MaxRate_1 \\ \vdots \\ MaxRate_n \end{bmatrix}$	Vectors of the minimum and maximum incorporation rates for each feed ingredient i .
(5)	$\begin{bmatrix} 105\% \times FLCA_{q-1ref} \\ \vdots \\ 105\% \times FLCA_{qref} \end{bmatrix}$	Vector of the maximum values for the environmental impacts not included in the MO function (eutrophication and acidification).
(6)	$[Cost_1 \dots Cost_n]$	Vector of cost of each feed ingredient i

Multi-objective optimization of the feed formulas with both economic and environmental indicators has been chosen in order to avoid pollution transfer and to produce formulas consistent with the current praxis of the feed manufacturers. The multi-objective formulation method calculates the nutritional contents, the cost and the LCA environmental impacts of the considered feed from the characteristics of each feed ingredient (FI) and the associated incorporation rates.

Feed ingredients characteristics

FI impacts came from the ECOALIM dataset of the AGRIBALYSE® database (Wilfart *et al.*, 2016) and included phosphorus demand (PD, in kg P/kg of FI), ILCD climate change including land use change (CC, in kgCO₂-eq/kg), ILCD acidification (AC, in molcH⁺-eq/kg), CML eutrophication (EU, in kgPO₄³⁻/kg), CED 1.8 non-renewable energy demand (NRE, in MJ/kg), and CML land occupation (in m².year/kg). The impacts of the feed ingredients transport from the storage

organization to the feed factory were added with background data from Ecovinvent v3.1. attributional database (Weidema *et al.*, 2013) considering average distances of pig production in Brittany, North-West of France to the main areas of cereals production, to the harbors of imported meals and the distances to mills and starch manufactures. Nutritional composition of the feed ingredients came from Sauvant *et al.* (2004) excepted for few co-products for which data were provided by R&D institutes. All the impacts were considered at the entry of the feed factory for an application in feed manufacturing.

Table 2. List and description of the variables, vectors and matrixes outputs of the feed formulation problem.

	Outputs of the problem	Description
(7)	$\begin{bmatrix} Rate_1 \\ \vdots \\ Rate_n \end{bmatrix}$ and $\begin{bmatrix} Rate_{1ref} \\ \vdots \\ Rate_{nref} \end{bmatrix}$	Decision vectors where each $Rate_{iref}$ and $Rate_i$ corresponds to the incorporation rate for the feed ingredient i in the reference feed and MO optimized feed
(8)	$\begin{bmatrix} FNut_1 \\ \vdots \\ FNut_p \end{bmatrix}$ and $\begin{bmatrix} FNut_{1ref} \\ \vdots \\ FNut_{pref} \end{bmatrix}$	Vectors of the nutrient contents of the feed formula where each $FNut_{jref}$ is the j^{th} nutrient content of the feed after LC formulation and $FNut_j$ is the j^{th} nutrient content of the feed after MO formulation.
(9)	$[FLCA_{1ref} \dots FLCA_{qref}]$	Vector of the environmental impacts of the reference formula produced after LC formulation where $FLCA_{kref}$ is the k^{th} environmental impact of the reference formula
(10)	$[FLCA_1 \dots FLCA_q]$	Vector of the environmental impacts of the formula produced after MO formulation where $FLCA_k$ is the k^{th} environmental impact of the formula

The multi-objective feed formulation problem

Like in the LC traditional formulation method, the multi-objective formulation method developed was based on linear programming (Figures 1 and 2). The incorporation rates of each available feed ingredient were determined under a series of linear constraints, while minimizing the objective function. The list and description of variables, vectors and matrixes utilized for the feed formulation problem is available in Tables 1 and 2. The method was developed in two steps: first one to produce a reference formula through LC formulation (Figure 1) and second step searching for the solution of the multi-objective optimization problem (Figure 2).

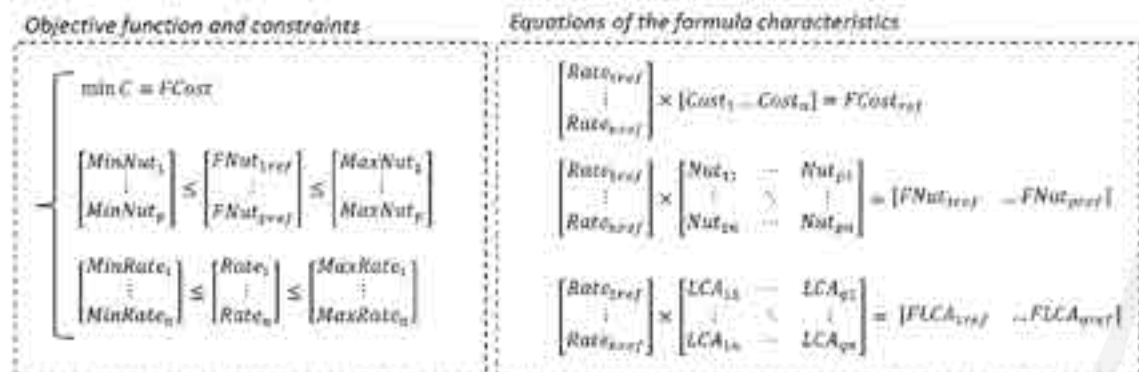


Figure 1. First-step of the multi-objective optimization problem involving traditional least-cost formulation minimizing function C to define a reference formula, and a second step with optimization of the multi-objective (MO) function.

First step corresponds to a traditional LC formulation which aims at producing a reference formula. The objective function is the feed cost which is calculated as the sum of the feed ingredients costs (6) multiplied by their respective incorporation rates. The optimization algorithm searches for

incorporation rates (7) that minimize the objective function while covering the nutritional requirements (3) of the pig and under constraints of incorporation rates (4) for each feed ingredient. Feeds are formulated while ensuring minimum nutrient contents, in order to cover the animal requirements in net energy and amino acids according to the performance objective. Minimum limits for standardized ileal digestible amino acids were calculated according to the regulation on pig feed protein content (CORPEN, 2003) and ideal amino acid profiles from van Milgen *et al.* (2008). Minimum and maximum values of feed ingredients incorporation rates have been established to account for both the availability of each feed ingredient on the market and the technological constraints of the feed fabrication. From the formula produced during this first step, the reference values for feed cost, LCA environmental impacts (9), and nutrient contents are calculated (8).

Multi-objective function

$$\min MOF = (1 - \alpha) \underbrace{\frac{FCost}{FCost_{ref}}}_{\text{Price index}} + \alpha \underbrace{\left(\frac{2\beta \times FLCA_1}{FLCA_{1ref}} + \frac{\beta \times FLCA_2}{FLCA_{2ref}} + \frac{\beta \times FLCA_3}{FLCA_{3ref}} + \frac{\beta \times FLCA_4}{FLCA_{4ref}} \right)}_{\text{Environmental impacts index}}$$

Constraints

$$\begin{cases} \begin{bmatrix} MinNut_1 \\ \vdots \\ MinNut_p \end{bmatrix} \leq \begin{bmatrix} FNut_1 \\ \vdots \\ FNut_p \end{bmatrix} \leq \begin{bmatrix} MaxNut_1 \\ \vdots \\ MaxNut_n \end{bmatrix} \\ \begin{bmatrix} MinRate_1 \\ \vdots \\ MinRate_n \end{bmatrix} \leq \begin{bmatrix} Rate_1 \\ \vdots \\ Rate_n \end{bmatrix} \leq \begin{bmatrix} MaxRate_1 \\ \vdots \\ MaxRate_n \end{bmatrix} \\ \begin{bmatrix} FLCA_{q-1} \\ \vdots \\ FLCA_q \end{bmatrix} \leq \begin{bmatrix} 105\% \times FLCA_{q-ref} \\ \vdots \\ 105\% \times FLCA_{q-ref} \end{bmatrix} \end{cases}$$

Equations of the formula characteristics

$$\begin{cases} \begin{bmatrix} Rate_1 \\ \vdots \\ Rate_n \end{bmatrix} \times \{Cost_1 \dots Cost_n\} = FCost \\ \begin{bmatrix} Rate_1 \\ \vdots \\ Rate_n \end{bmatrix} \times \begin{bmatrix} Nut_{11} & \dots & Nut_{p1} \\ \vdots & \dots & \vdots \\ Nut_{1n} & \dots & Nut_{pn} \end{bmatrix} = \{FNut_1 \dots FNut_p\} \\ \begin{bmatrix} Rate_1 \\ \vdots \\ Rate_n \end{bmatrix} \times \begin{bmatrix} LCA_{11} & \dots & LCA_{q1} \\ \vdots & \dots & \vdots \\ LCA_{1n} & \dots & LCA_{qn} \end{bmatrix} = \{FLCA_1 \dots FLCA_q\} \end{cases}$$

Figure 2. Second step of the multi-objective optimization problem with optimization of the multi-objective (MO) function.

The second step of the feed formulation problem utilizes the constraints of the LC formulation and calculates a multi-objective (MO) function including cost and environmental impacts criteria (Figure 2). All the criteria included in the MO function are normalized by their reference value calculated from the LC formulation. The MO function includes a price index which is the normalized feed cost and an environmental impacts index which comprises four normalized environmental impacts. Global impacts for which feed has a major contribution have been selected to be included in the MO function: climate change, phosphorus demand, non-renewable energy demand, and land occupation (Basset-Mens and van der Werf, 2005; Dourmad *et al.*, 2014). Eutrophication and acidification have been also included in the problem considering the algal bloom eutrophication occurring in several costs of Brittany, the main area for pig production in France. Both have been utilized as constraints (5) of the feed formulation problem, their value being limited to 105% of their reference value from the LC formulation (Figure 1). The MO function also includes two weighting factors, α and β . The α factor corresponds to the weight, ranging from 0 to 1, for the environmental impacts index, $1 - \alpha$ being the complementary weight on the price index. The β factor, which equals 0.2 in our case, manages the weighting between the four environmental impacts included. A double 2β has been allocated to the climate change impact considering the strong international efforts that are made to mitigate this impact (Gerber *et al.*, 2013). All the β factors included in the environmental impacts index sum up to 1. Therefore the MO function while moving the α factor from 0 to 1 allows investigating the trade-off between economic and environmental objectives. The optimization algorithm searches for

incorporation rates that minimize the MO function while respecting the constraints of the LC formulation and under constraints (3) (4) on local environmental impacts increase (5).

Simulation of scenarii

The MO method was tested for the formulation of growing and finishing feeds for fattening pigs considering that fattening has the major contribution in pig production to the feeding cost and to the environmental impacts (Garcia-Launay *et al.* 2014). To investigate the ability of the MO method developed to formulate low-impact feeds, we defined several scenarii to account for the variability of the situations encountered in France. Two scenarios of feed ingredients availability (to define the vector (4) and 4 scenarios of feed ingredients prices (6) have been developed.

The current limited (LIM) and increased (NLIM) availability scenarii have been developed from expert knowledge and correspond respectively to the current situation in France and to an increased potential availability of some feed ingredients such as spring peas, faba beans ... The 4 economic scenarii were constructed in order to cover a range of contrasted situations and correspond respectively to the market feed ingredients prices in September 2011, June 2012, August 2013 and February 2014. These four periods have been selected because they were characterized by varying prices of soft wheat, maize grain and soybean meal which resulted in contrasted soybean meal/soft wheat and maize grain / soft wheat ratios of prices. Costs were obtained from La Dépêche Commerciale (2011, 2012, 2013, and 2014) market newspaper and from Arvalis R&D institute.

Feeds of the scenarii limited availability and least-cost formulation (LIM-LC), limited availability and MO formulation (LIM-MO), and increased availability with MO formulation (NLIM-MO) were evaluated. Feeds were formulated using OpenSolver for Excel (Mason and Dunning, 2010), open source software which performs optimization of linear programming models using branch and bound, for problems with a large number of variables and constraints.

3. Results

Results provided are average values over the 4 economic scenarios.

Feed formulas

Average feed formulas for the finishing feed are provided in Figure 3. Proportion of cereals and oilmeals in feed formulas decreased from the LIM-LC to the NLIM-MO scenario while proportions of coproducts of wheat and of oilseeds and protein crops increased. Proportion of coproducts of wheat and of oilseeds and protein crops increased from LIM-LC to LIM-MO formulation because these feed ingredients were characterized by lower environmental impacts than cereals and oilmeals. Proportion of coproducts of wheat, and of oilseeds and protein crops amplified between LIM and NLIM in MO formulation because of the improved availability of coproducts and protein crops like wheat middlings, wheat feed flour and spring peas in the NLIM scenario. The same statement was also observed for growing feeds.

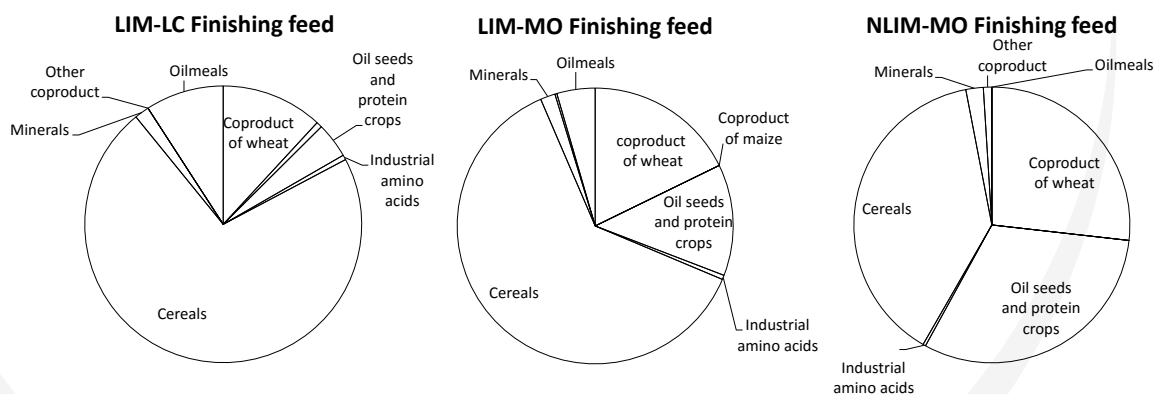


Figure 3. Average feed formulas over the 4 economic scenarii, obtained for LIM-LC (limited availability of feed ingredients and least-cost formulation), LIM-MO (limited availability and multiobjective formulation), and NLIM-MO (increased availability and multiobjective formulation).

Variation of feed cost and environmental impacts with MO formulation

Figure 4 shows the variations of the average price index and the average environmental impacts index (over the 4 economic scenario) of the feed formulas when α varies from 0 to 1. When $\alpha=0$ the price index and environmental index were close to 1 because it corresponds to LC formulation. When α varied from 0 to 0.5 the price index of the feeds in NLIM was increased by 2% while the environmental impact index was reduced by 17-20%. When further increasing α up to 1, the price index reached +5-6% while the environmental impacts index remained almost stable. The variations were similar for the LIM scenario but to a lower extent. This relationship between price and environmental impacts indexes shows that in our context it was not advisable to increase the value of α to more than 0.5 because no further mitigation of impact could be expected.

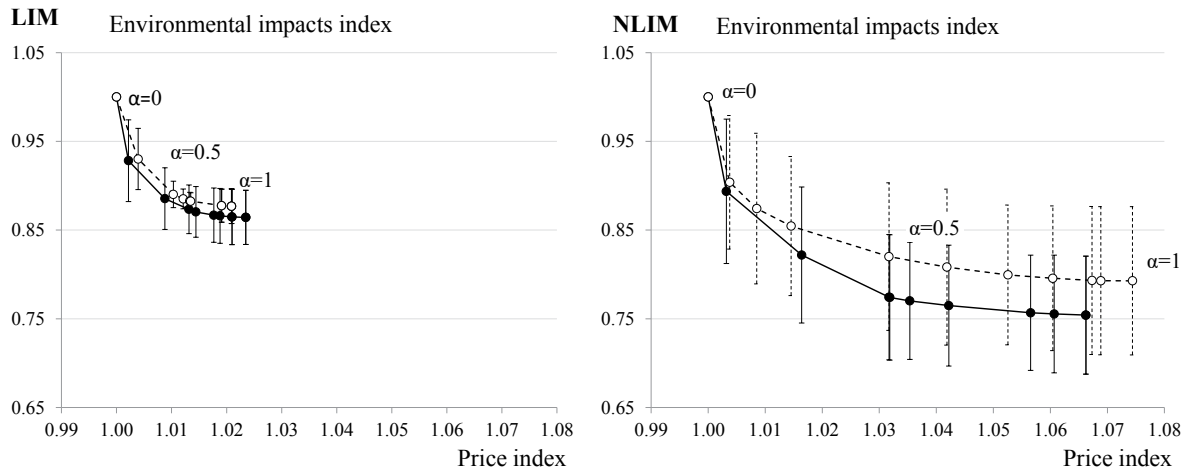


Figure 4. Variation of average environmental and price indexes in LIM and NLIM scenarios when formulating growing (●) and finishing (○) feeds with α ranging from 0 to 1.

Consequently, Table 3 provides the average prices and the average environmental impacts of the feed mix (40% growing / 60% finishing) formulated at $\alpha=0.5$. Relatively to LIM-LC, LIM-MO reduced all the environmental impacts included in the MO function as well as eutrophication and acidification impacts, while slightly increasing feed price (+1%). Relatively to LIM-MO, NLIM-MO further decreased all the environmental impacts excepted land occupation while further increasing feed price (+1%).

Table 3: Average price and environmental impacts (\pm s.d. | % relatively to LIM-LC) of 1t of feed mix (40% growing and 60% finishing) produced, for the reference scenario (LIM-LC) and the LIM-MO and NLIM-MO scenario at $\alpha = 0.5$.

Feed price and environmental impacts	LIM-LC	LIM-MO $\alpha = 0.5$	NLIM-MO $\alpha = 0.5$
Feed price (€)	216 \pm 12.4	219 \pm 12.2 +1.0%	222 \pm 8.4 +2.7%
Phosphorus Demand (kg P)	3.4 \pm 0.36	3.2 \pm 0.15 -6%	2.7 \pm 0.15 -21%
Non-renewable Energy (MJ)	5150 \pm 568.7	4442 \pm 351.2 -14%	3899 \pm 290.5 -24%
Climate Change (kgCO ₂ -eq)	499 \pm 18.2	426 \pm 4.6 -15%	337 \pm 11.5 -32%
Land Occupation (m ² .year)	1418 \pm 59.5	1238 \pm 13.1 -13%	1325 \pm 61.8 -6%
Acidification (mol H ⁺)	9.7 \pm 0.48	9.0 \pm 0.58 -7%	6.6 \pm 0.42 -32%
Eutrophication (kg PO ₄ -)	3.6 \pm 0.05	3.2 \pm 0.02 -11%	2.9 \pm 0.09 -19%

4. Discussion

This paper proposes a novel methodology for feed formulation, which aims at integrating the environmental impacts calculated by LCA in the traditional least-cost formulation approach. The development of the multiobjective formulation method was made possible by the development of the ECOALIM dataset of the AGRIBALYSE® database which provides homogeneously developed and reliable environmental impacts (Wilfart et al., 2016). Indeed optimization is useful for decision support but may lead to inappropriate decisions if the model of calculation and the underlying data are not robust in the range of situations investigated.

In the scenarii illustrated, the reduction of impacts through multi-objective formulation was obtained by incorporating less oil meals and cereals and more coproducts and protein crops (especially spring peas). Indeed, coproducts are characterized by relatively low impacts mainly associated to economic allocation of impacts adopted in the ECOALIM dataset and spring peas have lower impacts than meals like soybean meal and rapeseed meal (Wilfart *et al.*, 2016). The reduction of impacts was also improved in the NLIM scenario relatively to the LIM scenario. This statement suggests that better balance among various crop productions would benefit to the environmental impacts of the whole pork chain.

The inclusion of two indexes (price and environmental impacts) into the objective function with weighting factors allows investigating the relationships between feed price and environmental impacts. It gives to the end-user an overview of the possible trade-off that he can make between price and environmental impacts. In our scenarii, the reduction of impacts is of interest for $\alpha=0.5$ with a very moderate additional price and anyone can simply identify that there is no extra reduction expected when further increasing α . Therefore the end-user can choose the appropriate weighting of the price and environmental impacts indexes. Additionally, the multiobjective formulation approach proposed relies on the traditional customary least-cost formulation method for its first step and is consistent with the current formulation constraints and practices. This approach provides feed formulas in accordance with the concerns of the potential end-users. Finally, the incorporation of 4 environmental impacts into the objective function limits the risk of pollution transfer. However, the behavior of the MO linear programming model was also characterized by a decrease of the environmental index for α between 0.5 and 1 that was associated to further reduction of climate change but with concomitant augmentation of land occupation. Indeed, the reduction of this index may be associated to reduction of some impacts and increase of some other ones. In our case climate change with a weight of 2β compensated for land occupation augmentation. This statement highlights how important is the choice of the weighting factors of such a methodology.

Various formulation methods have already been proposed so far to account for the environmental burden of pig production (Castrodeza et al., 2005; Pomar et al., 2007, Nguyen et al., 2012, Garcia-Launay *et al.*, 2014). Some of them focused on the reduction of the crude protein and phosphorus supplies to the animals that are involved in the ammonia, nitrous oxide, nitrates and phosphates emissions occurring on farm. To our best knowledge, Nguyen *et al.* (2012) are the only authors that already included environmental impacts calculated by LCA in their feed formulation problem, but only as constraints. The multiobjective formulation proposed in the present paper is in fact complementary with these previous studies. Indeed the previous formulation methods mostly included nutrients excreted as constraints in order to modify the on-farm emissions, whereas the present method mitigates the upstream impacts.

The feed formulas obtained with the multi-objective formulation must be further evaluated on the pig unit. Indeed, formulas including high levels of coproducts and/or newly available products (microalgae, ...) may affect the animal performance, with an indirect effect on the environmental impacts.

5. Conclusions

Multi-objective formulation of pig feeds appears a promising approach to reduce upstream environmental impacts related to pig production. It refreshes the traditional least-cost formulation method by providing a methodology more in accordance with the current animal production issues and challenges. It gives an example of decision support using LCA studies and highlights the necessary precision and reliability of life cycle inventories to put into practice mitigation options. On-going work on broiler feeds will allow investigating the genericity of the proposed methodology.

Further work will include global assessment at farm gate of the feed formulas obtained through MO formulation.

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ORAL PAPERS

PEF, LABELLING AND COMMUNICATION

8. PEF, labelling and communication

11. Life Cycle Assessment and Product Category Rules for starch products

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ABSTRACT

A few years ago, a Life Cycle Assessment (LCA) was performed on products produced by the members of the European Starch Industry Association (Starch Europe, formerly AAF). Since then, different initiatives related to the assessment of environmental impacts of products from a life cycle perspective were launched, such as the European Commission's Product Environmental Footprint (PEF). Taking into account these developments, Starch Europe decided to update its LCA and develop Product Category Rules (PCR) for products of the starch industry. The study, which was completed in 2015, was performed in 3 steps: 1) analysing the compliance of the 2012 LCA for starch industry products with the PEF guide, 2) developing PCR and 3) updating the LCA. During the first step of this study an overview was made of the methodological requirements indicated in the "Commission Recommendation on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations" (PEF guide) and the LCA study was checked for its compliance. If the study did not comply on some point, a solution was proposed. It was decided together with Starch Europe which non-compliances were feasible to be made compliant and incorporated in both the PCR and the updated LCA. It became clear that many aspects were already compliant or could be made compliant easily. Those that required a larger effort were implemented when the added value for the study was considered large enough (e.g. using new agricultural data). For some aspects, compliance was not feasible (due to technical or budgetary constraints) or not desirable (not considered as an improvement to the study). PCR for products of the starch industry were developed taking into account the ISO 14025 standard, the PEF guide and the Guidance for the PEF pilots.

Keywords: Starch Industry Association, Product Category Rules, Product Environmental Footprint

1. Introduction

In 2012, VITO finalized a Life Cycle Assessment (LCA) for the European Starch Industry Association (Starch Europe, formerly AAF). Over the past years, different initiatives related to the assessment of environmental impacts of products from a life cycle perspective were launched. One of these is the Product Environmental Footprint (PEF) method published by the European Commission, as part of the Single Market for Green Products initiative. This method is being tested in a pilot phase at the moment. Taking into account these and other developments, Starch Europe decided to update its LCA and develop Product Category Rules (PCR) for products of the starch industry.

2. Methods

The study was performed in 3 steps: 1) analysing the compliance of the 2012 LCA for starch industry products with the PEF guide, 2) developing PCR and 3) updating the LCA.

During the first step of this study an overview was made of the methodological requirements indicated in the "Commission Recommendation on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations" (PEF guide). The 2012 LCA study was then checked for its compliance to the PEF guide. Each methodological aspect was rated with three possibilities: compliant, partially compliant, or not compliant. If the study did not or only partially comply on some point, a solution was proposed. Each solution was evaluated according to its feasibility, taking into account technical aspects (data availability, for example) as well as budgetary considerations (cost versus benefits). It was then decided together with Starch Europe which of the non-compliances were feasible to be made compliant.

The second step of the study was to develop PCR for products of the starch industry. These PCR take into account the ISO 14025 standard, the PEF guide and the Guidance for the PEF pilots. The ISO 14025 standard "Environmental labels and declarations - Type III environmental declarations - Principles and procedures" was used as a starting point for both the stepwise approach for preparing

the PCR document and the content of the PCR. The full procedure for developing Type III environmental declaration programmes as described in ISO 14025 was not followed, and no Type III environmental declarations (EPDs) were developed within this project. The development of this PCR was based on a participatory process with the members of Starch Europe. As many decisions were already made during the first LCA project, and due to the strict timing for this project, no open participatory consultation with interested parties has been organised nor has a PCR review by a third-party panel been performed. Although the PCR are not published in the official PEFCR format, the PEF guide and its guidance on how to create product category-specific methodological requirements for use in PEFCRs was followed as much as possible. When methodological aspects of the PCR do not comply with the PEF guide, a justification is included. The PEF document “Guidance for the implementation of the EU PEF during the EF pilot phase” was taken into account when feasible; bearing in mind that it is only a working document, which will be further elaborated during the PEF pilots.

As a third step, the LCA finished in 2012 was updated. As the PCR took into account the PEF guide, the original LCA study was updated to fully comply with these PCR. Furthermore, the update took into account the latest developments in the general field of LCA on academic level. The results are based on an extensive dataset, collected from a significant number of starch plants across the EU, representative for the European starch industry.

3. Results

As a result of the first step, an overview of the methodological requirements of the PEF guide, the compliance of the 2012 LCA study, and the feasibility of an update was composed. A short summary with some examples is shown in Table 1.

Table 1: Compliance of LCA study to methodological requirements of the PEF Guide and feasibility of update

Compliance to methodological requirements of PEF Guide	Examples
Compliant or compliance easily achievable	<ul style="list-style-type: none"> – Include the NACE-code in the functional unit. – Explicitly justify the exclusion of use and end-of-life stage.
Aspects that can be made compliant, but require a larger effort	<ul style="list-style-type: none"> – Add the impact of direct land use changes to the carbon footprint. – Perform a semi-quantitative assessment of data quality. – Use agricultural data from a new database (Agribalyse or Agri-footprint) to improve data quality.
Aspects difficult to be made compliant	<ul style="list-style-type: none"> – No cut-offs allowed by PEF Guide: <ul style="list-style-type: none"> ○ Including capital goods (machinery used in production processes, buildings and office equipment) would require a huge amount of work. ○ Including non-bulk chemicals is difficult, since some of this information is confidential even among Starch Europe members.
Aspects for which it is technically possible but not desirable to update the LCA study	<ul style="list-style-type: none"> – Modelling the electricity from the grid as precisely as possible giving preference to supplier-specific data. A specific electricity mix per product can be made (country or supplier specific). However, this does not seem very useful nor appropriate for a sector study. Instead, a weighted average mix for the grid is made, representing electricity use in an ‘average’ Starch Europe plant. – Using the International Reference Life Cycle Data

System (ILCD) nomenclature for the Resource Use and Emissions Profile. This can be added, but will require a lot of administrative work and will not add any value to the study.

The second step resulted in PCR for products of the starch industry. The PCR document is considered a position paper of the starch industry that will serve as a background document that summarizes and justifies the decisions taken by the industry with regard to rules relevant for LCA of products produced by the starch industry and that will prepare the industry for upcoming PEFCR rules. The PCR:

- specify the products from the starch industry for which this PCR applies (see Figure 1);
- identify and document the goal and scope of the LCA-based information for the product category;
- define the parameters to be covered and the way in which they are collated and reported;
- state which stage of a product’s life cycle is to be considered and which processes are to be included in the life cycle stages;
- include the rules for calculating the Life Cycle Inventory and the Life Cycle Impact Assessment, including the specification of the data quality to be applied;
- identify the rules for producing the additional environmental information for the product category.

Products from wheat		Products from maize		Products from potatoes	
<i>Name of product</i>	<i>Application</i>	<i>Name of product</i>	<i>Application</i>	<i>Name of product</i>	<i>Application</i>
Starch slurry	<i>FO, FE, I</i>	Starch slurry	<i>FO, FE, I</i>	Starch slurry	<i>FO, FE, I</i>
(Loose) Bran (as such, after grinding)	<i>FE</i>	Steep liquor	<i>FE, I</i>	Potato proteins	<i>FE</i>
Dry wheat feed (bran and solubles mixed, then dried) – pelletised or not	<i>FE</i>	Dry corn feed (steep liquor mixed with fibres, then dried)	<i>FE</i>	Fruit juice	<i>FE, I</i>
Dry (Solubilised or not) gluten	<i>FO, FE</i>	Wet corn fibres	<i>FE</i>	Concentrated fruit juice	<i>FE</i>
Wet solubilised gluten	<i>FO, FE</i>	Dry germs	<i>FO, FE</i>	Wet pulp	<i>FE</i>
Liquid solubles (as such, after evaporation)	<i>FE</i>	Oil	<i>FO</i>	Dry pulp (fibres)	<i>FE</i>
		Dry proteins	<i>FE</i>		
Products from wheat, maize or potatoes					
<i>Name of product</i>					<i>Application</i>
Liquid glucose (including hydrolysates, fructose and glucose syrups)					<i>FO, I</i>
Dry crystallized dextrose					<i>FO, FE</i>
Maltodextrin					<i>FO</i>
Liquid sorbitol					<i>FO, I</i>
Dry sorbitol					<i>FO, I</i>
Unfermented special polyols					<i>FO, I</i>
Native and lightly modified starches (e.g. light or dry modification)					<i>FO, FE, I</i>
Modified starch – liquid modification (e.g. PO, esters and ethers)					<i>FO, FE, I</i>
Dextrins					<i>FO, I</i>
Potable alcohol					<i>FO</i>
Broth (by-product from potable alcohol)					<i>FE</i>

(*FO = Food, FE = Feed, I = Industrial*)

Figure 1: Overview of reference products included in the study and their application

The third step led to an updated version of the 2012 LCA. The analysis of compliance showed that the main point that needed to be updated in the LCA was the agricultural data. The agricultural production has a large influence on the results, and more suitable and recent data have become available. The Agribalyse and Agri-footprint database were compared, and generally the Agribalyse (v1.1) database was found more suitable for this study. The Agribalyse database contains mostly French data, and the largest share of materials processed in the 33 plants that provided data for the LCA study came from France. French production is thus considered representative for the areas where the European starch industry sources its raw materials from. Also, capital goods are included in the Agribalyse database (compliant to the PEF Guide) and background data from Ecoinvent are used (e.g. for electricity production), which is compatible with the rest of the study. In addition to the basic data of Agribalyse, carbon dioxide emissions due to direct land use change were added with the method used in the Agri-footprint database. In addition to the use of new agricultural data for the LCA, the aspects on which PEF compliance could easily be achieved were updated.

In the following paragraphs, a short summary of the LCA and its results is given. The definition of the functional unit was agreed upon as “the production of 1 ton d.s. (dry substance) of reference products at the starch plant exit gate”. The system boundary diagram shown in **Error! Reference source not found.** generically outlines the different life cycle stages and the inputs and outputs that are included in the system boundaries of this study.

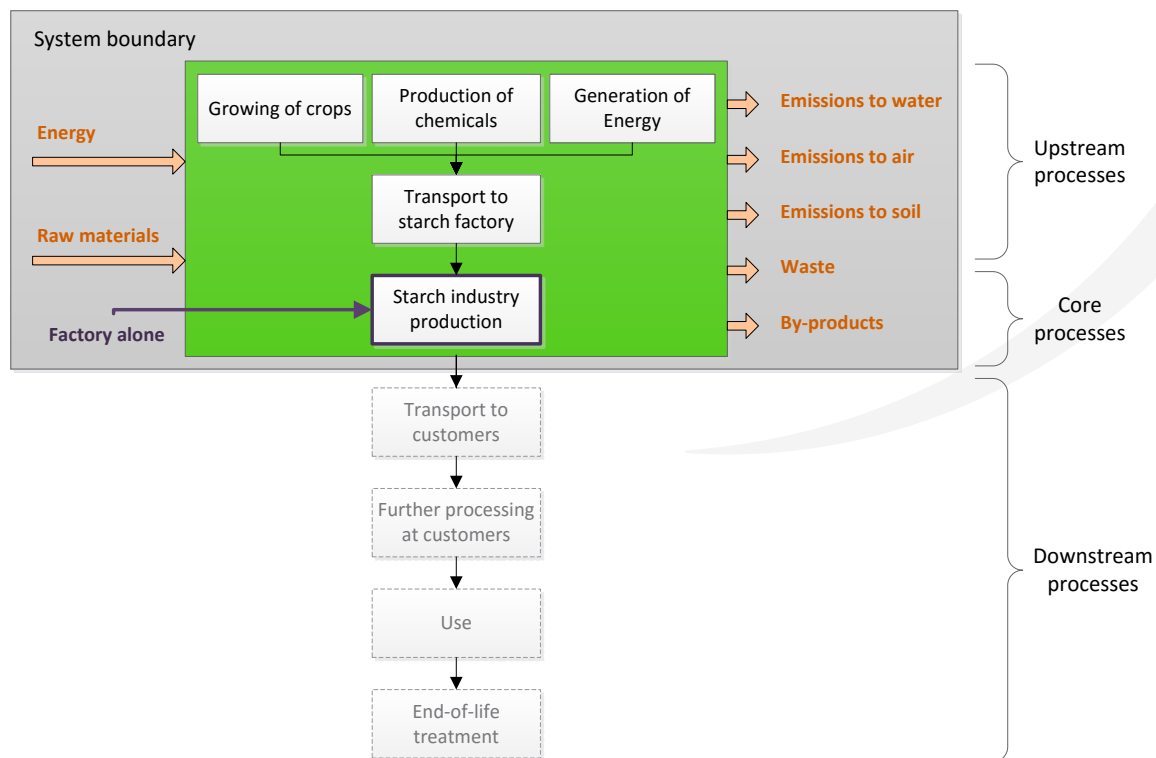


Figure 2: System boundaries

Allocation is very important in the LCA study of starch products, since the starch industry produces a wide range of different reference products per specific raw material (either maize or wheat or potatoes). Many process steps in the starch industry produce more than one useful output. For the starch industry, allocation cannot be avoided through subdivision or system expansion. Subdivision/disaggregation is done up to a certain level, as some processes can be attributed to one product only. However, many production processes in the starch industry are complex and may be considered as a ‘black box’ that cannot be subdivided further. System expansion is not useful either, as one of the goals of the LCA is to determine the environmental impacts per product, to allow

companies that use only one specific starch industry product to use the results as an input for the LCA of their products.

There have been continuous discussions amongst LCA practitioners about the choice of allocation methods. Both physical and economic allocation have benefits and drawbacks. In the previous LCA on starch products different allocation rules were applied under different boundaries and taking into account guidelines from the relevant standards and handbooks (e.g. ISO 14040/44, PAS2050, ILCD Handbook). Based on the results, Starch Europe decided to use mass allocation. It was chosen because it offers the clearest picture throughout the process tree, it is based on the best available data and it allows easy monitoring of process improvements.

When looking at the results of the life cycle impact assessment (LCIA), the importance of the agricultural life cycle phase in the environmental impact of products produced by the EU starch industry becomes clear (see examples in Figure 3). The relatively large impact of the growing of the crops is mainly due to the use of fertilizers, pesticides and energy during the cultivation process. For the production of native starch, agriculture is responsible for about two thirds of the impact for the carbon footprint, and over eighty percent for water resource depletion. When additional process steps are needed, the contribution of agriculture to the impact of other products from the starch industry remains major, though to a slightly lesser degree. This is shown by the liquid glucose profile in Figure 3. When looking only at the processes which occur at the starch plants, it is the use of electricity and heat which creates the largest impacts (especially for the categories climate change, ozone depletion and ionizing radiation). As the starch industry processes include drying or concentrating the final products, the use of heat is important, hence this result is logical. The contribution of the chemicals used in the processes to the overall impact is limited, although there are exceptions for some products. The impact of transport is relatively low, because the suppliers to the European starch industry are located close to the starch production plants and make use of train and ship when possible.

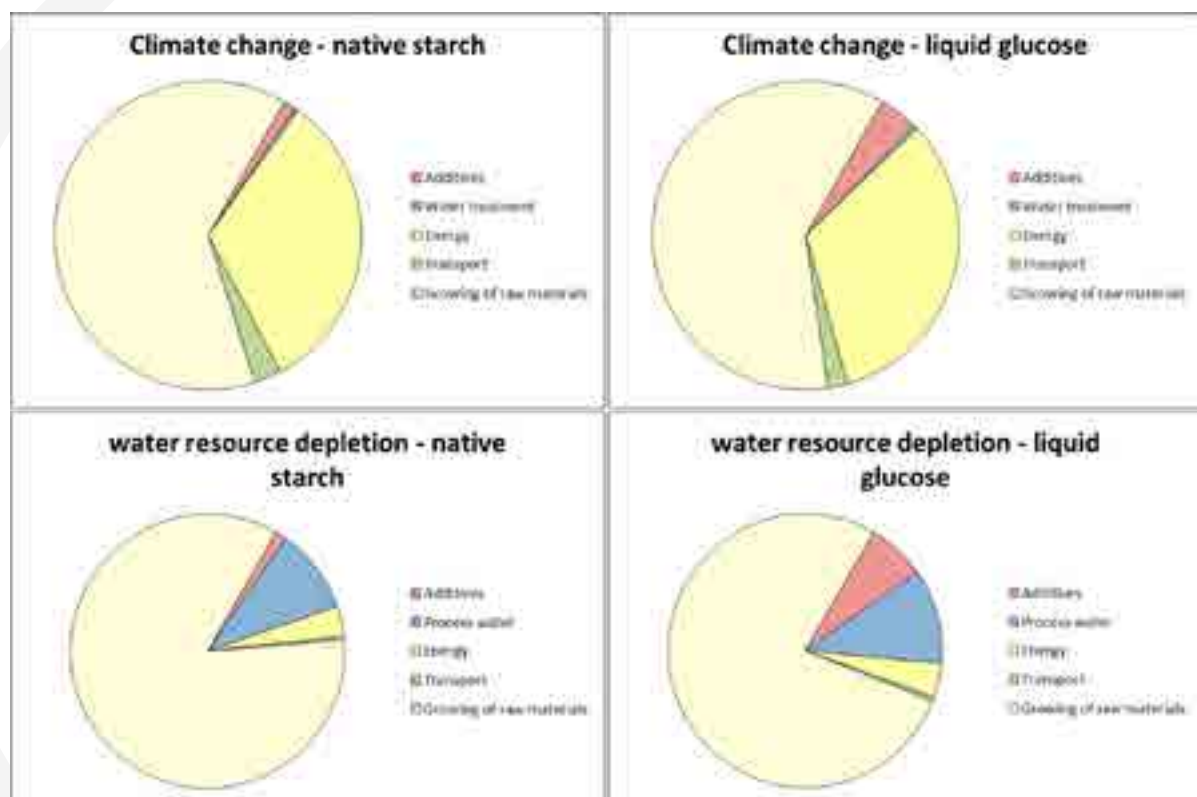


Figure 3: Relative contribution of different production inputs to the products environmental impacts – for water resource depletion, the use of water for cooling is also included

4. Discussion

From the analysis of compliance of the LCA study to the PEF Guide, it became clear that many aspects were already compliant or could be made compliant without a large effort. Others required a larger effort, and were solved when the added value for the study was considered large enough (e.g. using new agricultural data). For other aspects, compliance was not feasible (due to technical or budgetary constraints) or not desirable (not considered as an improvement to the study).

Lehmann et al. (2016) summarise the main challenges related to the PEF as follows: 1) open goal and intended policy application of the PEF process, 2) lack of visible added value of some PEF rules compared to current LCA-practice, 3) immaturity of several proposed LCIA methods and lack of a reliable method for prioritizing impact categories, and 4) the tendency to support reproducibility rather than comparability. In this study, the main issue encountered is the second one, since making the LCA fully compliant to the PEF Guide would require much work, which would not improve the quality of the LCA. For example, including all capital goods (even office equipment) would be a huge task, and probably not have any significant influence on the results. Also using the International Reference Life Cycle Data System (ILCD) nomenclature for the Resource Use and Emissions Profile would require a lot of administrative work without any influence on the outcome of the study. The other challenges concerning the PEF were not that relevant in this specific case. The LCA of starch industry products is a sector study, which does not have the objective to distinguish between producers, but to gain more insight in the environmental impact throughout the production chain and generate sector-representative environmental profiles to be used for B2B communication. The PCR can be used for both sector studies (based on weighted average data) and studies of a product of a specific company.

4. Conclusions

Some aspects of the LCA of starch industry products could be made compliant to the PEF Guide easily; while others were harder and sometimes not feasibly from technical or budgetary viewpoint; a few were feasible but not desirable for Starch Europe, since they did not seem to be the best choice for this LCA. Based on this analysis, it was decided to update some elements of the LCA, such as the data used for the agricultural production of the raw materials. Product category rules for starch industry products were composed in a process involving a large share of the EU starch industry.

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121. PEF in practice: learning from the supporting studies of the PEF FEED pilot

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ABSTRACT

In April 2013 the European Commission published a Communication to the European Parliament and the Council called 'Building the Single Market for Green Products: Facilitating better information on the environmental performance of products and organisations'. This communication introduces a method for measuring and communicating environmental performance throughout the life cycle of products, the Product Environmental Footprint (PEF) method. The Feed PEFCR (Product Environmental Footprint Category Rules) outlines how companies shall undertake PEF assessments for animal feed products. In order to assess the technical and practical validity of this draft Feed PEFCR, it is tested during the 'supporting study' phase (January – May 2016) of the PEF pilot project. In this phase, five feed companies applied the PEFCR to real products that exist on the European market. This paper presents the main outcomes of this supporting study phase. The feed PEFCR development is considered as very important by the European compound feed industry for harmonizing lifecycle assessment studies and communication of results. It is important that this is done in the overarching PEF framework where also the PEFCRs are developed for the animals that consume the feed. There is a big opportunity for cross sector harmonization of LCA rules for animal products. The supporting studies with the feed companies showed that the PEFCR is usable. Nevertheless further refinement in requirements is needed. The process also highlighted some needs which go beyond the PEFCR development such as streamlining of the PEF framework, development of supporting tools, improved data management systems and LCA training for companies.

Keywords: Product Environmental Footprint, Animal Feed, PEFCR application

1. Introduction

In April 2013 the European Commission published a Communication to the European Parliament and the Council called 'Building the Single Market for Green Products: Facilitating better information on the environmental performance of products and organisations'. This communication introduces two methods for measuring and communicating environmental performance throughout the life cycle, one for organisations (Organisational Environmental Footprint or OEF) and one for products, the Product Environmental Footprint (PEF) method.

The proposed PEF method is a framework of general requirements and principles to conduct environmental assessments of products. The European Commission aims to develop more specific technical guidance for specific product groups ('category rules') so that the main methodological choices in LCA studies can be standardised and are consistent across products of the same category. The Product Environmental Footprint Category Rules (PEFCRs) will standardise the use of primary data and background data. In addition, system boundaries, allocation rules and the method for calculating environmental impacts will be harmonised. Around thirty pilot studies have been selected for various product groups. Some examples in the food and agricultural sector are beer, dairy, olive oil and feed for food-producing animals.

The Feed PEFCR outlines how companies shall undertake PEF assessments for feed products. The relative importance of feed in the environmental footprint of animal products justifies the need to harmonize the methodology across all food-producing animals. There is no reason to develop different methodological approaches for the production of feed in lifecycle studies for pork, poultry meat, milk, fish etc. PEFCRs for food-producing animals should therefore use the same LCA method in relation to the feed that is used.

Feed is an intermediate product which composition varies depending on nutritional requirements of the animal, the desired characteristics of the animal product and the availability and prices of feed ingredients. Measuring the impacts associated with the production of feed, as well as the feed performance on the farm is necessary in order to achieve meaningful LCAs of food-producing animals. The feed performance in terms of production per unit of feed is closely linked with farm

management practices. In other words, the performance of the same feed used in two different farms can vary significantly according to the farm's specific conditions (animal breed, animal health status) and management. This is extremely important to bear in mind when considering the improvement of the environmental performance of feed production. Reducing the environmental footprint of feed production alone, without taking into account the potential consequences on its efficiency in the use phase would be counterproductive.

This means that to conduct an LCA at farm level, one needs information on the environmental impact of producing the feed and on the characteristics of this feed. The Feed PEFCR is meant to define the rules for deriving PEF compliant LCA information as an input for LCAs on food-producing animals. The Feed PEFCR is seen as a “module” to support the assessment of the environmental footprint of animal products in a harmonized “cradle-to-plate” way. It is not a stand-alone document answering all questions related to food production from animal systems. As a consequence of the cradle to gate approach chosen for the Feed pilot, the ‘on-farm’ feed efficiency will have to be measured when doing the assessment of animal products rather than predicted when assessing the feed.

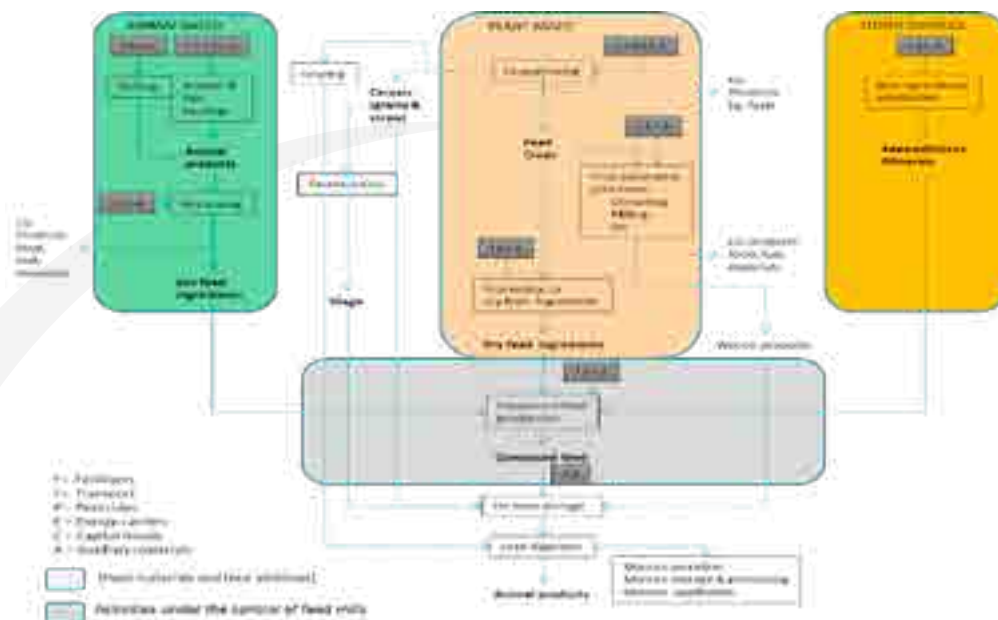


Figure 1: System boundaries of PEF studies using the feed PEFCR.

In order to assess the technical and practical validity of this draft Feed PEFCR, it is tested during the ‘supporting study’ phase (January – May 2016) of the PEF pilot project. In this phase, five feed companies applied the PEFCR to real products that exist on the European market. The learnings from this phase are outlined in this paper.

2. Methods

As mentioned in the introduction, the Feed PEFCR allows for cradle-to-gate assessments. However, it is possible to do assessments for different purposes:

- 1) Provision of LCI information of feed to PEF studies for food producing animals
 - a) without comparison nor external communication;
 - b) with comparison and/or external communication;
- 2) Cradle –to-gate PEF studies on compound feed for food-producing animals, without comparison
 - a) for internal use;
 - b) for external use/communication;

- 3) Cradle-to-gate PEF studies on compound feed for food-producing animals, with comparison
 - a) comparison between alternatives: such as, different manufacturing methods, evaluation of alternative feed configurations to the same or different nutrient profiles, feed ingredient sourcing and their manufacturing methods, on the basis of upstream life cycle emissions of feed ingredients and feed formulation;
 - b) comparison in time: monitoring trends/progress in environmental impact of feed products related to measures aimed at reducing environmental impact.

Different requirements apply according to the purpose of the PEF study. The rationale of this distinction in purposes is that methodology, accuracy and effort for data collection should be proportionate to the purpose of the study. For purposes that are about understanding differences between feed formulations such as performance tracking or evaluating innovations (purposes 3a and 3b) more detailed and restrictive requirements are set. In those cases the sensitivity to allocation choices need to be assessed. If an internal PEF study is conducted of a single feed product without external communication (1a) such a sensitivity assessment is not necessary. Also the level of use of primary data and required data quality is related to these purposes.

The PEFCR is based on the outcome of a screening study conducted on an average compound feed, whose composition was based on the consumption of feed ingredients by the EU compound feed industry. This screening study gave insight into the main drivers (due to emissions and resource use) of the environmental impact of the cradle to gate compound feed lifecycle. They are listed in Table 1.

Table 1: Most relevant flows, processes and activities for environmental impact of feed lifecycle

Most relevant elementary flows	Most relevant processes	Most relevant life cycle stage
Metals (Zn, Cu, Cr, Pb, Hg) relevant for human toxicity and eco-toxicity.	Fertilizer and manure application at farm	Raw material acquisition and primary processing
Pesticides mainly relevant for eco-toxicity scores	Farming	Raw material acquisition and primary processing
N relevant for eutrophication, acidification and climate change impact	Fertilizer and manure application	Raw material acquisition and primary processing
P relevant for eutrophication and resource depletion	Fertilizer and manure application	Raw material acquisition and primary processing
Land transformation	Soy bean growing in Argentina & Brazil	Raw material acquisition and primary processing
Land occupation	All crops	Raw material acquisition and primary processing
Water depletion	Irrigated crops in regions with water stress	Raw material acquisition and primary processing
Energy determines the impacts (photochemical smog formation, PM, Ionizing radiation) and has a relevant contribution to Climate Change, Acidification and Eutrophication	Energy production for, and/or fossil fuel use in processing, transport, field operations	Raw material acquisition and primary processing Distribution and transport Production of the main product

Most of the determining emissions and resource use of the compound feed lifecycle happen at processes in the supply chain (mainly cultivation) that are outside the scope of influence of most feed companies, and thus secondary data sources are needed. Still a minimum data quality needs to be achieved because of this high contribution. Only in case of a low contribution of processes that are outside the span of control low data quality could be acceptable.

The following processes are considered to be within the span of influence of a compound feed company, for which primary and accurate data need to be collected: 1) purchase of feed materials; 2) formulation of the compound feed; 3) operations in the feed mill; 4) delivery of the compound feed to the farm. Therefore, it was anticipated that the feed mill operator has specific knowledge of:

1. List of feed ingredients, and their origins;
2. Nutritional analysis data of compound feed (i.e. nutritional characteristics of the compound required for modelling of use phase of the feed on the farm);
3. Transport activity data related to procurement of feed materials, packaging and auxiliary materials from their suppliers. This transport information only concerns the last step of transportation from the supplier to the feed mill and not the transport of the supply chain of the supplier;
4. Activity data of the compound feed mill including use of energy carriers and potential on-site energy production, feed materials, packaging and auxiliary materials;
5. Transport activity data of delivering compound feed to farms.

For the supporting study phase, and the future use of the PEFCR, it was assumed that the user of this PEFCR has access to information from a specific feed manufacturer so that he has the necessary data on the list of feed ingredients, and nutritional composition. Moreover it was assumed that a feed manufacturer has knowledge, or access to knowledge, about the origins of purchased feed ingredients, and associated logistics.

The supporting studies are conducted to:

1. test the draft PEFCR for food producing animals
2. validate the outcomes of the screening study, such as the selection of relevant life cycle stages, processes and elementary flows.
3. perform supplementary analysis listed in the draft PEFCR
4. provide results that can be used as the basis for communicating the PEF profile

The focus of this paper is on the first 3 goals, as the communication phase is still ongoing.

The main purpose testing the draft PEFCR is to check if the requirements as formulated in the draft PEFCR are clear for the user and give sufficient support to conduct the PEF study unambiguously. In an ideal situation different users should get the same PEF results in the same situation. During the writing of the PEFCR, considerations on availability of data were taken into account. But data availability is different than access to data, meaning that feasibility needs to be tested. Another relevant question is whether the period for averaging data is the same for every animal type and feed type.

The second goal involves checking whether the results of the initial hot spot analysis of the screening are still valid. The PEF results of the supporting studies might reveal that in some specific cases the contribution of processes is different than the screening results. This is important in relation to the data quality requirements set in the PEFCR.

3. Results

Five different feed producing companies volunteered to conduct the supporting studies. The products that were analysed were different kinds of feed for different animal types, namely pigs, dairy cows, turkeys, salmon and broilers.

Different purposes were considered, companies did studies according to purpose 1b, 2b and 3a. During the process of the supporting studies, it became clear that the companies doing the studies were often facing similar challenges.

3.1 COLLECTION OF PRIMARY DATA

List of ingredients and origins

The current PEFCR requires considering the feed composition on a three-year period basis to achieve the highest data quality score. The rationale behind this time period is to be able to average over seasonal fluctuations driven by availability or price (rather than actual improvements). However, companies struggled to achieve this long timeframe for multiple reasons:

1. new product, with less than three years of existence
2. multiple variation of formulation , due to strategic purchasing decisions to optimize costs or due to varying quality of ingredients.
3. origins of the ingredients can vary a lot, making it a time consuming exercise to compile a representative list for three years. In several companies, purchasing information is not directly linked to formulation software, resulting in big efforts to collect the data.

Transport activity data

The feed producing companies also found it challenging to select the appropriate transport modes, types and distances. The complete transport information is not transmitted by the supplying companies, often only the last transport step is known.

Compound feed mill

The energy use of producing specific feeds is often not directly available, as multiple feed types may be produced on the same manufacturing line and some process steps are shared by multiple process lines (e.g. 2 grinders servicing three parallel mixing lines). The energy use is often measured on another level of aggregation, either by process line or on a factory level.

3.2 LINKING ACTIVITY DATA TO SECONDARY DATASETS

The European feed catalogue includes about 500 feed ingredients types, often with different sub-types. These ingredients may be purchased from a variety of countries (depending on the type of ingredient). Secondary data is not available for every ingredient used by the feed industry. Especially for fish meals and oils, additives and minerals, and new or special purpose ingredients, data availability is sometimes limited. When feed has a relatively large proportion of these aforementioned ingredients, the outcomes become less certain, and the data quality deteriorates.

3.3 IMPACT ASSESSMENT RELATED CHALLENGES

The companies performing the supporting studies often find it difficult to interpret the results and can draw unexpected conclusions when they are faced with the 15 pre-defined impact categories. The participants sometimes find it difficult to understand the meaning of the absolute values of the different impact categories, let alone communicate these outcomes to other stakeholders in their company and the supply chain. There is a risk that individual impact scores can be taken out of their context. For example a high score on human toxicity may be interpreted as if the feed is somehow toxic, while the impact originates elsewhere in the value chain and has no a relation to the chemical properties of the product itself.

Also understanding the various impact scores in relation to each other was challenging. For example is a low score for climate change more important than a low score for acidification?

Changing a raw material source to reduce climate change may increase acidification. So what is the 'optimum' composition for a feed? Unfortunately, the currently available normalisation and weighting sets (as provided in the PEF pilot) was not very helpful because of the uncertainty of both the characterisation factors as the data normalisation varies between impact categories.

4. Discussion

The feed PEFCR development is considered as very important by the EU compound feed industry for harmonizing lifecycle assessment studies and communication of results. It is also important that this is done in the overarching PEF framework where also the PEFCRs are developed for the animals that consume the feed. This is a big opportunity for cross sector harmonization of LCA rules for animal products.

The supporting studies with the feed companies showed that the PEFCR is usable. Nevertheless further refinement in requirements is needed. This process also highlighted some needs which go beyond the PEFCR development such as streamlining of the PEF framework, development of supporting tools, improved data management systems and LCA training for companies.

A PEFCR should to be very clear and unambiguous on primary and secondary data collection. And the current PEFCR needs improvement from that perspective. The procedure for selecting secondary data of feed materials and making proxies, if data are not available, requires a lot of attention. In the case of feed products this latter step can influence the results tremendously. This is poorly defined in the ISO 14026 standard. A closer look at existing PCRs for instance from the EPD system shows that there has not been much attention to this in the past. Basically, it was possible to use multiple sources of information. Without having a mandatory use of a harmonised secondary dataset that is consistent with the PEFCR requirements it is simply not possible to derive comparable and reproducible results.

Supporting tools are key to obtain a widespread implementation of a (feed) PEFCR. Supporting tools are foremost an easily accessible database to low or no costs and tools to use the generated information in environmental reporting and eco-design of feed materials. The role of the recently launched Global Feed LCA Institute (GFLI) is to develop these supporting tools.

The selection of environmental impacts in the PEF context requires a sound and understandable process. It is currently mandatory to calculate all 15 ILCD impact categories although it is acknowledged by the European Commission at the same time says that only 9 of them provide meaningful results at the moment. This approach is maybe defensible in a pilot phase, but not in the long run.

Finally, there is a need for a governance structure after the PEF pilot that makes it possible to maintain and improve the PEFCR and supporting tools. A PEFCR and its supporting tools should not be static but further developed in alignment with key stakeholders on a permanent basis. This is a prerequisite to create an effective single market for green products in Europe.

5. Conclusions

The PEFCR supporting studies led to many detailed suggestions on how to improve the draft PEFCR, which made the testing quite successful. The main amendments are in the area of data collection and the data quality requirements. It is essential that requirements on data collection and quality assessment are very detailed so that the user of the PEFCR can unambiguously collect data and report about data quality.

The preferred timespan for collecting feed formulation data shall be adapted in relation to the type of feed. For some feeds and study purposes the preferred time span is quite short while for others this is much longer. PEFCR requirements on this aspect need to be formulated in a way that covers these differences.

Detailed guidelines will be implemented to support the user in selecting secondary LCI data for feed materials as a proxy when the specific feed material or origin is not available in the secondary database. Furthermore, requirements will be implemented on how to present the PEF results based on the level of approximations in data collection and thus the level of uncertainty and data quality.

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242. The EU Organisation Environmental Footprint Sector Rules (OEFSR) for the Retailer sector

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ABSTRACT

The European Commission is soon finishing the pilot to create a “Single Market for Green Products”, that aims at facilitating better information on the environmental performance of products and organisations. 24 Product Environmental Footprint Category Rules (PEFCR) and 2 Organisation Environmental Footprint Sector Rules (OEFSR) are being drafted by stakeholders. The pilot drafting the OEFSR for the retail sector is composed by retailers (Carrefour, Colruyt, Decathlon, Kering, Office Depot, Picard, PERIFEM), public agencies (ADEME from France, EAA from Austria, and ENEA from Italy), one NGO (Global 2000), and one LCA consultant (Quantis).

As of February 2016 an assessment of the impacts of an average retailer has been performed. A first OEFSR has been drafted and accepted by the Steering Committee. The results of the assessment, the draft and its main methodological points (e.g., for direct, as well as upstream and downstream indirect contributions), as well as the benefits of this OEFSR for companies will be presented highlighting latest developments and feedback, including from the supporting studies. These points also include the issue pertaining to consistency with the product approach for a sector as interdisciplinary as the retail sector. As an example of results, an average general retailer supplying products for 3'000'000 people can have a carbon footprint of 10'000'000 t CO₂-eq per year, most of it being associated with the life cycle of its products sold.

Interaction between OEFSRs and PEFCRs such as cross cutting issues and consistency will also be addressed. As an example of methodological agreement that has been reached among sectors is how allocation among meat, milk, pet food and leather should be performed among cattle co-products. Such type of agreement is key for a sector like the retail to be able to consistently perform its Environmental Footprint.

One of the significant differences with traditional corporate footprint is that assessment and reporting for OEF goes beyond the traditional carbon footprint and includes 15 impact categories such as water footprint, pressure on resources as well as impact on human health through environmental pollution. Pressure on biodiversity or deforestation throughout the supply chain is also included.

Keywords: Life cycle assessment, Corporate, Company, Disclosure, Consumer products

1. Introduction

On April 9th 2013, the European Commission (EC) published: “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” (EC 2013).

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 EC 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 three years (official launch in November 2013), is composed by six retailers: Carrefour, Colruyt, Decathlon, Kering, Office Depot, and Picard; one retailer association: PERIFEM; three public agencies: French Environment and Energy Management Agency (ADEME), Environment Agency Austria (EAA), 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 for the Retail sector encompasses the four following principles: (1) life cycle-based approach; (2) multi-criteria; (3) physically realistic modelling; (4) reproducibility /comparability.

This pilot tests how approaches such as Product Environmental Footprint Category Rules (PEFCRs) and the “Chain OEF” (approach covering the indirect/upstream part of the value chain) 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.

An OEFSR can have several goals (Figure 1): from purely informative use within the company to use outside of the company, to communicate with the stakeholders. Comparative assertion, though theoretically feasible, is not meant to be done with the OEFSR for Retail.

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	✓		

Figure 1: Goal 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.

The system boundary of the OEFSR (Figure 2) comprises of an organizational boundary (only organization itself) and an OEF boundary (including upstream and downstream activities). When adapted to the retail sector (Figure 3), the system boundary typically consider seven life cycle stages: Product production and service provision; Logistics; Retail place; Distribution of product sold; Use of product sold; End-of-life of product sold; and Support of the entire retailer activities.



Figure 2: System boundary of the OEFSR in general



Figure 3: System boundary of the OEFSR adapted to the retail sector

The categories of product typically captured by retailers are:

- food
- beverage
- tobacco
- fruit and vegetables
- meat and meat products
- fish, crustaceans and molluscs
- bread, cakes, flour and sugar confectionery
- automotive fuel
- information and communication equipment
- textiles
- hardware, paints and glass
- carpets, rugs, wall and floor coverings
- electrical household appliances
- furniture, lighting equipment
- cultural and recreation goods
- clothing
- footwear and leather goods
- dispensing chemist
- medical and orthopaedic goods
- cosmetic and toilet articles
- flowers, plants, seeds, fertilisers, pet animals and pet food
- watches and jewellery

3. Results

An OEF screening was conducted for a virtual retailer (providing the full range of retailing activities for about 3'000'000 persons) for one year of its activities, including – where relevant – its product portfolio life cycle impacts. Figure 4 shows the contribution to the different impact categories for the virtual retailer, over the full OEF.

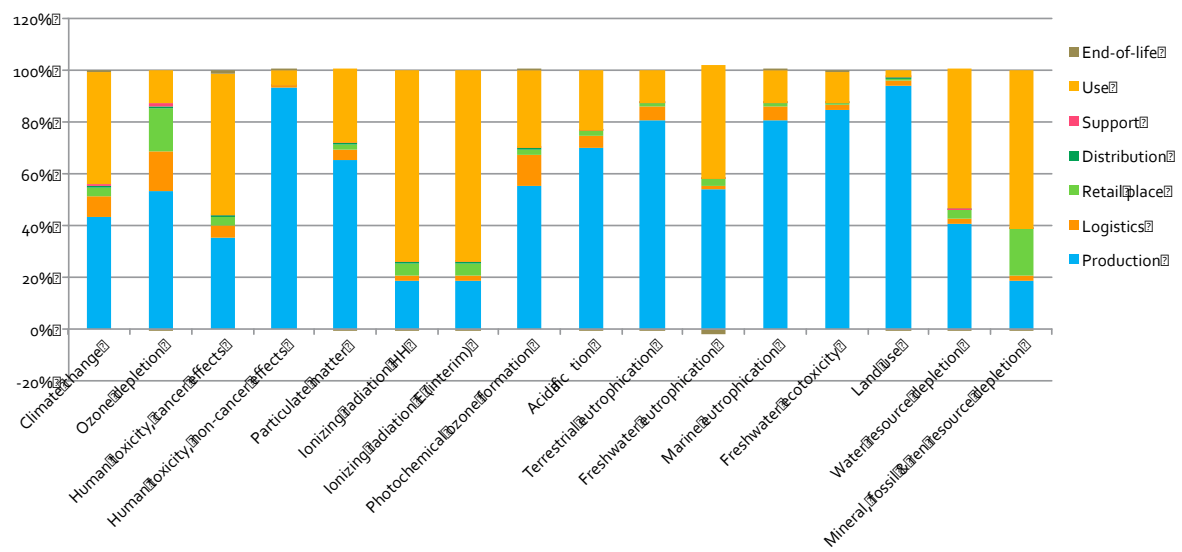


Figure 4: Contribution to the different impact categories for our virtual retailer, over the full OEF

The production stage impact on climate change for one year of retailer activity is detailed in Figure 5.

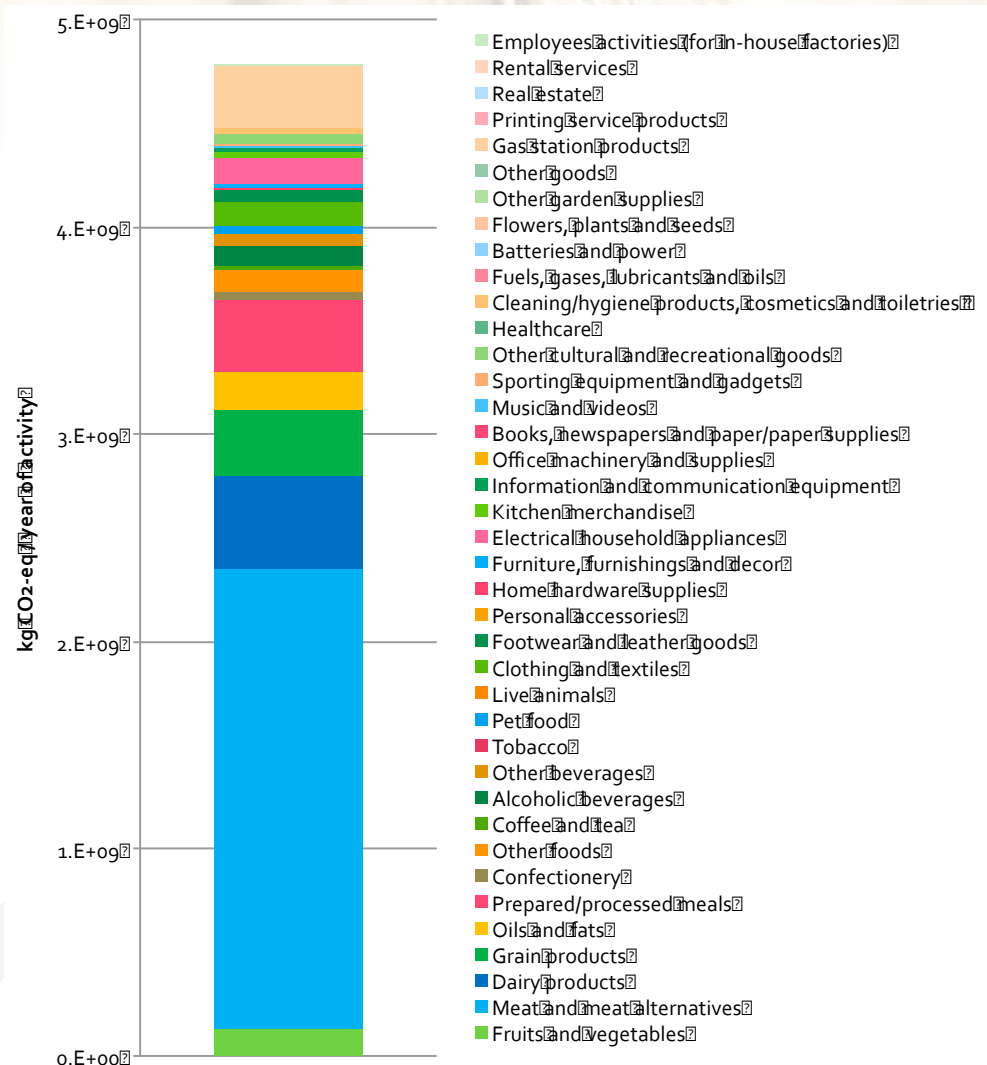


Figure 5: Production stage impact on climate change for one year of retailer activity

4. Discussion

The total impact on climate change of the representative retailer studied corresponds to 11'000'000 tonnes CO₂-eq per year. As a matter of comparison, to put that in the context of overall environmental impacts of Europe, this equals to 3.7 tonnes CO₂-eq per consumer (considering the 3'000'000 consumers for this retailer), i.e., about 1/3 of the impacts of an average European person (using for example, the normalization factor for the carbon footprint impact category provided by JRC to be used in PEF (JRC 2015) or by IMPACT 2002+ (Joliet et al. 2003)).

Key learning for the OEFSR retail are:

- as soon as a retailer has a non-negligible part of its activity related to animal products, attention should be given to model this activity properly, and
- consistency modeling among products is a challenge when background database are not complete and have to be mixed → the simplified models created for several products for this screening could be used by practitioners while waiting background LCI.

The draft OEFSR developed in this pilot can be used by any retailer to assess its annual impact in a simplified way (the document provides lots of guidance and default data – Figure 6) and in a consistent way with other retailers.

An example of default data that are provided by the draft OEFSR in case no primary data are available (in this case for the transport from the retailer to the consumer) is presented in Figure 6.

7. Distribution from retail to clients

Distribution consists in the transport of goods from retail places to home and can be done either by the consumer or by the retailer.

When looking from organization perspective, if number of total customer visit throughout the year is known, this value should be used for the number of trips.

When looking from a product perspective, it is assumed that for:

- retailers selling lots of small products (typically food based supermarket), 20 items are bought per trip and therefore 1/20 of the trip is allocated per item or retail unit.
- retailers specialized in specific medium to large size or value products, 3 items are bought per trip for medium size or value products (e.g. sports or IT equipment) and 1 item per trip for large size or value (e.g. fridge or luxury watch or diamond).

The distribution to clients is assumed to be done as follows:

If done by the person (client) himself/herself:

- 80% by car on 4.8 km distance (roundtrip) (meaning 3.84 km of car) (Note that for retailers mainly present in urban centers, the fraction of 80% by car can be significantly lower)
- the remaining part is assumed to be done by bus and walking; as a matter of simplification, the impacts for this part is neglected (as it would not change significantly the value provided by 80% of the persons shopping by car)

If done by the retail ("home delivery"):

- distribution by van with 4.8 km (roundtrip) allocated per delivery (of 20 items, which means 240 m of van per item)

When it is unknown whether the distribution is done by the client or by the retailer, an average distribution scenario can be done:

- 80% by car on 4.8 km distance (roundtrip) (meaning 3.84 km of car for 20 item) (meaning 192 m by car per item)
- 5% by van (with 4.8 km (roundtrip) allocated per delivery of 20 items) (meaning an additional 5% of 240 m of van per item)
- 15% by bus or walking; which is neglected as a matter of simplification

Figure 6: Example of default data provided by the draft OEFSR in case no primary data are available

5. Conclusions

The project, challenges, 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.

6. References

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66. The Product Environmental Footprint Category Rules (PEFCR) Packed Water Pilot Testing

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ABSTRACT

Within the European Commission “Single Market for Green Products” initiative, a Technical Secretariat which is responsible for developing the Product Environmental Footprint Category Rules (PEFCR) for Packed Water was selected and launch in July 2014.

It is composed of four federations: the European Federation of Bottled Waters (EFBW), the European Container Glass Federation (FEVE), Petcore Europe, and the Union Européenne des Transporteurs Routiers (UETR); five natural mineral water producers: Danone Waters, Ferrarelle, Nestlé Waters, San Benedetto (since July 2015) and Spadel; and one Life Cycle Assessment (LCA) consultant: Quantis.

The objectives of the pilot phase are: i) to set up and validate the process of the development of product group-specific rules; ii) to test different compliance and verification systems; and iii) to test different B2B and B2C communication vehicles for Environmental Footprint information in collaboration with stakeholders.

Preparing a PEFCR encompass: i) definition of PEF product category and scope of the PEFCR; ii) definition of the product “model” based on representative product(s); iii) PEF screening; iv) draft PEFCR; v) PEFCR supporting studies; vi) confirmation of benchmark(s) and determination of performance classes; vii) final PEFCR.

The project, results, challenges, and expected 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 specific as the packed water sector (e.g., three sub-categories of application have been defined in the scope: “at horeca” (i.e., hotel, restaurant and café), “at the office” and “other channels”).

Keywords: Life cycle assessment, Mineral water, Bottled water, Disclosure, Labelling

1. Introduction

On April 9th 2013, the European Commission (EC) published: “Communication from the Commission to the European Parliament and the Council: Building the Single Market for Green Products (SMGP), facilitating better information on the environmental performance of products and organisations” (EC 2013).

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 organisations has been selected by the EC to develop the guidance for calculating and reporting environmental impacts of packed water. The Technical Secretariat which is responsible for developing the Product Environmental Footprint Category Rules (PEFCR) for Packed Water was launch in July 2014. By 2016, it was composed of four federations: the European Federation of Bottled Waters (EFBW), the European Container Glass Federation (FEVE), Petcore Europe, and the

Union Européenne des Transporteurs Routiers (UETR); five natural mineral water producers: Danone Waters, Ferrarelle, Nestlé Waters, San Benedetto (since July 2015) and Spadel; and one Life Cycle Assessment (LCA) consultant: Quantis.

According to the European Commission, the objectives of the pilot phase are: i) to set up and validate the process of the development of product group-specific rules; ii) to test different compliance and verification systems; and iii) to test different B2B and B2C communication vehicles for Environmental Footprint information in collaboration with stakeholders.

2. Methods

The approach used to develop the PEFCR for the Packed Water category encompasses the four following principles: (1) life cycle-based approach; (2) multi-criteria; (3) physically realistic modelling; (4) reproducibility /comparability.

The following steps have been followed when preparing the PEFCR: i) definition of PEF product category and scope of the PEFCR; ii) definition of the product “model” based on representative products; iii) PEF screening; iv) draft PEFCR; v) PEFCR supporting studies; vi) confirmation of benchmarks and determination of performance classes (optional); vii) final PEFCR.

Figure 1 presents the goal of the PEFCR.

Application	Use of a PEFCR		
	Optional "May"	Recommended "Should"	Mandatory "Shall"
Any PEF study not used in B2B or B2C communication (e.g. for in house improvements) declared to be in compliance with the PEF Guide	✓		
B2B or B2C green claim without comparisons or comparative assertions		✓	
B2B or B2C green claim with comparisons or comparative assertions			✓

Figure 1: Goal of the PEFCR (scenarios that do and do not necessitate the use of PEFCRs)

The draft PEFCR for packed water uses a default functional unit of 100 ml of water “at the mouth”. Water is assumed to be chilled.

Figure 2 presents the system boundary of the PEFCR adapted to the packed water category.

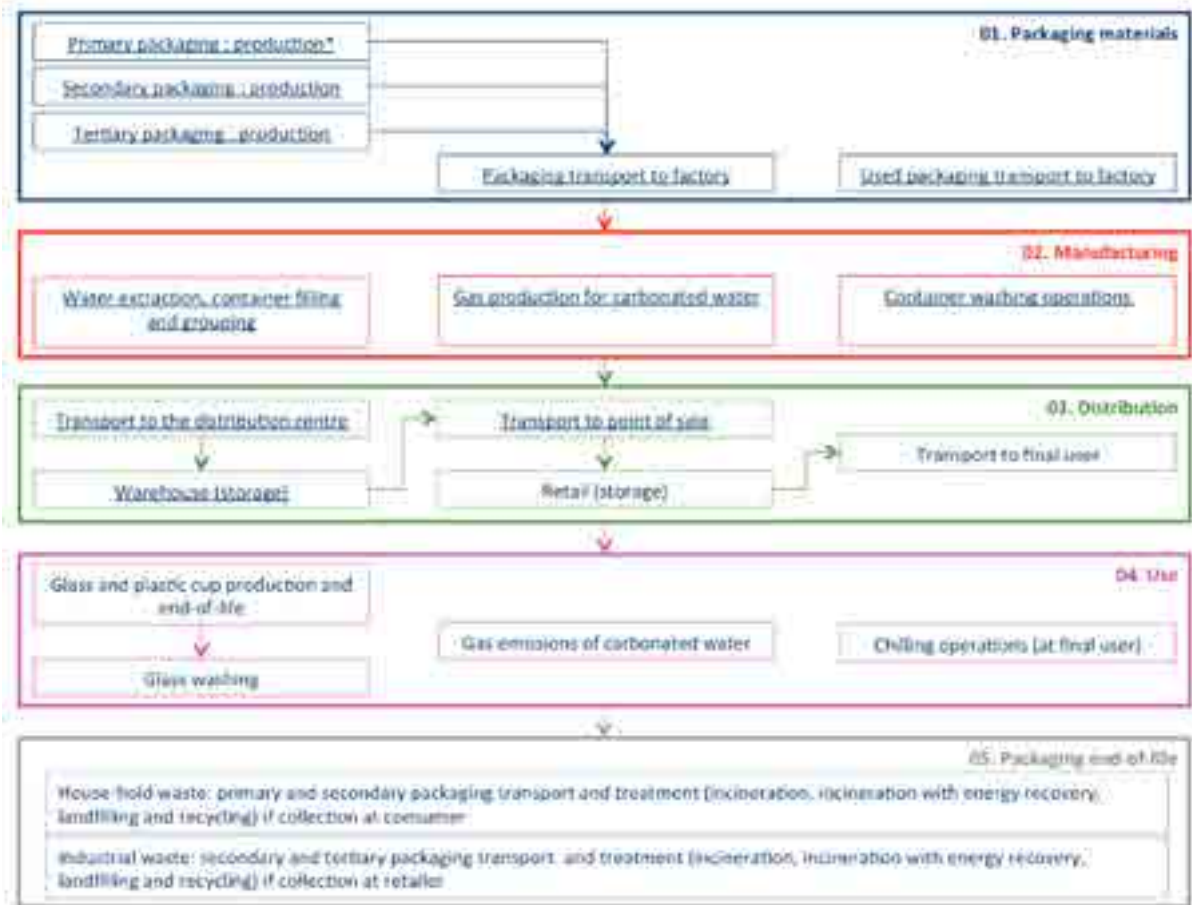


Figure 2: System boundary of the PEF CR adapted to the packed water category (processes being part of the foreground system are underlined).

Three sub-categories of application have been defined: “at horeca” (i.e., hotel, restaurant and café), “at the office” and “other channels” (Figure 3).



Figure 3: Illustration of the sub-categories (blue) and the representative products (green)

3. Results

A PEF screening was conducted for each of the three representative products. As an example, Figure 4 shows the contribution for each impact category with relative contributions of the different life cycle for the representative product “Other channels; PET one-way 1.5L”.

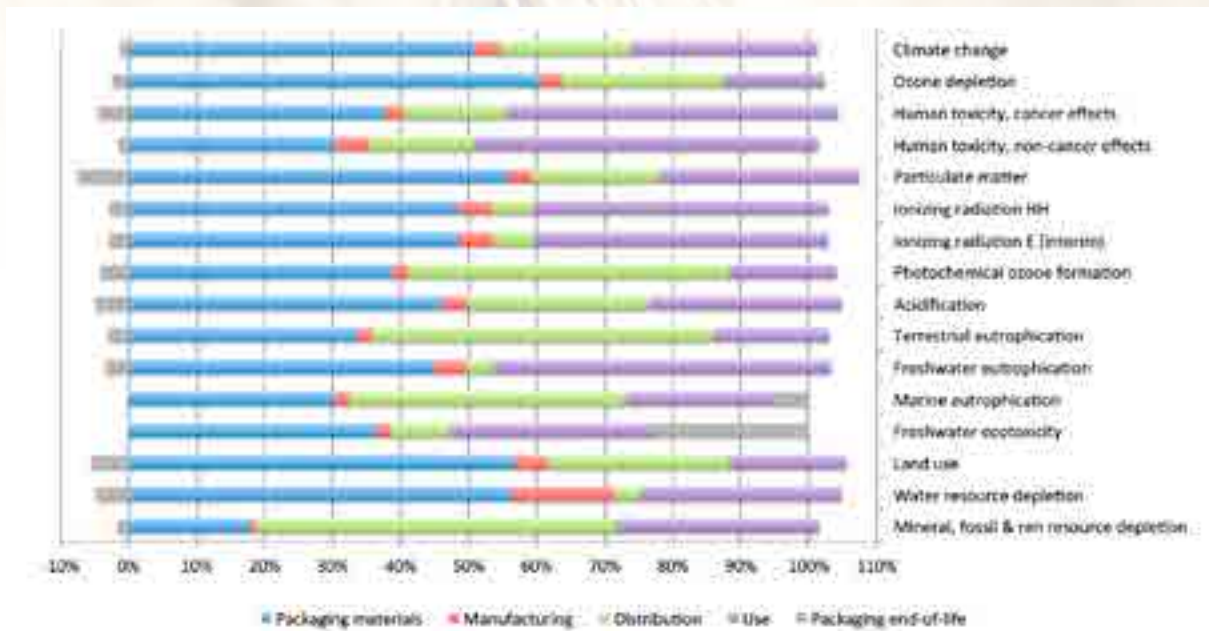


Figure 4: Contribution to the different impact categories for for the representative product “Other channels; PET one-way 1.5L”. The full list of results can be found on the documents available on the pilot website (accessible from the SMGP website (EC 2013)).

Figure 5 presents the carbon footprint for each representative product, considering the functional unit, defined as being 100 ml of water drunk.

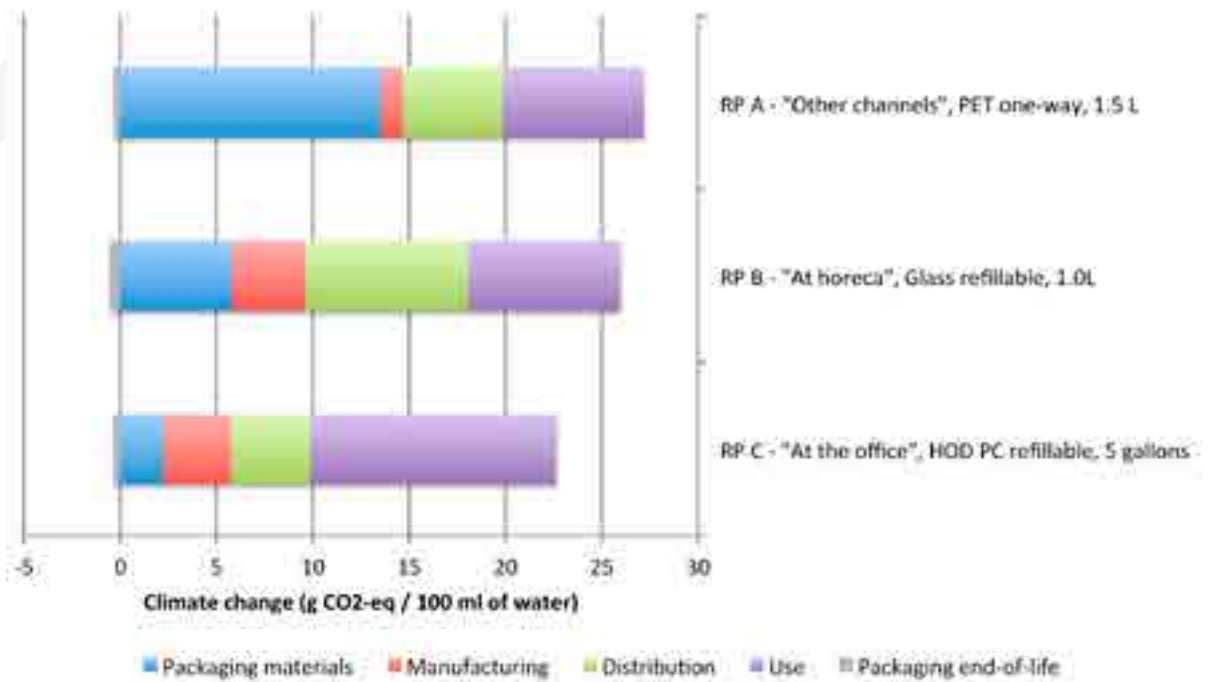


Figure 5: Carbon footprint for each representative product. The results are presented per functional unit, defined as being 100 ml of water drunk.

4. Discussion

The carbon footprint of 1 dl of packed water (drunk) range from 20 to 30 g CO₂ eq. When expressed by liter, it ranges between 200 and 300 g CO₂ eq.

- The most relevant processes (per life cycle stage) are:
- o Primary, secondary and tertiary packaging production;
 - o Water extraction, container filling and grouping; Container washing operations;
 - o Transport from water factory to distribution center; from distribution center to point of sale; and from retailer to final user;
 - o Glass or plastic cup production and end-of-life; Glass washing and Chilling operations (at final user);
 - o House-hold waste: primary and secondary packaging transport and treatment; and Industrial waste: tertiary packaging transport and treatment.

The draft PEFCR developed in this pilot can be used by any packed water producer to assess the environmental impacts of its product. This draft contributes to making the assessment easier (the document provides lots of guidance and default data – Figure 6) and in a consistent way with other companies.

Figure 6 presents an example of default data that are provided by the draft PEFCR in case no primary data are available (in this case for the transport modeling).

Table 15 Key information to be used for modelling truck operations :

Transport type	Net vehicle weight (tonnes)	Fuel consumption (kg/100 km)	Load (tonnes)	Dataset for direct emissions related to the combustion of 1 kg of diesel
40 tonnes, full	18	29.4	27.0	Articulated lorry transport, Euro 0, 1, 2, 3, 4 mix, 40 t total weight, 27 t max payload RER (ELCD)
40 tonnes, empty	18	19.1	0.0	Articulated lorry transport, Euro 0, 1, 2, 3, 4 mix, 40 t total weight, 27 t max payload RER (ELCD)
3.5-20 tonnes, full	7.5	20.0	5.4	Lorry transport, Euro 0, 1, 2, 3, 4 mix, 22 t total weight, 17,3 t max payload RER (ELCD)
3.5-20 tonnes, empty	7.5	15.7	0.0	Lorry transport, Euro 0, 1, 2, 3, 4 mix, 22 t total weight, 17,3 t max payload RER (ELCD)

Table 16 Key information to be used for modelling truck infrastructures :

Transport type	Dataset to be used for modelling truck infrastructure excluding operations :
40 tonnes	Transport, lorry >28t, fleet average/CH (ecoinvent V2.2) excluding Operation, lorry >28t, fleet average/CH U (ecoinvent V2.2)
3.5-20 tonnes	Transport, lorry 3.5-20t, fleet average/CH (ecoinvent V2.2) excluding Operation, lorry 3.5-20t, fleet average/CH U (ecoinvent V2.2)

Figure 6: Example of default data provided by the draft PEFCR in case no primary data are available

5. Conclusions

The project, challenges, 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 the packed

water sector such as defining system boundaries and choosing life cycle impact assessment categories (e.g. which indicators are relevant, which weighting scheme to use).

6. References

EC 2013. <http://ec.europa.eu/environment/eussd/smgp/>

243. How to reconcile eco-design and eco-labelling in LCI database construction? AGRIBALYSE experience and links with database harmonization initiatives

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ABSTRACT

Until recently, food Life Cycle Inventory (LCI) data were relatively rare. As more food LCI data sets are being released, we enter a new phase where, at the same time data coverage remains to be improved, but also data consistency and database harmonization become real challenges so that users can access LCI data covering the large diversity of agricultural systems and products.

In this article, we discuss how LCI databases should be built, considering the different kind of uses they serve. We argue that the databases that are more ecodesign/upstream oriented such as AGRIBALYSE, will go towards increasing modelling accuracy and complexity, whereas other databases that are more focused on downstream users and labelling schemes may prefer more simple approaches and models, and easier repeatability. We propose a solution that would enable combining these data with different scales and accuracy, so that harmonization does not lead to overall lower quality for users. This scheme could be used for instance in projects such as the Product Environmental Footprint (PEF).

Keywords: Food and Agriculture, Life Cycle Inventory, modeling scale

1. Introduction

AGRIBALYSE is a public Life Cycle Inventory (LCI) database containing data for a large number of French agricultural products (www.ademe.fr/agribalyse) (Colomb et al. 2015). It aims to promote both eco-design and eco-labelling in agricultural and food systems and is used by several hundreds of Life Cycle Assessment (LCA) practitioners in France and abroad.

LCA is a framework that requires its practitioners to adjust each study to its “goal and scope”. However, for database providers the final use of the data is largely unknown. Therefore, most database developers do not claim any specific goal, or stick to very general ones such as “supporting eco-design and environmental information”. As data quality improves and new methods for emission modelling are being developed, the question of harmonization of the AGRIBALYSE database with other LCI agri-food databases is raised. Indeed, the food system is largely globalised, and it is unlikely that a single LCI database will be able to cover the diversity of all production systems in all countries regarding soil, climate and socio-economic conditions. Consequently, in the interest of the LCA community, we are convinced that LCI databases should aim for complementarity and avoid overlaps in order to progressively improve coverage of agricultural systems worldwide.

Therefore in this paper we discuss

- which strategies can be implemented so that existing LCI databases can be used in a complementary way.
- whether it is feasible or not to use the same LCI databases to support contrasting goals such as environmental labelling schemes and eco-design strategies.

2. Users with different positions in the food life cycle

LCI databases such as AGRIBALYSE are used by different stakeholders along the food chain (Table 1) Their field of expertise and access to primary data also differ. The closer they are to the farm stage, the more sophisticated emission models they will want to implement and the more detail on agricultural production practices they will dispose of, and the more options concerning production practices they may want to consider. On the contrary, downstream players (retailers, restaurants etc.) are likely to be mainly interested in eco-labelling, where the focus is on the choice of foods rather than on the optimization of farming practices. They will prefer the implementation of relatively

simple models to estimate pollutant emissions at farm level. Their priority is having the widest coverage for the foods they are using, and being able in a simple way to add new foods to account for their main characteristics (localisation, season of production, main labels on the market such as organic). These simple approaches can contribute to quickly enlarge databases for global coverage of main food products, supporting eco-labelling and changes towards more sustainable diets. However such data will not sustain the improvement of agricultural practices.

Table 1. Main aims of AGRIBALYSE data users along the food supply chain (not covering all possible use of LCI data).

Users	Aim	Level of detail required in LCI data
Agronomist/zoo-technician	Improve farming systems based on implementation of innovative agronomic practices, and by comparing to benchmark	+++
Food industry, food processing, R&D	Improve food products by modifying the ingredient composition/recipe	+ or ++
« Full production chain »	Implementation of a full eco-design strategy by a sector/branch, from farming practice to logistics and packaging, including communication.	++ to +++
Food industry, retailers	Communicate on improvement of a product compared to an existing/competing product or other benchmark.	+ or ++
Retailers	Environmental labelling scheme: providing the environmental performance data for a large range of food products	+ or ++
Nutritionists, NGOs	Work on sustainable diets at national scale, links between nutrition and environment.	+
Catering and restaurants	Working on sustainable diets and dishes for out of home catering	+
Research, policy makers	Studies on citizen's consumption footprints and prospective strategy, assessment of the effect of policy schemes	+
Research, policy makers, industry	Supporting new sectors based on environmental efficiency: example: compare bio-based products to fossil-based products	From ++ to +++

3. Agricultural production stage: being able to account for environmental improvement and eco-design strategies

The modification of agricultural systems and practices can strongly contribute to eco-design of food products, as the farm stage is a major hotspot for many impact categories. To begin with, an accurate picture of the most common production systems is necessary to provide a reliable benchmark for improvement solutions. National benchmarks are a good start, but not necessarily sufficient, especially for very large countries and in countries with high soil/climate diversity such as France. Once available, improvement options can be looked for.

Accurate accounting of the environmental consequences of changes in agricultural systems requires sophisticated, dynamic and data-intensive emission models. Indeed, many improvement options, especially those based on agro-ecological mechanisms, are designed at the cropping sequence scale rather than at the single crop scale (Willmann et al, 2012, Nemecek et al, 2008). To identify the most promising options, it is for instance paramount to be able to distinguish between different fertilization options (e.g. mineral, crop residues, manure, compost, digestate, sludge etc.), to accurately account for irrigation techniques, and to consider the consequences of farming practices in a given

environmental context (soil, climate, previous crop etc.). Similar reasoning applies to animal production, where it is crucial to accurately account for herd management (productivity, mortality, duration of fattening, time spent outdoors in pasture or yards, etc.), feeding strategies (composition of feeds, origin and production mode of feed ingredients, input levels and yields, etc.) and manure management systems (type of building and storage, biogas production) (Gac et al. 2007).

All these farming practices will affect emissions. Simple approaches such as IPCC Tier 1 and even Tier 2 will ignore many of the effects of these practices and potentially ignore improvement options to reduce direct emissions. For example, in the ECOALIM project (Wilfart et al, 2015), which was part of the AGRIBALYSE program, different models were tested to assess nitrate leaching for different scenarios: 1) no cover crop during the intercrop period following the crop under consideration, 2) presence of a cover crop or oilseed rape during this intercrop period. The IPCC tier 1 default emission factor ignored the reduction of nitrate leaching due to the cover crop (Fig. 1), which has been largely demonstrated by validated mechanistic models (Indigo: Bockstaller et al, 2008; Syst’N: Parnau deau et al, 2012). While such mechanistic models are more sensitive to farming practices, their application, in particular at a large spatial scale, is also a lengthy and data-intensive process. The AGRIBALYSE model for nitrate leaching (Koch and Salou, 2015) was not the most accurate model at the field scale, but provided a satisfying ranking of the different situations and coherent average nitrate leaching values. This analysis is in line with recent publications (Peter et al. 2016, Ponsioen and van der Werf 2016), confirming that while more sophisticated emission models require more input parameters and consequently more data collection efforts, they also brings useful added value to identify and promote more sustainable agricultural practices. Once these solutions are clearly identified, then simplified indicators can be used in ecodesign strategies, ensuring significant environmental benefits.

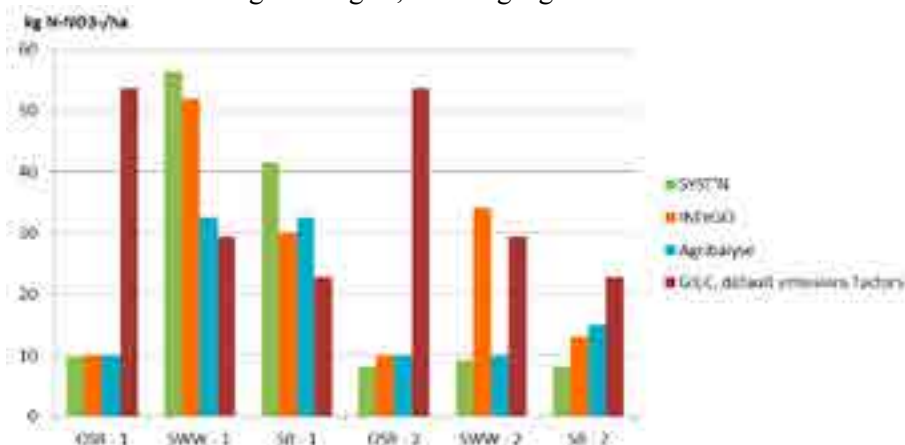


Figure 1. Estimated nitrate leaching by four models for a case study in western France (ECOALIM project). OSR: oilseed rape, SWW: soft winter wheat, SB: spring barley. Scenario 1: no cover crop, scenario 2: presence of cover crop.

4. Database harmonization, where to draw the line?

Since AGRIBALYSE wants to support eco-design of French farming systems, it tends to integrate detailed farmer production practice data and to implement increasingly sophisticated emission models. However, this degree of model sophistication may pose a problem for the integration of AGRIBALYSE LCIs in international databases/frameworks, such as the Product Environmental Footprint (PEF), the World Food LCA Database or Agri-footprint, which tend to promote less data-intensive emission models.

We propose a solution to this dilemma. Just as a variety of characterization methods can be used to produce different sets of impact indicators for a given LCI data set (Fig. 2a), several sets of emission models (corresponding to different objectives) can be used to produce different LCI data from a given data set of farmer practices and soil and climate data (Fig. 2b).

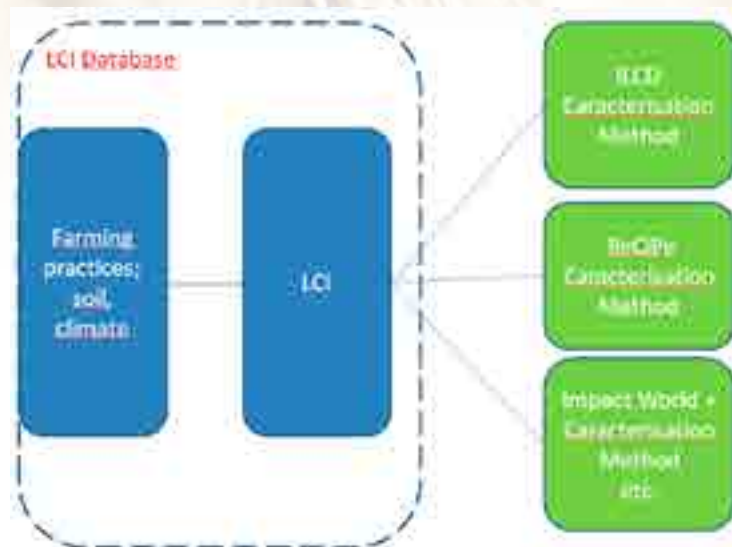


Figure 2a. A variety of characterisation methods can be used to produce different sets of impact indicators for a given LCI data set.

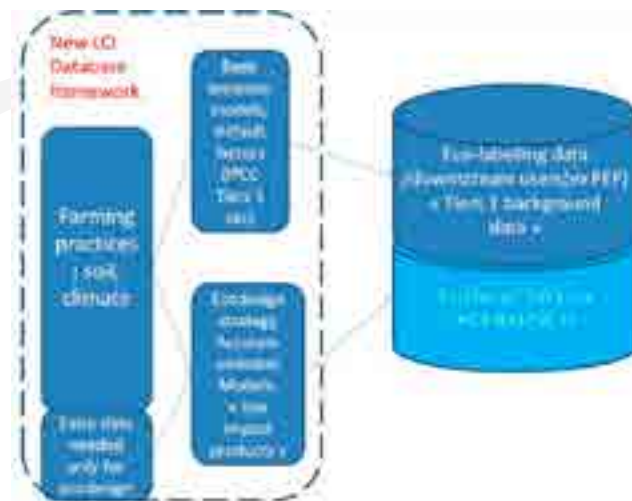


Figure 2b. Several sets of emission models can be used to produce different LCI data from a given data set of farmer practices and soil and climate data.

From a practical point of view, this solution can be implemented in the MEANS-InOut software platform (INRA 2016), where users will be able to choose which set of emission models they wish to implement. Such software solutions can help users in implementing the more complex models through a user-friendly data capture interface, also limiting risks of errors.

While some flexibility due to different goals from database developers and users seems justified, criteria not related to scale or data accuracy should be harmonised. Database harmonization should not lead to lower overall quality of LCIs. In general, heterogeneity is acceptable only when it allows important time saving or avoids data gaps, and when those data are not required for all kinds of users (Table 1). Table 2 summarises our view on harmonisation requirements for food LCI databases.

Table 2. Proposed harmonisation requirements for characteristics of LCI databases.

Characteristic	Harmonisation required	Comments
Scope	Yes	Scope should be similar, even if for minor inputs basic assumptions can be enough (ex : infrastructure)
Time-related coverage	Yes	Some 3 to 5 years average should always be considered to avoid atypical results due to climatic variability
Allocation to co-products	Yes	Provide a standard default option, but give the possibility to modify
End of life	Yes	No reason for heterogeneity, not a hotspot for food products
Background LCI database	Yes	Choice of background LCI database (ILCD or different versions of ecoinvent) will affect results. Unit processes should be used to allow switching background databases
Direct emission modelling	Not necessarily	From Tier 1 to Tier 3 approach, depending on database strategy
Accounting for crop sequences and their consequences	Not necessarily	Irrelevant for downstream users, useful for ecodesign
Fertilization practices	Not necessarily	For downstream users, N input is sufficient, more detail is necessary for eco-design strategies
Manure management and feed practices	Not necessarily	For downstream users a simplified representation is sufficient, full detail is required for eco-design
Data for key parameters	Not necessarily	For key parameters (such as yield, N input etc.), data sources should be based on best data available (national statistics for some countries, FAO for others etc.).
Data quality rating	Yes	ILCD rating system seems a good starting point
Naming	Yes	Consistent naming of LCIs is necessary.
Formats	Yes	All informatics barriers should be removed as soon as possible

5. Link between characterization methods and LCI databases

Life Cycle Impact Assessment (LCIA) is not directly within the scope of LCI databases. LCI databases provide flows which can theoretically be connected to any characterization method in LCA software, as long as the substance names are developed correctly. So far, most LCI databases try to provide all relevant flows for the main characterization methods (ILCD, ReCiPe, etc.). However, as characterization methods become more comprehensive (ex : water scarcity indicators, more biodiversity indicators in the future ?), including new flows at the LCI database level can become a real challenge, considering that the new flows must also be completed in all background processes for the new indicator to be fully operational. Cooperation between databases developers can be really useful on this topic. Also, it is inevitable that a significant delay will remain between LCIA developments, and their full implementation in LCI databases. One strong side of LCA is that it can assess all kind of processes and economic sectors. To keep this flexibility, we think that LCI

databases should try to remain as complete as possible regarding flows, leaving the possibility and responsibility to users to choose the most relevant characterization method for each situation.

6. Conclusion: who can do more can do less?

Until recently, food LCI data were relatively rare. As more food LCI data sets are being released, we enter a new phase where simultaneously, data coverage should be improved, and data consistency and database harmonization will become real challenges. Considering the difficulty of defining a clear “goal and scope” for databases, we propose to accept that full harmonization of databases is not necessarily a target. The focus should rather be on transparency and repeatability. Database developers should be encouraged to clearly state their priority, and whether their methodology is more appropriate for eco-design (including the farm stage) or environmental labelling. In our view, heterogeneity between databases is only acceptable for parameters related to data accuracy and spatial scale (ex: direct emission modelling). On the contrary, heterogeneity is not acceptable on parameters not linked to spatial scale or quality of emissions, i.e. methodological choices such as scope, allocation or data quality rating. This approach would enable users to (a) benefit from high quality data when available, those being required mainly for ecodesign and by upstream users, (b) and at the same time to have a broad range of data to cover the diversity of food and origin for downstream users. Since AGRIBALYSE aims to support eco-design strategies, it will probably implement more complex methodologies and emission models compared to other databases that are more focused on eco-labelling, and looking for broader coverage and easier repeatability. While extra efforts required for the development of databases to support eco-design strategies may seem costly, they are essential to guide changes in farming practices. User-friendly software and database tools will allow flexibility in database development based on the principle that who can do more can do less.

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68. Credible LCA Communications: the yellow brick road to building budget and buy-in

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ABSTRACT

The objective of this paper is to share a commonly used communications framework and provide examples of how to effectively and credibly communicate life cycle assessment (LCA) results to different stakeholder profiles using this framework. In short, one should first identify the positioning of the message, compile one's evidence (i.e. your LCA), then define the intended purpose, audience, content, channel, and measure of success. Using this framework will result in effective communication of life cycle thinking and results. The implications for LCA practitioners are vast: learning how to present credible results beyond the spreadsheet, creating value for the LCA work, getting bigger budgets to embed LCA into organizations' strategies, and increasing supplier and other stakeholder engagement.

Keywords: communication, framework, canvas

1. Introduction

Life cycle assessment (LCA) practitioners are faced with the challenge of creating understanding for the life cycle thinking process among their internal and external stakeholders. Internally, it is essential to generate understanding of the benefits of life cycle thinking and LCA to build executive buy-in as well as multi-functional cooperation. According to the MIT Sloan Management Review, when it comes to being informed about sustainability issues and efforts, "there is a lack of communication within corporations and investment firms and between them" (Kiron et al 2016).

Similarly, companies are struggling to communicate effectively with consumers about sustainability issues. According to the Boston Consulting Group's (BCG) report on Social Responsibility (Huet et al 2014), while consumers care and want to know that the products they purchase are responsibly made, there is rising consumer skepticism about green product claims. The increase in the number of labels, certifications, and claims has some consumers confused. Fortunately, some consumers are diligent enough to understand the difference between the labels and favor those with more stringent standards or verified claims. However, some consumers do not understand the difference or relevance between the labels, leading to the assumption that any label generally indicates that the product "does no harm".

What can the LCA community do to improve this situation? According to BCG, we can "reduce consumer confusion by bringing more transparency to production standards and the impacts that products might have. [Our] efforts can focus consumers on product characteristics that have the greatest impact on society." Good LCA communications can help to build cooperation, engagement, and trust with external stakeholders – whether that be suppliers along the value chain or consumers in the store.

In response, LCA is increasingly being embedded into sustainability strategies and therefore metrics-based communications are on the rise. However, it remains a struggle for LCA practitioners to effectively communicate the opportunities and benefits of LCA beyond the spreadsheet (Halloran 2012). The effective communication of LCA results is so critical that the European Commission created a separate working group dedicated solely to this topic for the Single Market for Green Product initiative. The objective of this paper is to share a commonly used communications framework and provide examples of how to effectively and credibly communicate LCA results to different stakeholder profiles using this framework.

2. Framework

The method we employed to create these metrics-based communications is not new (Steinberg 1994). It is commonly used by communications professionals. The difference between typical communications and the examples we are providing here is that the following communications are based on scientifically robust metrics (LCA) rather than just perceptions of what is “green”. Here, the framework of “how to communicate” is being applied to LCA results.

The first step in developing an effective communication is to understand the underlying positioning that you need to take. Be able to answer the following questions:

- 1) Why is it important for me / my company to talk about sustainability?
- 2) What is material / important to my audience?

Once you know where to position your messaging, then you can start to use the Communications Canvas (a basic framework), answering the following questions:

- 1) Compile your evidence: What are the data, facts, or metrics that I can use to provide scientific credibility for my communication?
- 2) Define the purpose: Why are you communicating? What are your goals with this communication?
- 3) Define the audience: To whom are you communicating?
- 4) Define your content: What message are you trying to communicate? What should the audience take away or understand after you have presented your communication?
- 5) Define your channel: Where and when will this communication take place?
- 6) Define your success: What would be the key performance indicators (KPIs) of my communication?

3. Examples

Provided below are six examples where the Communications Canvas has been applied to LCA results or similar metrics-based assessments.

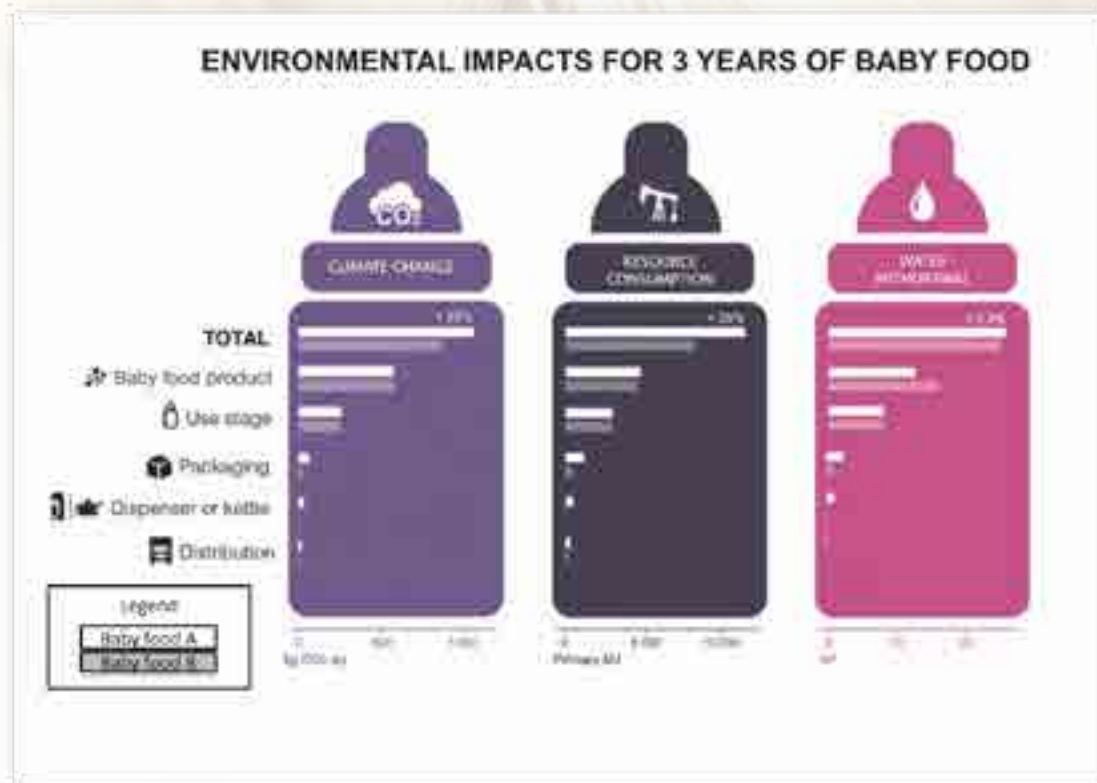


Figure 1: Example of communicating results from a comparative LCA

This Figure 1 shows an example of a simple figure presenting LCA results of two baby food products. Figure 1 was the result of answering the Communication Canvas as follows:

- 1) Compile your evidence: Comparative LCA of two baby food products.
- 2) Define the purpose: Inform and raise awareness internally, provide a proof-point, provide reassurance about the environmental impacts along the value chain of a baby food product
- 3) Define the audience: Internal company management
- 4) Define your content: Most impacts occur during the production of the food product and the use stage; product A (in white) is more impacting than product B (in grey)
- 5) Define your channel: Printed and presentation material (slides)
- 6) Define your success: Use of this information by managers and product designers

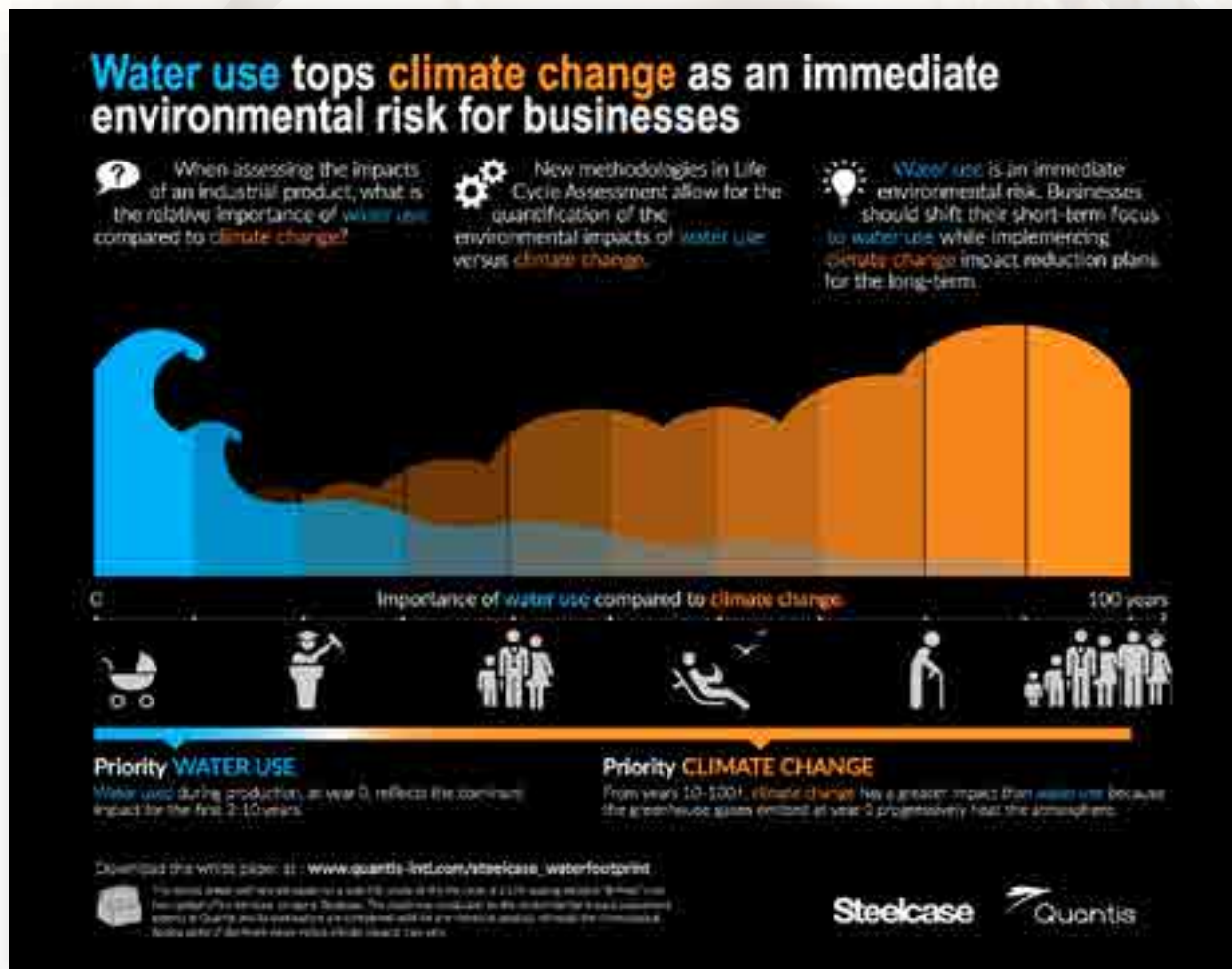


Figure 2: Example of communicating the prioritization of different impact indicators

Figure 2 shows an infographic used to explain the results of a complex comparison of impact indicators. This infographic was the result of answering the Communication Canvas as follows:

- 1) Compile your evidence: LCA of a lounge-seating product, with special focus on chronology of impacts
- 2) Define the purpose: Position Steelcase as a thought leader on the debate around whether climate change or water use is priority
- 3) Define the audience: Key Opinion Leaders, Scientific community
- 4) Define your content: Water is the more urgent priority while climate change is a long term concern.
- 5) Define your channel: web downloading
- 6) Define your success: number of downloads



Figure 3: Example of communicating the effect of re-designing a package

Figure 3 shows a video used to explain the effect of a packaging re-design. This video was the result of answering the Communication Canvas as follows:

- 1) Compile your evidence: Comparative LCA of several generations of packaging
- 2) Define the purpose: Build trust, reputation, and brand loyalty with consumers
- 3) Define the audience: Consumers who care about the environment, but who are not experts
- 4) Define your content: Sprint cares about the environment and has made measurable improvements to reduce its impacts from packaging
- 5) Define your channel: Online video
- 6) Define your success: Number of views



Figure 4: Example of comparing the life cycle impacts of different scenarios for making coffee

Figure 4 shows a web-based tool used to compare the life cycle impacts of different scenarios for making coffee. This tool was the result of answering the Communication Canvas as follows:

- 1) Compile your evidence: Comparative LCA of several coffee systems
- 2) Define the purpose: Win sales with potential customers
- 3) Define the audience: Customers / procurement contacts
- 4) Define your content: Nestlé Professional can provide the coffee system that performs best (environmentally) for your situation
- 5) Define your channel: Web-tool to be used on a tablet during a sales meeting
- 6) Define your success: \$ won in sales, supported by meetings with this tool



Figure 5: Example of teaching life cycle thinking to a non-expert audience

Figure 5 shows a board game used to teach life cycle thinking and build understanding of LCA results to a non-expert audience. This game was the result of answering the Communication Canvas as follows:

- 1) Compile your evidence: LCA of a single product relevant to the audience
- 2) Define the purpose: Challenge players' perceptions of what drives a product's environmental impacts/performance. Teach players "life-cycle thinking".
- 3) Define the audience: Any layperson / non-expert – from marketing, C-suite, logistics, consumers, etc.
- 4) Define your content: Products have a "life cycle", with impacts occurring along the way. Certain life cycle stages may be more/less impactful than what you expect.
- 5) Define your channel: Interactive board game that promotes learning
- 6) Define your success: Number of players with perceptions challenged, new vocabulary, understanding of "life cycle"

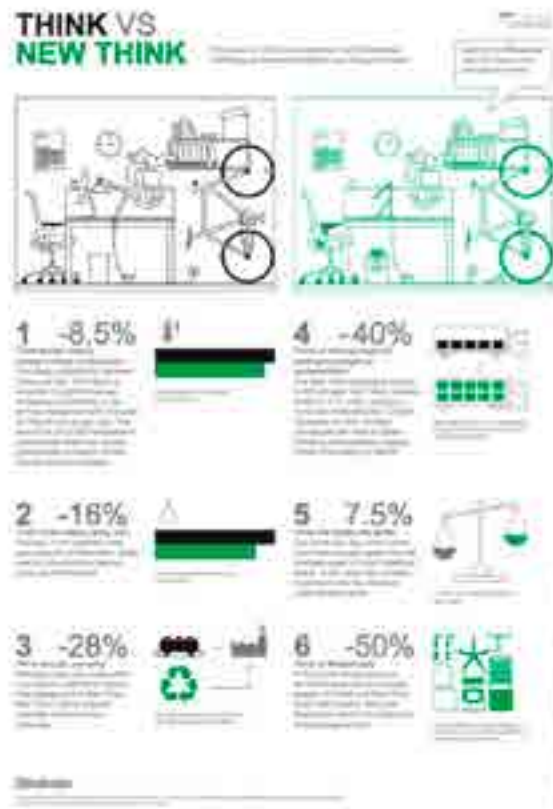


Figure 6: Example of gamification communication based on LCA outcomes

Figure 6 shows a "6 differences game" made out of LCA results aiming at highlighting key focus areas of a sustainability strategy to a relatively expert audience. This infographic was the result of answering the Communication Canvas as follows:

- 1) Compile your evidence: LCA of New Think Chair
- 2) Define the purpose: Position Steelcase as a thought leader in the eco-design field, get visibility on key topics in a fun way
- 3) Define the audience: Key Opinion Leaders, B2B clients, Scientific community
- 4) Define your content: New rules of thumb for eco-designers

- 5) Define your channel: Online report
- 6) Define your success: Number of downloads

4. Discussion

This short selection of examples shows that possibilities are endless when it comes to communicating LCA results. In order for your communication to be effective, it needs to be tailored appropriately based on your questions to the Communications Canvas framework. Employing this framework can help bridge the gap in communication between those who assess/measure sustainability metrics and those who want to understand environmental performance. The implications for LCA Practitioners are vast: present credible results beyond the spreadsheet, create value based on your LCA work, get bigger budgets to embed LCA into your organization's strategies, and increase supplier and other stakeholder engagement.

5. Conclusion

In conclusion, this Communications Canvas is a simple but effective tool that can help those of us without expertise in communications. It can help LCA practitioners who struggle with turning their LCA results exported from a software or spreadsheet into something understood by non-practitioners. This will allow LCA results to be understood, highlight the value of LCA, and allow the insights from LCA work to be integrated into decision-making. LCA practitioners should employ this framework that is used by communications professionals in order to effectively communicate on environmental performance and the valuable insights that can be gained from using LCA.

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158. Web based communication of farm LCA results to farm managers

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ABSTRACT

Farm level life cycle assessment (LCA) has been recognized as an appropriate farm management tool in order to identify the environmental burden of farming. So far, little attention was directed to the targeted communication of the results to farm managers and further users, such as farm consultants or scientists. However, communication is the key to the understanding of the results by the decision makers and hence to taking actions for improving the environmental impact of farming. This triggered the aim of developing a comprehensible, practical, and user-friendly web based communication tool for environmental management at farm level.

In the design process, experts from linguistics, software engineering, and agricultural research worked closely together. Decision-making and communication patterns established in psychological, linguistic and pedagogic theories were taken into account as well as long-lasting experience in farmer consultancy.

The tool *FarmLife-Report* is structured into three steps of result data communication: i) information about means of production, ii) agronomic key figures, and iii) environmental impacts. For the steps 2 and 3 the result information is structured into three levels of detail: First, an overview level for farm managers. Second, a consultant level addressed at farm advisors providing more details, and third, the complete set of figures and result, i.e. the expert level, directed at scientists. Different types of tables and charts (bar and bubble charts) are used to give a targeted information on the farm results.

FarmLife-Report, the farm LCA communication tool, was applied on a network of 51 farms in Austria. The tool proved to fulfil the expected tasks. The feedback from the applicants was very encouraging. The key to success was to apply a balanced combination of the tool's user-friendliness, its extensive possibilities of analyses, and its targeted communication on different levels of detail. In a further step, the identified weaknesses will be removed and a new version targeting at educational purposes will be developed. At this time, a dairy commissioned the application of *FarmLife-Report* for its milk supplier to offer them environmental consultancy.

Keywords: Knowledge transfer, Decision support, Communication tool, farm consultancy, FarmLife-Report

1. Introduction

Over the last decade, farm life cycle assessment (farm LCA) has been recognized as an appropriate farm management tool to identify the environmental burden of farming. Several efforts were made in the development of tools, methods, emission models, and databases. However, the ultimate challenge is to create a benefit for practitioners by transferring this knowledge to a wide range of farm managers and further users. Communicating environmental management results, e.g. from LCA, and practical advice is complex. Still, communication is the key to the understanding of the results by the decision makers and hence to taking actions for improving the environmental impact of farming. A further complexity arises from the requirement to provide a communication tool empowering the addressee to autonomous use and allowing him to obtain explanations regarding the respective LCA results. This calls for a comprehensible, practical, and user-friendly web based tool.

In the project FarmLife, financed by the Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management (BMLFUW), we aimed at elaborating such a web based communication tool for environmental management at farm level, which is presented in this paper.

2. Designing a knowledge transfer Farm LCA Tool for decision making

The tool *FarmLife-Report* is part of a whole set of farm LCA tools developed within the project FarmLife (Herndl et al., 2016). Its purpose is the communication of farm LCA results to different stakeholders, i.e. farm managers, farm advisors, scientists and further interested parties. When designing the tool, two aspects were at core: integrating knowledge from the fields of communication science and psychology, and using the experience from previous consultancy activities for farm managers. Hence, experts from linguistics, software engineering, and agricultural research closely collaborated for the design. We identified three conditions, which have to be fulfilled in order to facilitate a change of attitude by the farm manager: i) Assurance that all data are correct and the calculation process is performed correctly; ii) a quantitative appraisal of agronomic key figures of the

farm; and iii) providing expertise with appropriate information for consultancy. The first is indispensable to build trust in the results. The second allows classifying whether a farm is comparatively performing on average, well, or not. The third condition displays competency. All of them have to be met to enable emotional amplification, i.e. an inner emotional conviction as a result of further information search and appraisal (Finotti, 2015), by the farm manager of the assessed farm.

As a part of the project FarmLife (Herndl et al., 2016) the study “Life cycle assessment: decision under ambivalence” (Finotti, 2015) explicitly refers to the problems of communication of the results as well as to possibilities of decision support for the farm managers. The study is based on psychological (emotion- and attitude-psychological) (e. g. Hänze, 2002) as well as linguistic (e. g. Grice, 1975) and pedagogic theories, and on findings from the field of knowledge transfer.

Induced by individuality in terms of sociocultural and emotional aspects, the LCA results are likely to be experienced ambivalently by the involved farm managers – especially, if a decision on further farm management is to be made. As far as it is no routine decision, each decision does not only have cognitive (rational) but also emotionally action-guiding background. In many cases, decisions are supported by inner values and attitudes. The latter enable humans to have a stable and structured view of the world. If the option of an action (operation) includes positive as well as negative aspects, this will be experienced as being rather aversive, i.e. according to Hänze (2002) emotionally threatening, and emotionally incriminatory. Conditioned by character, humans deal differently with this ambivalence. Mostly, we try to build up a structure of dominance in order to develop polarised emotions, based on which we are able to make a decision and become capable of acting, i.e. decision under ambivalence. Hänze (2002) argues for an automatic processing of emotional polarising and amplification, which is always repeated until somebody is capable to make a decision (Figure 1). Emotional amplification combines provision of information on a problem as well as estimation of utility, and probability of different consequences of actions, the search for social strengthening and emotional imagination or mental simulation of possible consequences of a decision.

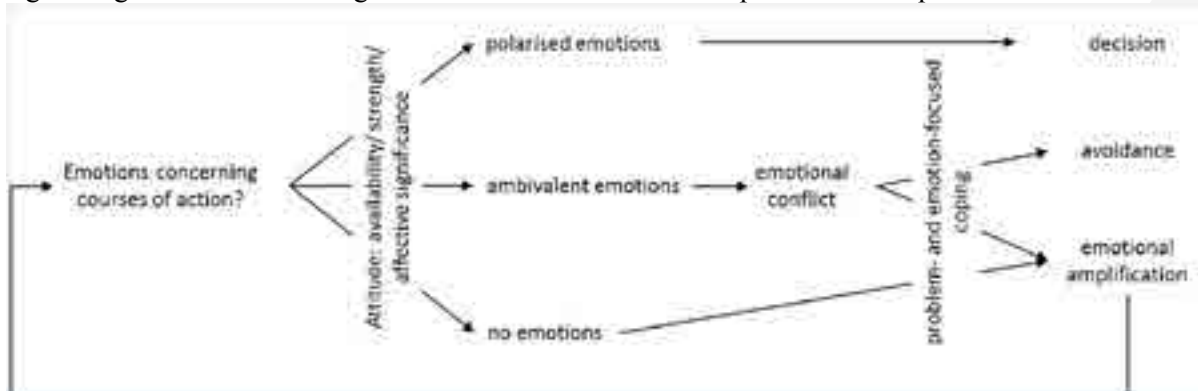


Figure 1: Simplified diagram of the integrative frame-model of emotional decision-making according to Hänze (2002).

Of course, the way of knowledge transfer plays an important role in this matter. Knowledge transfer has to be accomplished in a way technically and linguistically accommodated to the customers (e. g. Wichter und Antos, 2001; Busch und Stenschke, 2004). Strategies, which are able to produce action-guiding emotions, function in a supporting way. In order to implement a successful knowledge transfer to the farm managers with respect to their individual results, we developed a concept for farm consultancy both in terms of knowledge transfer and of a more conscious communication, based on the abovementioned facts and theories.

The concept enabled us to provide commonly used agronomic key figures and options of action, and to connect them to environmental impacts. This combination allows practical recommendations for farm-internal optimisation as well as outward-directed communication of the environmental performance. Previous experience showed that environmental impacts are still novel key figures, which are better understood by farm managers in a bottom-up strategy. For example, the amount of fertilizers and the nitrogen balance of a field have to be addressed before discussing eutrophication.

In addition to autonomous use of the web based tool *FarmLife-Report*, consulting can be performed in workshops with the help of the same tool. Still, a direct and personal consulting in combination with *FarmLife-Report* is recommended. Thus, it is easier to consider individual aspects, and communication can be better adapted to needs of the farm manager helping to support actions on the farm.

3. Structure of the communication tool *FarmLife-Report*

The tool *FarmLife-Report* has three steps of presenting data and results (Figure 2): The first step addresses the means of production and offers an overview of the farm characteristics.



Figure 2: Three steps of presenting data and results for farm LCA communication

The second step provides a comparison of agronomic key figures of a group of farms, allowing an appraisal of the assessed farm. The key figures are depicted with bar and bubble charts. The bar charts are divided into quartiles with the first quartile and the fourth quartile representing the favourable and unfavourable situation, respectively. The bubble charts are split empirically into four sections (Figure 3): These sections give a first classification of the assessed farm. This classification is performed using six key indicators, i.e. non-renewable cumulative energy demand (nrCED,ecoinvent, Hischer et al., 2010), global warming potential (GWP, IPCC 2007), aquatic eutrophication with nitrogen and phosphorus (EDIP03, Hauschild & Potting 2005), total nitrogen fertilization on farm, and direct-cost-free output. For each of the four farm classes, there is a basic message: For the extensive farms (i.e. moderate input use) it is: “Continue acting with moderate use of inputs on your entire farm, including labour time.” For efficient farms: “Pay attention that the efficiency is not depleting the natural resources of your farm. Nutrient balances in the soil and the organic matter content have to be considered in the long term.” Inefficient farm: “Try to corner the challenges and search for assistance”. Intensive farms: “Take management decisions on your farm with regard to its environmental impacts.” In step 2 hyperlinks are provided, leading to further consultancy documents.

The third step comprises environmental impacts and options for action suitable for the assessed farm. At this level, *FarmLife-Report* depicts the environmental impacts related to so-called “input groups”. Input groups are a scheme, where all resources, means of production, and direct emissions of the farm are grouped thematically. They are: ‘Buildings and equipment’, ‘Machinery’, ‘Energy carriers’, ‘Fertilisers and field emissions’, ‘Pesticides’, ‘Purchased seeds’, ‘Feedstuffs, concentrates (purchased)’, ‘Feedstuffs, roughage (purchased)’, ‘Purchased animals’, ‘Animal husbandry’, and ‘Other inputs’. This allows identifying the main contributors to each environmental impact.



Figure 3: Classification of farms into four farm classes.

Within step 2, i.e. comparison of agronomic key figures, and step 3, i.e. environmental impacts, *FarmLife-Report* provides three levels of expertise: i) An overview level, addressed at inexperienced applicers of environmental management, comprising the most important figures and results (Table 1). The indicators are reduced to the core information, necessary to build up knowledge and understanding. These indicators are interlinked and should be understood without assistance from a farm consultant. ii) A level for farm advisory services offering a larger number of figures and results as well as information for argumentation, which shall support the farm consultant in his work. iii) An expert level providing the most detailed level of figures and results, i.e. 139 agronomic key figures and 37 impact categories. This level allows in-depth analyses of environmental impacts and addresses environmental scientists.

Table 1: The three levels of expertise and the amount of available data in each of them, referred to different areas of interest.

	Step 2: Agronomic key figures			Step 3: Environmental impacts		
	Overview	Consultant	Expert	Overview	Consultant	Expert
Resources	13	23	54	4	10	14
Nutrients	20	46	70	2	3	8
Pollutant				2	6	15
Economy	6	11	15			

The environmental impacts are given for two functional units, i.e. the livelihood preservation function, which is expressed per hectare utilized agricultural area (UAA), and the productive function, which is expressed per nourished person, i.e. according to FAO (2001) 3879 megajoule digestible energy. The LCA is performed according to the SALCA methodology (Nemecek et al., 2010) using SimaPro 7.3 for computing life cycle impact assessment results (PRé Consultants, Amersfoort, The Netherlands). The life cycle inventories employed in this study originate from the ecoinvent data base v2.2 (ecoinvent Centre, 2010) and from the SALCA database (Nemecek & Kägi, 2007).

4. Application on a farm network and acceptance of users

In the frame of the project FarmLife the set of farm LCA tools “FarmLife” including the analysis and communication tool *FarmLife-Report* were applied on a network of 51 farms in Austria (Bystricky et al., 2015). Four farm types were assessed with the tool: arable farms, dairy farms, fattening farms (pigs and cattle), and wine-growing farms.

An example of figures of results of all three steps is given below for a dairy farm. Figure 4 depicts figures on means of production and agronomic key figures within the farm network for fertilization. The comparison shows that the assessed farm has a comparatively high fertilization rate per ha compared to the other dairy farms of the network.

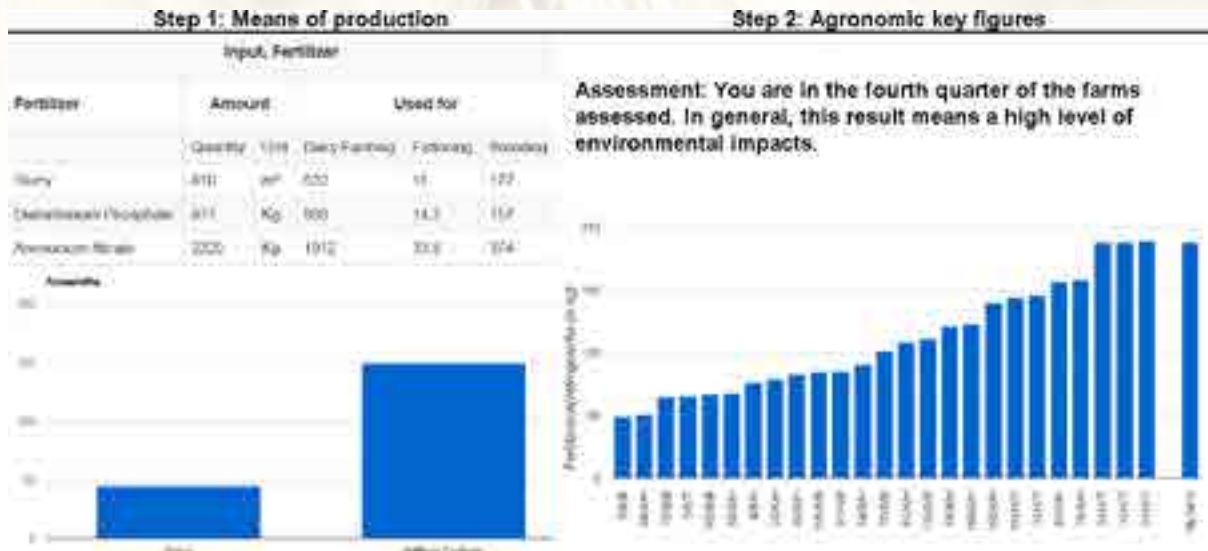


Figure 4: Key figures on fertilization of the assessed farm (Step 1: means of production) and comparison within the network of farms (Step 2: Agronomic key figures). The assessed farm is at the right end of the graph.

The bubble chart (Figure 5) indicates the result of the assessed dairy farm (red dot) compared to the other dairy farms in the Austrian network for GWP and both functional units, i.e. per ha utilized agricultural area as well as per nourished person. According to farm classification, the assessed farm is considered an intensive farm.

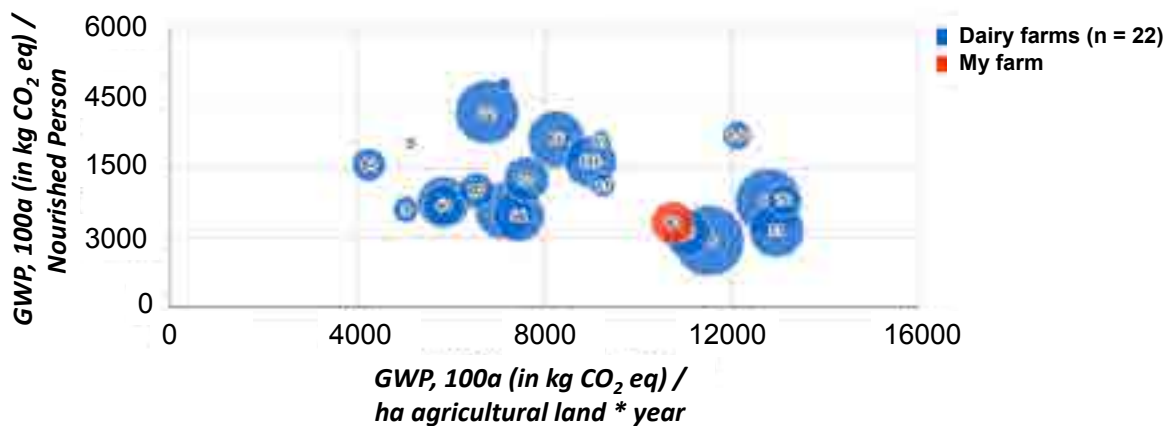


Figure 5: Bubble chart illustrating the global warming potential, 100 years (GWP) of an analysed dairy farm (red dot) in comparison with 21 other dairy farms. The size of the bubbles indicates the total GWP of the farm (without reference to the functional units). The numbers in the bubbles are the farm identification number

In order to be able to derive mitigation options for environmental impacts, a contribution analysis is performed with the help of results of each input group. In Figure 6, the contribution analysis is given for the GWP of the exemplary dairy farm. The areas of possible actions are 'Animal husbandry', 'Concentrates, purchased', and 'Fertilizers and field emissions'.

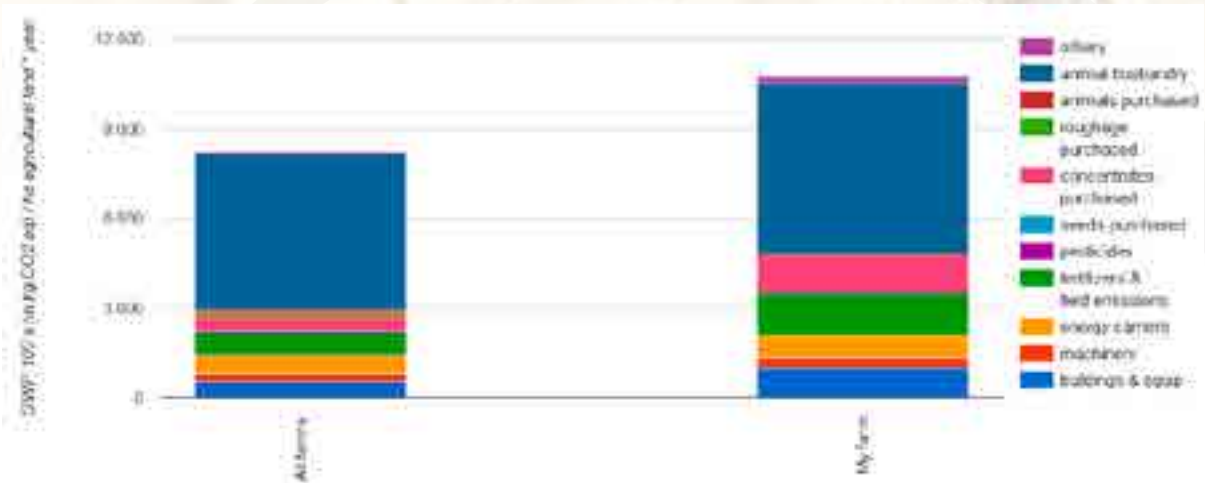


Figure 6: Contribution analysis: Global Warming Potential, 100 years (GWP in kg CO₂ / ha UAA) of an analysed dairy farm. The mean for all farms is shown in the left bar, the results for the analysed farm is shown in the right bar. Depicted is the contribution of the different input groups.

An important lesson we learned in the application of the tool and in exchange with the farm managers was not to confront them with too many details at the beginning. What proved to be more constructive was showing them straight away the strengths and challenges of their farm. Later on, insights that are more detailed are desired and must therefore be ready-to-use to underline the findings in a counselling interview.

Overall, the communication tool was very well received by farm managers as well as farm consultants. It was highlighted that *FarmLife-Report* had a clear structure, had informative text elements, allowed flexibility when choosing the reference farms, and excelled at user-friendliness. However, some weaknesses were detected: For example it became apparent that new users of *FarmLife-Report* need assistance to utilise the whole potential of the tool. Despite the clear structure, the amount of data and information was overstraining for some users. Furthermore, the empirical approach for classifying the assessed farms worked well, but needs reconsideration on a scientific basis. We plan to eliminate those weaknesses in the coming months.

This first series of application triggered interest from further stakeholders. A private holding of the dairy sector is currently involved in applying the farm LCA tool, including the analyses and report functions, on its supplying dairy farms, offering them analyses and extension services on a voluntary basis. Furthermore, educational institutes, such as agricultural high schools or the Austrian University College for Agrarian and Environmental Pedagogy, showed their interest. Hence, plans exist to develop a version of *FarmLife-Report* for the use in agricultural education.

5. Conclusions and outlook

We conclude that with the development of *FarmLife-Report* alongside with the entire set of the FarmLife-Tools, we were able to reach the aim of offering the farm manager and his advisor a strongly integrated tool that covers data collection, calculations, assessment, and communication. The key to success was to apply a balanced combination of the tool's user-friendliness, its extensive possibilities of analyses, and its communication on three different levels of detail addressing different target groups (e.g. farmers, advisors, and scientists). This was achieved with the support of communication sciences, which were integrated in the design process, combined with experience in farm consultancy and expertise in tool programming. We identified three conditions, which have to be fulfilled in order to facilitate a change of attitude by the farm manager: i) Ensuring the correctness and plausibility of input data and LCA results; ii) a quantitative appraisal of agronomic key figures; and iii) providing expertise with appropriate information for consultancy.

However, an important lesson we learned was not to confront the farm manager with too many details at the beginning, but straight away show the strengths and challenges of his farm. Hence, the

structure of the communication tool, including three steps of presenting data and results for farm LCA communication as well as the three levels of expertise, proved to be expedient.

The test on 51 Austrian farms proved to be very successful. It revealed not only the abovementioned strengths of the tool, but also weaknesses. In a next step, those weaknesses shall be eliminated. As new groups of applicants have shown their interest in such a tool, the set of tools including *FarmLife-Report* shall be developed further.

Finally, further collaborations with the public and private sector for applying the farm LCA tool are intended.

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ORAL PAPERS BIODIVERSITY

9. Biodiversity

85. Assessing the impact of animal production on biodiversity: looking beyond land use

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ABSTRACT

On the one hand, life cycle assessments (LCA) of the livestock sector have largely focused on greenhouse gas (GHG) emissions despite evidence of impacts on biodiversity. On the other hand, recent developments of LCA characterization models to estimate impacts on species have focused on a single driver of biodiversity loss: land use. The objective of this paper is to quantify the impact of the global cattle production on biodiversity through land use and GHG emission, and the relative importance of these two drivers. At inventory level, we used a reference model for global livestock production and its GHG emissions. For impact assessment, we used land use characterization factors recommended by the UNEP-SETAC as well as the available methods to link GHG emissions and impacts on biodiversity. We show that global cattle production has an impact on biodiversity both through land use (2.25% loss of species-eq.) and climate change (0.96% loss of species-eq.). The relative importance of these two drivers varies spatially and for certain regions such as Europe and North America, the impact of cattle production on biodiversity is higher through GHG emissions and climate change than through land use. We identify the need for more integrated LCA approaches addressing several drivers of biodiversity loss and their potential interactions.

Keywords: life-cycle assessment (LCA), sustainable agriculture, environmental impact, food products, biodiversity conservation

1. Introduction

The livestock sector is crucial for the global food security and rural economy, it is also facing a growing demand driven by increase in population and per-capita income (Alexandratos & Bruinsma 2012). At the same time, the sector is a major user of natural resources (land, water) and substantially contributes to environmental pressures as greenhouse gas (GHG) emissions and nutrient pollution (Steinfeld et al. 2006). These pressure turn into important impacts on biodiversity which is at the endpoint of environmental cause-effect relationships (Gerber et al. 2015). Emphasis has been put on the impact of livestock on biodiversity through land use. Livestock production directly or indirectly modifies biodiversity habitats in about 30% of the global terrestrial area dedicated to pastures or feed crops (Ramankutty et al. 2008).

During the past decade, significant research efforts have been undertaken on the quantification of land use impacts on biodiversity in life cycle assessment (LCA) (Koellner & Scholz 2008; Michelsen 2008; Schmidt 2008; Goedkoop et al. 2012; DeBaan et al. 2013a; b). The UNEP-SETAC Life Cycle Initiative has achieved important progress in building scientific consensus on the integration of land use related impacts on biodiversity in LCA (Frischknecht et al. 2016). A theoretical framework was developed (Koellner et al. 2013) and very recently an impact model and characterization factors (Chaudhary et al. 2015) are being recommended. This model builds on previous work from de Baan et al. 2013b and combines several advantages – a global cover and a high spatial differentiation between 804 ecoregions – and innovations as the consideration of several levels of conservation value for biodiversity and the use of countryside species area relationships which improves the relevance of the model in agricultural landscapes.

A limitation for the application of Chaudhary et al. (2015) model in the context of agriculture is the inclusion of broad “grassland” and “cropland” land use classes, without distinction between levels of management intensity. Characterization models linking land use to biodiversity often face a trade-off between the level of spatial differentiation and the level of definition of land use classes (Teillard et al. 2016a). On this trade-off also stands the characterization model of Alkemade et al. (2009; 2012) which is global without any spatial differentiation but distinguishes between four levels of land use

intensity for grassland, and two for cropland. The biodiversity indicators also differ between these two models – it is based on global species loss in Chaudhary et al. (2015) and on species abundance in (Alkemade et al. 2009; 2012).

Besides their intrinsic limitations, characterization models focusing on land use do not capture the range of impacts that livestock production have on biodiversity (Teillard et al. 2016b). In particular, livestock has a significant contribution to human-related GHG emissions (14.5% according to Gerber et al. 2013) and climate change is the second but increasingly important driver of biodiversity loss, after habitat change (MEA 2005). The model of Alkemade et al. (2009) includes characterization factor linking species abundance to climate change in addition to land use. Schryver et al. (2009) specifically developed a characterization model addressing the effect of GHG emissions and climate change on species loss. However, models and methods on this topic are very few and do not have a level of scientific development and consensus comparable to those on land use.

The objective of this paper is to quantify the impact of the global cattle production on biodiversity through both land use and GHG emission, and the relative importance of these two drivers. We used the most recent LCA characterization models to quantify these impacts, to test their relevance and applicability to livestock production, and to reveal possible limitations and future research needs.

2. Methods

2.1. Inventory analyses of land use and GHG emissions

Inventory flows were derived from the FAO Global Livestock Environmental Assessment Model (GLEAM¹). GLEAM models livestock production on the global scale and enables a comprehensive analysis of its GHG emissions. All computations are GIS based, i.e. all inputs and outputs are global raster layers (maps) with a resolution of 10*10km grid cells at the equator. We focused on cattle and its two main commodities – milk and meat. The functional unit was 1 kg of animal proteins, summing proteins from milk and from meat.

GHG emissions were directly taken from the study of Gerber et al. (2013) that relied on GLEAM to estimate the global contribution of livestock supply chains to anthropogenic emissions. The emissions (in kton/year) of the main GHGs in the context of livestock production – CH₄, N₂O and CO₂ – were accounted for in the inventory.

Computing land use for feed is an intermediary output of GLEAM. The variety of feed components described by GLEAM (several types of roughages, crop residues and by-products) were grouped into three categories: grassland, on-farm cropland and off-farm cropland. Land use for grassland and on-farm cropland is localized in the same place (i.e. in the same grid cell) than livestock production. Land use for off-farm cropland is related to imported feed and its localization is unknown. We focused on land use occupation and the land use inventory flows were the area (km²·year) occupied by the three categories (grassland, on-farm cropland and off-farm cropland).

2.2. Impact assessment: from land use to impact on biodiversity

Two different characterization models were used to translate land use into impact on biodiversity. The first characterization model was the one from Chaudhary et al. (2015) which is recommended by the UNEP-SETAC life cycle initiative (Mila i Canals et al. 2016). We used the characterization factors quantifying the average effect of occupation on global species richness, aggregated across taxa (mammals, birds, amphibians, reptiles). These characterization factors provide a value of the global species equivalent lost per km² of occupation by grassland or cropland. For grassland and on-farm cropland, we used the country specific value of the characterization factors. For off-farm cropland we used the global average value as the land use localisation was unknown. This global average value for grassland and cropland is presented in Table 1.

The second characterization model was the one from Alkemade et al. (2009; 2012). In this model, the biodiversity indicator is expressed as Mean Species Abundance (MSA) that reflects the total number of individuals across species and taxa. Characterization factors provide an MSA value for

¹ <http://www.fao.org/gleam/en/>

different land use and intensity classes, which corresponds to the relative MSA of these land use classes compared to an undisturbed reference. 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.

In order to convert relative MSA values into a global MSA loss associated with the occupation of 1km² of the different land use classes, we applied the following formula:

$$GlobalMSALoss/km^2 = \frac{(1 - relativeMSAvalue)}{A}$$

where A is the total land surface on earth in km². We used the same total land surface area on earth than Alkemade et al. (2009; 2012), i.e. 147650500 km² (Bartholomé 2005). Because the MSA reflects species abundance and not species richness, there is a direct relationship between local and global loss of MSA. For instance, the local occupation of 1% of the global terrestrial area with a land use class having a relative MSA value of 0.5 corresponds to a 0.5% loss of the global MSA.

Characterization factors from Alkemade et al. (2009; 2012) provide global values without spatial differentiation; however, they differentiate between two intensity classes both for grassland and cropland (Table 1). The production system of the grid-cell was used to determine the grassland intensity level: extensive grassland in grassland-based production systems and intensive grassland in mixed production systems. For on-farm cropland, we weighted the relative MSA value by the relative proportion of intensive and extensive cropland in the region of the grid-cell (Dixon, Gibbon & Gulliver 2001), as described in Alkemade et al. (2009). Off-farm cropland was assumed to always come from intensive cropland.

In a given grid cell, the impact of livestock production on biodiversity through land use was computed as the sum over land use categories of their area (as calculated by the inventory analysis) multiplied by their global MSA loss or species eq. loss by km². This impact was summed over all grid cells to obtain the global impact of livestock production on biodiversity through land use.

Table 1. Source and value of the characterization factors used to quantify the impact of land use on biodiversity. eq. = equivalent, MSA = Mean Species Abundance.

Land use	Global species eq. lost / km ² (Chaudhary et al. 2015)	Relative MSA value (Alkemade et al. 2009; 2012)	Global MSA loss / km ² (calculated from Alkemade et al. 2009; 2012)
Extensive grassland	1.15*10 ⁻⁹	0.6	2.69*10 ⁻⁹
Intensive grassland	1.15*10 ⁻⁹	0.5	3.36*10 ⁻⁹
Extensive cropland	2.04*10 ⁻⁹	0.3	4.70*10 ⁻⁹
Intensive cropland	2.04*10 ⁻⁹	0.1	6.04*10 ⁻⁹

2.3. Impact assessment: from GHG emissions to impact on biodiversity

As for land use, two different characterization models were used to translate GHG emissions into global loss of species and MSA. The first step was to translate the inventory emissions of CH₄, N₂O and CO₂ into global mean temperature increase. To do this, we used the factors provided by Schryver et al. (2009) under the hierarchist cultural perspective (Hofstetter 2000). These factors model the links between GHG emission, GHG air concentration, radiative forcing and global mean temperature increase. Their value is shown in Table 2.

The study of Schryver et al. (2009) also provides characterization factors linking global mean temperature increase to global Potentially Disappeared Fraction (PDF) of species. Authors multiplied characterization factors by the total surface of (semi)natural terrestrial area of the world (10.8*10⁷ km²) to express them in PDF.km².°C⁻¹. We divided the characterization factors by this area in order to obtain the global PDF per °C of global mean temperature increase (Table 2). We assumed that this global PDF was equivalent to the global species eq. loss of Chaudhary et al. (2015).

The study of Alkemade et al. (2009) was used to link global mean temperature increase to global MSA loss. Authors used the IMAGE model (Bouwman, Beusen & Billen 2009) to calculate characterization factors quantifying the loss of MSA per °C of global mean temperature increase in

different biomes. We computed the weighted average of these characterization factors, by the area of each biome in the IMAGE model, to obtain the global MSA loss / °C of global mean temperature increase (Table 2).

Table 2. Source and values of the characterization factors used to quantify the impact of greenhouse gas (GHG) emissions on global mean temperature increase and biodiversity. PDF = Potentially Disappeared Fraction of species, MSA = Mean Species Abundance. GMTI = Global Mean Temperature Increase.

	Value	Unit	Reference
GHG emissions to global mean temperature increase			Schryver et al. (2009)
CO2	$4.17 \cdot 10^{-8}$	°C / kton	
CH4	$8.24 \cdot 10^{-7}$	°C / kton	
N2O	$1.36 \cdot 10^{-5}$	°C / kton	
Global mean temperature increase to impact on biodiversity			
Global PDF	$5.91 \cdot 10^{-2}$	Global MSA loss / °C GMTI	Schryver et al. (2009)
Global MSA loss	$7.44 \cdot 10^{-2}$	Global PDF / °C GMTI	Alkemade et al. (2009)

2.4. Analyses

Impact assessment resulted in a global quantification of the impact of cattle livestock production on species richness and abundance (MSA) through land use and climate change. To investigate the relative importance of these two drivers, we computed the ratio of climate change impacts vs. land use impacts. Furthermore, we explored whether the spatial variation of this ratio across the globe was rather influenced by differences at inventory level or by the country specific values of land use characterization factors (Chaudhary et al. 2015). To do this we computed the country averages of (i) the ratio between total GHG emissions (in CO₂-eq) and total land occupation (in km²) and (ii) the ratio of climate change vs land use impacts on PDF. We compared these values with the country specific values of land use characterization factors (average value of the grassland and cropland characterization factors, Chaudhary et al. 2015).

3. Results

Fig. 1 shows the contribution of cattle livestock production to biodiversity loss on a global scale. Land use by the global cattle livestock production was estimated to cause the a 2.25% loss of terrestrial species richness (PDF, Fig. 1a) and a 4.29% reduction of terrestrial species abundance (MSA, Fig. 1b). Compared to land use, the global impact of cattle livestock production through climate change was lower for both biodiversity indicators. It was estimated to a 0.96% and 1.14% loss of PDF and MSA, respectively.

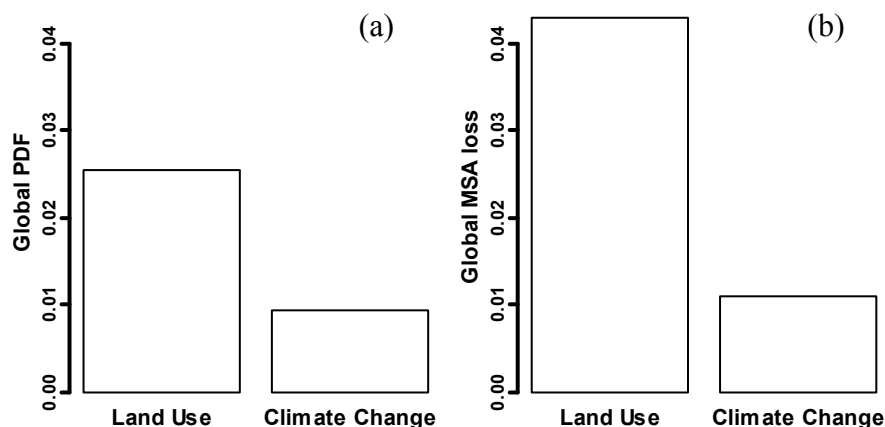


Figure 1. Total impact of cattle livestock production on biodiversity through land use and climate change. (a) Total impact in global Potentially Disappeared Fraction (PDF) of species, (b) Total impact in global loss of Mean Species Abundance (MSA).

The relative importance of impacts on biodiversity from climate change and land use (% CC/LU impact) varied among global regions, with contrasting patterns for PDF and MSA indicators (Fig. 2). For the MSA indicator, the impact of cattle livestock production through climate change was lower than the impact from land use in most areas (i.e. % CC/LU impact < 100, Fig. 2b). For the PDF indicator however, there were large areas in North America, Europe and Asia where the impact of cattle production was higher through climate change than through land use (Fig. 2a).

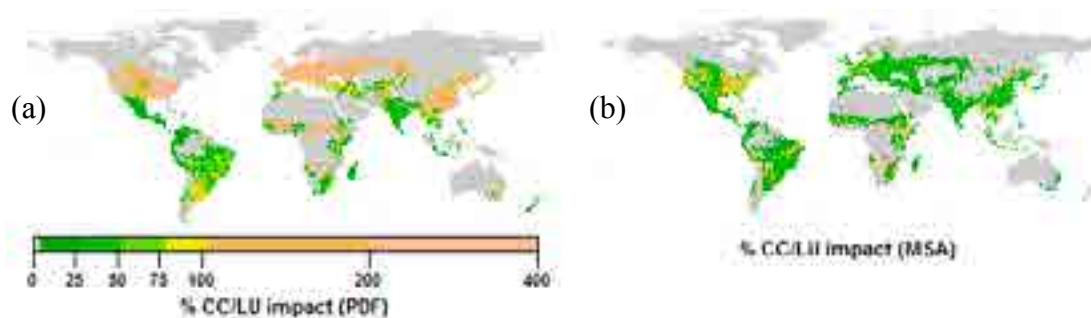


Figure 2. Relative impact of cattle livestock production on biodiversity through climate change compared to land use (% CC/LU impact). (a) Potentially Disappeared Fraction (PDF) biodiversity indicator, (b) Mean Species Abundance (MSA) biodiversity indicator. The two subfigures have the same color scale.

4. Discussion

In this study, we combined a reference model for global livestock production and its GHG emissions with biodiversity characterization factors recommended by the UNEP-SETAC. We show that land use related to cattle beef and dairy production has a significant impact on biodiversity on a global scale (2.25% loss of species-eq.). We also underline that land use is not the only driver of impact, cattle production has a strong impact on biodiversity through climate change. Although impacts through climate change on a global scale (0.96% loss of species-eq.) are lower than those from land use, they are higher in several regions (Europe and North America in particular). By quantifying the relative importance of these two drivers and by using two different characterization models, we identify research needs for the development of biodiversity characterization models and their applicability the agriculture and livestock sectors.

The two characterization models that we used for land use have opposite limitations. While the model of Alkemade (2009; 2012) distinguishes several classes of cropland and grassland intensity, Chaudhary et al. (2015) considers only on broad grassland and cropland land use classes. Failing to consider several levels of intensity is an obvious limitation for applicability to the agriculture and livestock sectors. A consequence will be that intensive systems will always appear to have a less detrimental impact on biodiversity than extensive systems. Intensive production systems use less land per unit of product but this land has a higher impact on biodiversity (Green et al. 2005), which cannot be addressed with characterization models including only one land use class for grassland or cropland. Conversely, certain management practices and low level of intensity in livestock production can have a positive impact on biodiversity (Watkinson & Ormerod 2001). To our knowledge, no land use characterization model is currently able to reflect such positive effects. Despite differentiating between several intensity classes, the model of Alkemade (2009; 2012) considers nil impacts but no positive impacts because of the type of land use reference selected. The development of more precise land characterization factors that can incorporate the effect of intensity levels and key farming practices (including those benefiting biodiversity) is a priority for increasing the capability of LCA as a decision support and analytical tool for agricultural systems (Teillard et al., 2016b).

Beside this limitation on intensity levels, the model of Chaudhary et al. (2015) provides a high level of spatial differentiation – at country or ecoregion scale – while Alkemade et al. (2009; 2012) only provides global characterization factors values. The variation of species richness levels on a

global scale shows strong spatial patterns (Jenkins, Pimm & Joppa 2013) and a better spatial differentiation of characterization factors is clearly an important improvement for their ecological relevance. The country specific value of land use characterization factors had a strong influence on the relative importance of the land use vs. climate change drivers of impacts on biodiversity (result not shown). The higher biodiversity impact of climate change compared to land use in Europe and North America is likely to be explained by lower species richness and endemism levels in these regions, reflected by the characterization factor values, and to a lower extent by differences at inventory level. Indeed, more intensive systems in these regions tend to be more efficient, with lower GHG emissions and lower land use by unit of product at the same time (Teillard et al., 2014). These results highlight the importance of spatial differentiation and getting the numbers right when it comes to country or region specific characterization factors. Moreover, the range of variation of the country specific values of Chaudhary et al. (2015) characterization factors (for cropland: $\min = 6.2 \cdot 10^{-11}$, $\max = 2.36 \cdot 10^{-7}$) is much greater than the range of variation of Alkemade et al. (2009) characterization factors between intensive and extensive land use class (for cropland: extensive = $4.70 \cdot 10^{-9}$, intensive = $4.70 \cdot 10^{-9}$). If spatial differentiation and differentiation between intensity levels were combined, the former would have a stronger influence on differences of biodiversity impacts between ecoregions or countries. However, it would still be crucial for decision makers within a country to be able to compare the impact of different production systems with various intensity levels.

Several elements limit the comparability of the results obtained with the different characterization models used in this study. We used characterization models with two different biodiversity indicators: species abundance and species richness. The impact of land use and climate change was higher (in terms of % loss at global scale) on abundance than on richness. This result is consistent with the ecological theory predicting that decreases in population size do not necessarily cause species extinction but increase its risk (Lande 1993). Compared to the characterization models of Alkemade et al. (2009; 2012) and Schryver et al. (2009), Chaudhary et al. (2015) introduce an important improvement for ecological relevance by considering differences in conservation values between species. This means that species-eq. lost in Chaudhary et al. (2015) are not exactly similar to PDF in Schryver et al. (2009). Moreover the taxa and number of species used for the development of the two characterization models differ. This results in a partial comparability of the impacts estimated from the two characterization models because the species richness of the main taxa, as well as the species richness of threatened and non-threatened species are only partially correlated on a global scale (Jenkins et al., 2013)

To our knowledge, we used the only two characterization models available to link GHG emissions to impact on biodiversity (Alkemade et al., 2009; Schryver et al., 2009). The low number of available models and the absence of more recent alternative strongly contrast with the recent and prolific scientific development of characterization models for land use (e.g. de Baan et al., 2013a; b; Souza et al., 2013; Chaudhary et al., 2015). Yet, we demonstrate that the impact of cattle production on biodiversity through GHG emissions and climate change can be higher than its impact through land use. In this case, focusing on land use would lead to severely under-estimate the impacts on biodiversity. Beyond the sole livestock sector, it has been suggested that climate change could surpass habitat change as the most important global driver of biodiversity loss in the next few decades (Leadley et al., 2010). Estimating the impact of climate change scenarios on biodiversity is an active field of research in ecology (Bellard et al. 2012), which could benefit LCA researchers to develop new characterization models.

In our study it is not clear how the impacts of cattle production from land use and climate change could be combined to compute a total loss of species richness or abundance. The main hypotheses for combining drivers of biodiversity loss are partial additivity (total impact lower than the sum of the individual impacts), full additivity (total impact equals the sum of the individual impacts) or synergy (total impact higher than the sum of the individual impacts). In the case of species abundance, Alkemade et al. (2009) stress that little information exist on the interactions between drivers of biodiversity loss and adopt a cumulative approach where MSA impacts from land use and climate change are multiplied. For the global loss of species richness the question is more complex because whether local extinctions lead to global extinctions depends on the endemism of species. As a general trend, studies suggest that synergies or reinforcing feedbacks are frequent and in particular, that climate change can accelerate other biodiversity threats such as climate change (Brook, Sodhi &

Bradshaw 2008). Such effects will remain invisible in LCA if characterization models focus on single drivers of biodiversity loss.

5. Conclusions (or Interpretation)

We show that the impact of cattle production on biodiversity on a global scale is not restricted to land use, climate change also plays an important role. While there is a high level of availability and consensus on characterization models to compute land use impacts on biodiversity, this is not the case for climate change. There is a need for development of characterization models on drivers of biodiversity impact that have received less attention until now (climate change, pollution). More importantly, more integrated LCA approaches addressing several drivers should be developed. Integrated approaches are necessary to (i) avoid double counting and address interactions between drivers of biodiversity loss (additivity, synergies) and (ii) to bridge methodological gaps (e.g. taxa and number of species included, consideration of endemism and conservation value) existing between current characterization models that have been developed separately for different drivers.

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164. Are land use and biodiversity midpoint indicators redundant or complementary?

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ABSTRACT

Objective: Land use and biodiversity impact assessments are increasingly important in agri-food Life Cycle Assessment (LCA). These impact indicators are crucial to fully depict the environmental performance of food products in areas of protection insufficiently covered by other impact categories. However, there are disparities in methodologies and types of indicators used and it is unclear whether there is redundancy or complementarity between indicators. This study assesses how characterization factors (CF) for land use and biodiversity are related.

Method: To assess how CFs are related, we performed a correlation analysis using Pearson's r and Spearman's ρ . We correlated land use and biodiversity CFs at country scale to match the geospatial scale of current inventories, and also at eco-region/climate region when possible. We also analyze and interpret conceptual differences in the different models.

Results: Most models are strongly correlated. Among land use indicators, the Brandão and Milà i Canals (2013) and Morais et al (2016) have strong r and ρ correlations, as do de Baan et al (2013) and Chaudhary et al (2015) among biodiversity indicators. Land use SOC depletion indicators are also strongly correlated in general to biodiversity (species loss) indicators. Correlations are weak at the country level, in comparison to eco-regions or climate regions levels, when was possible performed correlations.

Conclusion: The strong correlations between land use and biodiversity indicators permits us to conclude that there is redundancy between approaches and indicators, although methodologic difference between models and indicators used.

Keywords: Land use; Biodiversity; Life Cycle Impact Assessment

1. Introduction

Life Cycle Impact Assessment (LCIA) has been increasingly regionalized. Rather than using single characterization factors (CFs) to depict the contribution of each inventory flow to an impact category, spatialized models are now the norm in cases where the environmental damage has a spatial dimension. For instance, several spatialized land use characterization models have been proposed in the last decade (Vidal-Legaz et al., 2016; Souza et al., 2015). For products deeply rooted in land use processes, namely those where the agricultural stage is particularly important, these spatial models are essential to fully depict the environmental performance in areas of protection insufficiently covered by other impact categories.

While the advent of such spatial models was a necessity, a new issue has surfaced from the availability of different options to model the complex pathways involved in land use impacts – in particular those targeting the effects on biotic production and biodiversity. This issue is how to address the disparities in methodologies and types of indicators used. To quote some important models that provide CFs to practitioners, Brandão and Milà i Canals (2013), Alvarenga et al. (2015), Taelman et al. (2016) and Morais et al. (2016) all present characterization models depicting impacts from land use on biotic production. Regarding biodiversity and ecosystem services impacts from land use, de Baan et al. (2013), Chaudhary et al. (2015) and Cao et al. (2015) are some notable examples. These models are described next.

Brandão and Milà i Canals (2013) is an adaptation from Milà i Canals et al. (2007), which is the model recommended by the ILCD handbook (EC-JRC, 2011) for land use. This model uses Soil Organic Carbon (SOC) as an indicator of soil quality, as a proxy indicator for the biotic production capacity of the soil, translating natural resources and natural environment AoPs effects. This model provides CFs for occupation and transformation processes, covering nine land uses classes (long-term cultivated, full tillage, reduced tillage, no tillage, permanent grassland, paddy rice, perennial/tree Crop, set-aside (< 20 yrs) and sealed land). CFs are aggregated by climate region.

Morais et al. (2016a) are the first operationalization of the same model proposed by Brandão and Milà i Canals (2013) using a large number (more than 19,000) of field measurements for the European Union, namely the LUCASOIL database (Tóth et al., 2013). The model provides CFs for nine land uses classes (evergreen/deciduous needleleaf trees, evergreen broadleaf trees, deciduous broadleaf trees, mixed/other trees, shrubs, herbaceous vegetation, cultivated and managed vegetation,

regularly flooded vegetation and urban/built-up). CFs are aggregated by climate region, ecoregion and NUTS II region, and also by country in the European Union.

Alvarenga et al. (2015) uses Human Appropriation of Primary Production (HANPP) as an indicator. HANPP is the Net Primary Production (NPP) portion that is not available for nature, due to human land use. This indicator reproduces also impacts on the natural resources and natural environment AoPs. This model provides CFs for 5 land use classes (forest areas, wilderness and areas with no land use, infrastructure area, pasture land and cropland), where “cropland” land use is divided by specific crops. This model provides only occupation CFs aggregated by country.

Taelman et al. (2016) uses Net Primary Production (NPP). The model provides country-level CFs for seven main land use classes (forest, agriculture, shrub, grassland, urban, wetland and bare areas), where each class is divided according to the state; e.g. forest class is divided in: protected, virgin, with agricultural activities and with moderate or higher livestock density. This models provides only occupation CFs aggregated by country.

de Baan et al. (2013) uses the total potential damage to biodiversity caused by land use, using potential global extinction of endemic species (mammals, birds, plants, amphibious, reptiles and the aggregated total for all taxonomic groups) as indicators. This model provides occupation and transformation CFs for four land use classes (managed forest, agriculture, pasture, urban areas) for each taxonomic group. CFs are aggregated by ecoregion.

Chaudhary et al. (2015) uses regional and global species loss due to land occupation and transformation (mammals, birds, amphibious, reptiles and aggregated), using the countryside species–area relationship model (Pereira et al., 2014). The model provides occupation and transformation CFs covering six land use classes (intensive forestry, extensive forestry, annual crops, permanent crops, pasture and urban) for each taxonomic group. CFs are provided aggregated by ecoregion and also by country.

Cao et al. (2015) uses the decrease in value of the ecosystem services due to land use, at midpoint and endpoint level, for the cases of biotic production potential (BPP), erosion resistance potential (ERP) and climate regulation potential (CRP). The model provides occupation and transformation CFs covering seven land use classes (forest, shrubland, grassland, pasture/meadow, permanent and annual crops, urban and urban, green areas). CFs are aggregated by country.

Life Cycle Assessment (LCA) practitioners thus need guidance on how to use these models or which they should choose. Usually the approach chosen to provide guidance to practitioners is the direct comparison between modelling choices, assumptions, results and institutional support. For example, the UNEP/SETAC Life Cycle Initiative is a platform promoting life cycle thinking. Its Phase III activities (2012-2017) targeted consensus-building on indicators/models in LCIA for impact categories previously prioritized as highly relevant and with sufficient (Jolliet et al., 2014) such as land use impacts on biodiversity (Teixeira et al., 2016). The Initiative promoted a thorough review of biodiversity assessment of ecological models within the field of LCA and outside (Curran et al., 2016).

The aim of this study is to present a different approach to the typical ways characterization models are compared. Rather than judging model assumptions and data or institutional support, or testing the models for particular case studies, in this paper we assessed how CFs for land use and biodiversity are related, using Pearson's r and Spearman's ρ . Thereby, we assess whether there exists redundancy or complementarity between land use and biodiversity models. This approach can inform practitioners of whether there are substantial differences between models; if there are, a multi-indicator study is probably needed; if no significant differences can be found, it would probably be safe to select only one of the highly correlated models.

2. Methods

To perform the correlation analysis, we used Pearson's r (Pearson, 1895) and Spearman's ρ (Spearman, 1904). Both methods assess the degree of correlation between arrays of CFs aggregate at a given scale. Pearson's r is a measure of the linear correlation between two variables. Spearman's ρ is a nonparametric measure of statistical dependence between two variables and that can be described using a monotonic function. Both methods correlations range between +1 and -1, where 1 is total positive correlation, 0 is no correlation, and -1 is total negative correlation. The main difference

between methods is that Pearson's method assumes a linear relation between variables (and both variables should be normally distributed), while Spearman's method does not make any presuppositions about the distribution of the variables (appropriate when variables are measured on a scale that is at least ordinal).

Correlations are performed at country scale to match the geospatial scale of current inventories, but also at ecoregion/climate region when possible. Cao et al. (2015) and Alvarenga et al. (2015) are only models that provides only CFs at country scale, while the remaining models provide CFs at ecoregion and/or climate region scale. Table 1 shows all models assessed in this study and respective CFs scale. We converted CFs between country/ecoregion/climate region using averages weighted by area (i.e. if two countries belong to one ecoregion, and only country-level CFs are available, the CF for that ecoregion would be the average of the CFs for the two countries weighted by the area of each country).

Table 1: Most relevant land use and biodiversity methods considered and respective scale, land use classes included, indicators and units.

C - Carbon; MJ - Mega Joule; PDF - Potentially disappeared fraction of species; BPP - Biotic production potential BPP; ERC - Erosion resistance potential; CRP - Climate regulation potential

Models	Characterization factor scale	Land use classes	Indicator	Units
Brandão and Milà i Canals (2013)	Climate region		Soil organic carbon	kg C deficit
Alvarenga et al. (2015)	Country	Forest areas, wilderness and areas with no land use, infrastructure area, pasture land and cropland	Human appropriation of primary production	kg dry matter
Taelman et al. (2016)	Country	Forest (eight classes), agriculture (six classes), shrub (five classes), grassland (five classes), urban (one class), wetland (four classes) and bare areas (four classes)	Net primary production	MJ _{ex}
Morais et al. (2016a)	Ecoregion, climate region and NUTS II region	Evergreen/deciduous needleleaf trees, evergreen broadleaf trees, deciduous broadleaf trees, mixed/other trees, shrubs, herbaceous vegetation, cultivated and managed vegetation, regularly flooded vegetation and urban/built-up	Soil organic carbon	kg C
de Baan et al. (2013)	Ecoregion	Managed forest, agriculture, pasture, urban areas	Potential global extinction of endemic species	-Potentially lost endemic species (for taxonomic groups) -PDF (for aggregated class)
Chaudhary et al. (2015)	Ecoregion and country	Intensive forestry, extensive forestry, annual crops, permanent crops, pasture and urban	Global species loss	-Global species eq. lost (for taxonomic

Models	Characterization factor scale	Land use classes	Indicator	Units
				groups) -PDF (for aggregated class)
Cao et al. (2015)	Country	Forest, shrubland, grassland, pasture/meadow, permanent and annual crops, urban and urban, green areas	BPP, ERP and CRP	-BPP: ton carbon yr ⁻¹ -ERP: ton soil -CRP: ton C

Regarding the correspondence between models' land use classes, Brandão and Milà i Canals (2013) do not provide CFs for the "forest" land use class, while Morais et al. (2016a) suggests that the "forest" class is an average of evergreen/deciduous needleleaf trees, evergreen broadleaf trees, deciduous broadleaf trees and mixed/other trees classes. For Taelman et al. (2016) we aggregated the CFs using an average for each of the main seven land use classes, to allow comparability with the other models. In all other cases we used the original classes from studies. In some cases we made assumptions regarding the nomenclature of classes. We assumed that the "pasture" land use class in Morais et al. (2016a) was equivalent to the "herbaceous vegetation" class, and to the "grassland" class from Cao et al. (2015). Agriculture and urban land use classes required no adaptations.

3. Results

Table 2 shows Person's r correlation for all models assessed using occupation CFs and the "agriculture" land cover, at country scale. For occupation CFs (and not transformation) all models may be correlated, since Alvarenga et al. (2015) and Taelman et al. (2016) provide only occupation CFs.

Land use models that use SOC depletion are significant correlations, mainly between Brandão and Milà i Canals (2013) and Morais et al. (2016a) where all correlation are above 0.5. These are the highest achievable correlations. Figure 1 presents the occupation CFs' spatial representation of these two models for the "agriculture" land use class in Europe. The correlation between SOC depletion (Brandão and Milà i Canals (2013) and Morais et al. (2016a)) with HANPP and NPP is not significant at the 5% level if CFs, at all scales and both aggregation approaches.

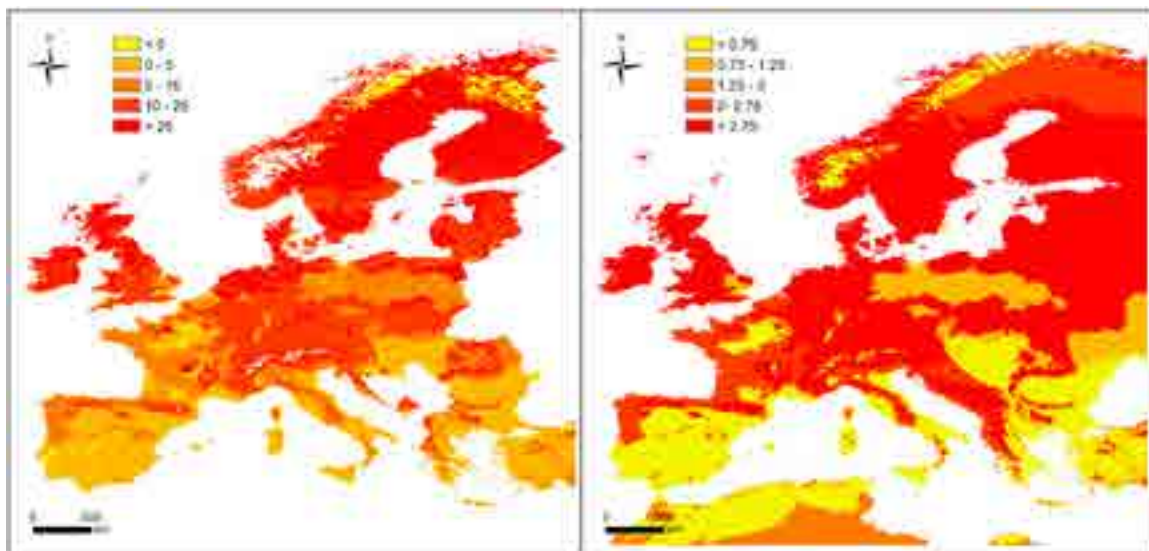


Figure 1: Occupation characterization factors for "agriculture" land use class, according Morais et al. (2016a) (left) and Brandão and Milà i Canals (2013) (right).

In biodiversity models, the correlation between de Baan et al. (2013) and Chaudhary et al. (2015) is significant only in the taxonomic group of mammals, birds and amphibious. The others taxonomic groups are not significant correlated (using country-level aggregation) as the absolute value is close to zero. However, the correlations at ecoregion scale are all significant and higher than 0.5. This is probably due to a distortion effect caused by the conversion from ecoregion (original) to country aggregation. The correlations with Cao et al. (2015) are low. This fact can be justified due the fact that there is complementarity between models, as verified between NPP and HANPP with SOC depletion, in land use indicators.

The correlations between Morais et al. (2016) and Brandão and Milà i Canals (2013) with biodiversity models are negative due to the way models are constructed, i.e. when SOC depletion is higher, species abundance should be lower. Alvarenga et al. (2015) and Taelman et al. (2016) are the opposite case, when compared with the other two land use models. The correlations between land use, using Morais et al. (2016a) (ecoregion and climate region aggregations), and biodiversity models are significant and higher than 0.5, independently of group taxonomic, as well as with Cao et al. (2015). The NUTS II regions aggregation led to lower or non-significant correlations, which can be justified by aggregation problems previously justified by the authors (Morais et al., 2016a). Alvarenga et al. (2015) and Taelman et al. (2016) are weakly correlated (in absolute value), when compared with Morais et al. (2016a), but the correlations are still significant at 5% level. However, Brandão and Milà i Canals (2013) correlations are the lowest and mostly non-significant, which may also demonstrate some issues with the conversion from climate regions CFs (original) to country aggregation.

Results for other land use classes (rather than agriculture) assessed in this study are similar to those shown in Table 2 for “agriculture”, but absolute values are lower and in fact for “forest” almost all correlations are not statistically significant at 5% level. Further, results from Pearson’s r and Spearman’s ρ are similar, despite methodologic differences between Pearson and Spearman methods. This may mean that the linearity assumption required by Pearson’s method is fulfilled in the CF datasets.

4. Discussion

Land use and biodiversity indicators are currently some of the most discussed topics in LCIA. Thus far the evaluations of models are based on the conceptual *ex ante* comparison of their assumptions and characteristics. This was the procedure followed by the UNEP/SETAC Life Cycle Initiative (Curran et al., 2016). The novelty of our study is that we looked at outcomes of models, which has two main advantages. First, even models with different assumptions can have similar results. It is important to know that a simplified model that does not score high on the quality of its assumptions may be as accurate as a more complex one that, *a priori*, looks more thorough. Second, many of these models use different indicators and cover different aspects in the impact pathway from land use to biodiversity. Ultimately, however, they may be expressing very similar or interconnected issues. Assessing the CFs allowed us to determine their complementary or redundancy. This could not have been assessed using conceptual analyses only.

This study results suggests redundancy between some land use and biodiversity models. Morais et al. (2016a) is the land use model with highest and most statistically significant Pearson’s r and Spearman’s ρ correlation not only with other land use models, but also with biodiversity models. Soil carbon is an overarching indicator connected to many environmental issues, but beforehand there was no reason to believe it could be as strongly connected with biodiversity loss as this analysis indicates. Although Morais et al. (2016a) was only an updated version of the model proposed by Brandão and Milà i Canals (2013) (with no methodological updates), the use of local data for the European Union was crucial to calculate CFs that are deeply connected with other indicators. In fact, the original model is the least correlated with biodiversity models (close to zero). This fact suggests that regionalization is an essential issue in impact assessment, as verified in Morais et al. (2016b) in Portuguese studies.

Even in models where we aggregated land use classes (the “forest” class in Morais et al. (2016a) and all classes in Taelman et al. (2016)), it was still possible to correlate models significantly, i.e. at

5% level (e.g. correlation with de Baan et al. (2013) and Chaudhary et al. (2015), independently of taxonomic group).

Note, however, that the concept of statistical correlation as we analyze it here just measures spatial coherence between models (highest/lowest impacts take place in the same regions). It does not validate each individual model, and it says nothing about the absolute values of the CFs which, in many cases, vary considerably even for similar indicators. Significant correlation does not mean that there are no differences in absolute value. For example, Morais et al. (2016a) are strongly and significantly related to Brandão and Milà i Canals (2013), but in absolute value CFs from Morais et al. (2016a) CFs are about 10 times higher, although both models used the same methodology. The models difference are only that Brandão and Milà i Canals (2013) used 30 cm depth and data from IPCC (2006) whereas Morais et al. (2016a) used a 20 cm depth and field measurements for the European Union.

Finally, note that this study relates land use and biodiversity models that use different indicators, though these indicators are related between them according to the impact pathway proposed by Koellner et al. (2013). According to this pathway, for example, SOC is one midpoint indicator conducting to ecosystem services damage potential (endpoint), as also CRP and ERP. Thus, models assessed should be expected to be related between them (e.g. SOC depletion should be related to species loss or HANPP should be related do BPP). However, even statistically significant correlations are far from 1 (perfect correlation). This lack of perfect redundancy was also expected, as different models target particular elements of the land use impact pathway.

5. Conclusions

Land use and biodiversity models show some strong correlations, despite methodological differences between models and indicators used. These findings suggest that some land use and biodiversity models may be redundant. Two examples are Morais et al. (2016a) and Brandão and Milà i Canals (2013), but also (and more surprisingly due to the different methods used to calculate the CFs) between Morais et al. (2016a) and Chaudhary et al. (2015). NPP and HANPP models relates weakly and non-significantly correlated with other models, except with Cao et al. (2015). These conclusions were drawn due to the method of spatial correlation used, and would not be discernible using simple textual or conceptual comparisons of assumptions and characterization methods in each model.

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Table 2: Pearson's r correlation coefficient between CFs for "agriculture" land cover class at the country scale. Significant correlations at the 5% level are indicated with an *
 C.R.: Climate region; E.R.: Ecoregion; N.R.: NUTS II region; M.: Mammals; B.: Birds; P.:Plants; Am.: Amphibians; R.: Reptiles; Ag. Aggregated; BPP: Biotic production potential; ERP: Erosion resistance potential; CRP: Climate regulation potential

de Baan et al (2016)			Brandão & Milá i Canals (2013)	Alvarenga et al (2015)	Taelman et al (2016)	de Baan et al (2013)					Chaudhary et al (2015)			
E. R.	N. R.					M.	B.	P.	Am.	R.	Ag.	M.	B.	Am.
1			0.791*	-0.095	-0.293	-0.718*	-0.672*	-0.718*	-0.714*	-0.752*	-0.737*	-0.786*	-0.454*	-0.000
0.73*	1		0.550*	0.290	-0.482*	-0.642*	-0.611*	-0.601*	-0.682*	-0.574*	-0.649*	-0.697*	-0.335	-0.000
0.71*	0.709*	1	0.567*	0.065	-0.310	-0.420*	-0.378	-0.360	-0.481*	-0.369	-0.423*	-0.488*	-0.265	-0.000
			1	0.155	0.075	-0.010	-0.058	-0.054	0.033	-0.035	-0.045	-0.035	-0.094	-0.000
				1	0.338*	0.303*	0.286*	0.296*	0.307*	0.323*	0.363*	0.508*	0.404*	0.000
					1	0.339*	0.279*	0.281*	0.282*	0.310*	0.350*	0.384*	0.190*	0.000
						1						0.414*		
							1						0.144*	
								1						0.000
									1					
										1				
											1			
												1		
													1	
0.36*	0.570*	0.504*	-0.165*	-0.121	0.208*	-0.082	-0.142*	-0.069	-0.028	-0.009	-0.115	-0.176*	-0.195*	-0.000
0.339	0.027	0.191	0.105	0.477*	0.453*	-0.026	-0.016	-0.017	-0.004	-0.011	-0.014	-0.032	-0.028	-0.000
0.32*	-0.559*	-0.329	-0.059	0.311*	0.430*	0.187*	-0.015	0.024	0.177*	-0.002	0.004	0.125	0.127	0.000

189. Global guidance on land use impacts on biodiversity indicators in LCA

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ABSTRACT

The life cycle impact assessment guidance flagship project of the UNEP-SETAC Life Cycle Initiative aims at providing global guidance and building scientific consensus on indicators for assessing land use impacts on biodiversity. This paper summarizes the activities and highlights the main conclusions of the project.

During the last two years the Initiative conducted four main tasks: 1) organization of workshops where invited experts and stakeholders discussed and agreed on recommendations for assessing biodiversity loss due to land use interventions 2) a critical review of the existing framework for land use impact assessment in LCA; 3) a review of existing biodiversity models in and out of the field of LCA in order to identify models of particular promise for further application and development and 4) a Pellston Workshop™, where conclusions were drawn from the prior tasks with participation of selected experts.

From the workshops and this review process we concluded that an acceptable model should reflect both local measures of land use intensity and regional weighting reflecting vulnerability aspects. Finally, during the Pellston Workshop™ the Potential Species Loss from Land Use (Regional and Global - PSL_r and PSL_g) were provisionally recommended to assess biodiversity loss due to land use. The recommended model links land use to species loss through the Countryside-SAR model and includes vulnerability scores expressed as ratio of threatened endemic richness with the total species richness hosted by the region. In addition, recommendations to improve the indicator were also suggested.

Keywords: Potential Species Loss, Regional and global change

1. Introduction

Land use (LU) and land use change (LUC) are main drivers of biodiversity loss and degradation of a broad range of ecosystem services. The conservation of biodiversity represents a global priority due to its substantial contribution to human well-being (MA, 2005) and ecosystem functioning (Mooney and Mace, 2009). The term “biodiversity” is plural and encompasses a wide range of biological features with distinct attributes (ecological composition, function and structure), and is nested into multiple levels of organization (genetic, species population, community, and ecosystem). Due in part to this complexity, and despite substantial contributions to address biodiversity in Life Cycle Assessment (LCA), no clear consensus existed on the use of specific impact indicator(s) to quantify land use impacts on biodiversity within LCA (Souza et al 2015).

The life cycle impact assessment guidance flagship project of the UNEP-SETAC Life Cycle Initiative aims providing global guidance and building scientific consensus on indicators for assessing, among others, land use impacts on biodiversity (Jolliet et al., 2014). This paper summarizes the activities and highlights the main conclusions of the project.

During the last two years the Initiative conducted four main tasks: 1) organization of three workshops where invited experts and stakeholders discussed and agreed on recommendations for assessing biodiversity loss due to land use interventions (Teixeira et al 2016); 2) a critical review of

the existing framework for land use impact assessment in LCA; 3) a review of existing biodiversity models in and out of the field of LCA in order to identify models of particular promise for further application and development (Curran et al 2016) and 4) a Pellston WorkshopTM, where conclusions were drawn from the prior tasks with participation from selected experts in the area.

2. Methods

2.1 Workshops conducted

Three expert and stakeholders consultations were organized to obtain feedback from specialists on the main aspects to be considered when modeling impacts on biodiversity. Several specialized domain experts on the land use impacts, in biodiversity metrics, LCA and ecology, as well as institutional and business partners representing the user community, were invited. The first 'Expert Workshop on Biodiversity Impact Indicators for Life Cycle Assessment' took place in San Francisco, on October 7th, 2014, hosted by the US EPA. The workshop was attended by fourteen invited experts. The second workshop took place in Brussels, on November 18th and 19th, 2014, hosted by the Alliance for Beverage Cartons and Environment (ACE). It was attended by twenty-four invited experts from equally diverse backgrounds. The events included discussions centered on four key areas where existing approaches differ the most and consensus is required. These topics are: (a) concept of biodiversity and modelling strategies, (b) data availability and feasibility, (c) desired characteristics of indicators, usability and consensus and (d) concerns and limitations about using biodiversity indicators in LCA. In addition, a stakeholder consultation was held in Brazil, on November 12th, 2014, gathering about 40 participants from Brazilian academia and industry. Key concerns expressed related to the potential unfair results when allowing comparison between very different realities; and to treating different biomes as equal (geographically and temporally). Such considerations lead to methodological aspects related to the choice of reference state, in particular. The importance of considering biodiversity and ecosystem services together (at least considering indicators beyond species richness, including ecosystem quality) was highlighted too (Teixeira et al. 2016).

2.2 Impact pathway review

All relevant aspects potentially affecting the impact pathway to damage on biodiversity have been considered. Following expert advice and taking the proposal of Koellner et al. (2013a) as the starting point, a thorough review on which aspects need to be considered in the land use impact pathway was conducted. An important subject of debate was the risk of double accounting biodiversity effects for different impact categories (e.g. ecotoxicity, land use climate change). This discussion also considered the on-going work by the European Commission's Joint Research Centre (JRC) in the frame of the Product Environmental Footprint (PEF) (Vidal et al, 2016). JRC's focus is on midpoint land use categories, while this task force aims at achieving biodiversity loss indicator(s) closer to endpoint ecosystem damage; the collaboration is thus focused on land use interventions and impact pathways, which are independent of the final assessment.

2.3 Model evaluation

In order to provide an overview of cutting-edge impact assessment approaches and methods relevant the Biodiversity Land Use Task Force identified 73 publications in the literature. 31 out of the 73 publications matched the criteria for method documentation and were found suitable for characterization. The documentation criteria used in the selection were the following: 1) the main description of the model was published in a peer-reviewed scientific journal, and 2) models should enable characterization of impacts on biodiversity in at least two different land use/cover classes or intensities of generic land use archetypes (e.g. forest or agriculture, intensive or extensive). Among the 31 methods that passed the selection (see Curran et al. 2016), 20 were developed specifically for environmental impact assessment in LCA and 11 were proposed from non-LCA domains (environmental policy, ecology and conservation).

2.4 Pellston WorkshopTM

From 24th to 29th January, 2016, a Pellston WorkshopTM, took place in Valencia, Spain. A group of selected experts were invited to discuss and agree on indicators to be recommend based on the work

performed previously by the task force. The final goal of the Pellston Workshop™ was to provide recommendations on impact assessment indicators and their characterization factors representing land use impacts on biodiversity, for a specific enough land use classification system (based on Koellner et al. 2013) and at sufficient geographic differentiation level, or, alternatively, provide clear guidance on how to reach them.

3. Results

3.1 Workshops conducted

The experts who participated in the expert workshops stressed the importance of the framework established by the Life Cycle Initiative, appreciating the early engagement of stakeholders in the consensus-building process. Experts agreed that LCA should go beyond the simple summation of elementary flows for LU/LUC (squared meters transformed or occupied each year). Experts believed there is sufficient data and quality models to relate those elementary flows to their respective impacts on biodiversity, while paying attention on how final results of LCA studies are communicated. Moreover, there was also an agreement that a good LCA indicator for biodiversity necessarily has to consider geographical location, several aspects that depict the state of ecosystems at that location, and a measure of land use intensity. Species richness was considered a good starting point for assessing biodiversity loss. However, complementary metrics need to be considered in modelling, such as habitat configuration, inclusion of fragmentation and vulnerability (Teixeira et al. 2016).

3.2 Impact pathway review

From inventory flows (land use interventions) to endpoint damage (biodiversity loss), there is a very complex pathway with several interconnections, including impacts on habitat structure. As a starting indicator, species richness is used, acknowledging that ecosystem and genetic levels are not explicitly addressed. LCA reports loss in species as biodiversity damage potential (BDP). The higher the damage potential the more species are potentially lost. A small BDP has to be perceived as positive. It is important to highlight the effort performed in order to simplify inventory collection as well as the agreement in final endpoint towards a combination of local biodiversity damage and a regional/global weighting factor. The agreement and definition on pathway helped to perform evaluation and selection on recommended indicators (see next sections).

3.3 Model evaluation

Curran et al. (2016) provide a detailed and comprehensive evaluation of the 31 methods reviewed. Main results from the revision conducted showed that approaches in LCA are nested within those from ecology and conservation as shown in Figure 1. These are the results of applying Gower similarity coefficient which is a composite measure of different type of variables (quantitative, binary and nominal). As a summary of results, we conclude that the most common pathway assessed was the direct, local degradation and conversion of habitats. Regarding biodiversity representation most of the current models are based on compositional aspects of biodiversity, namely species richness followed by species abundance. Different spatial scales of assessment have been used being ecoregion the one with the highest potential for consensus. Several taxonomic groups are covered by distinct models, plants being the most common taxon assessed across models. Measures of habitat quality are largely subjective in nature, and include the “naturalness” of land cover classes (i.e. “Hemeroby” scores, Brentrup et al. 2002). At the regional scale, indicators of the overall species pool size were most common, followed by habitat quality and extinction risk.

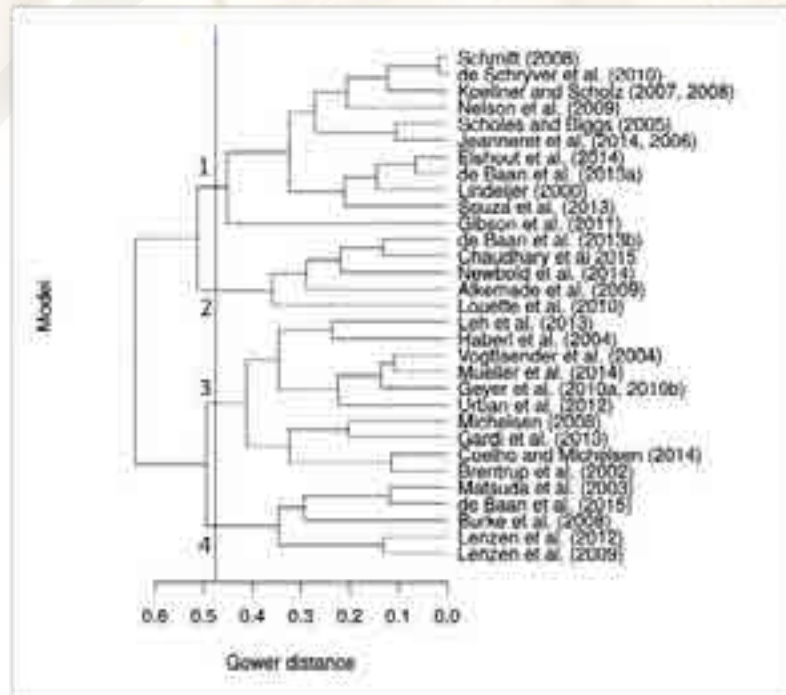


Figure 1: Complete-linkage hierarchical clustering dendrogram applied to quantitative scores showing that approaches in LCA nested within those from ecology and conservation measured using Gower distance, which gives an idea of proximity between models with different data types.

3.4 Pellston WorkshopTM

From the workshops and this review process we concluded that an acceptable model should reflect both local measures of land use intensity and regional weighting reflecting vulnerability aspects. The local impact component puts the main focus on what and how an activity is performed, while the regional/global impact component puts the main focus on where an activity is performed. During the Pellston WorkshopTM the experts present considered that the approach proposed by Chaudhary et al. (2015) (fig 2) fits those criteria. This model was provisionally recommended by the experts attending the Workshop to assess biodiversity loss due to land use. The approach links land use to species loss using the Countryside-SAR model and includes vulnerability scores expressed as ratio of threatened endemic richness with the total species richness hosted by the region.

The indicator selected is the potential species loss (PSL) from land use based on the method described by Chaudhary et al. (2015). The indicator represents regional species loss taking into account the effect of land occupation displacing entirely or reducing the species which would otherwise exist on that land, the relative abundance of those species within the ecoregion, and the overall global threat level for the affected species. The indicator can be applied both as a regional indicator (PSL_{reg}), where changes in relative species abundance within the ecoregion is included, and as a global indicator (PSL_{glo}), where also the threat level of the species on a global scale is included.

The indicator covers 5 taxonomic groups; birds, mammals, reptiles, amphibians and vascular plants. The taxonomic groups can be analyzed separately or can be aggregated to represent the Potentially Disappeared Fraction (PDF) of species. Land use types covered by the method include intensive forestry, extensive forestry, annual crops, permanent crops, pasture and urban land. The reference state is a current natural or close to natural habitat in the studied ecoregion.. The model provides characterisation factors down to 804 ecoregions based on Olson et al. (2001), as well as country level and global average characterisation factors. The characterisation factors are provided for both land occupation in global PDF eq. $lost \cdot m^{-2}$ and land transformation in global PDF eq. $lost \cdot year \cdot m^{-2}$. The model includes both average and marginal factors.

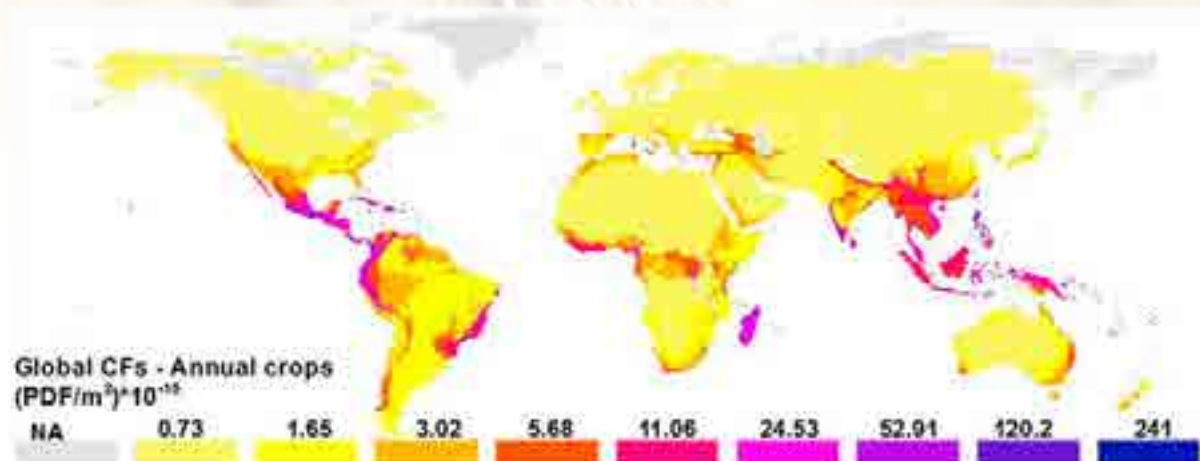


Figure 2: Example for Annual crops median taxa-aggregated global CFs for the land use type “annual crops” per ecoregion expressed as Potentially Disappeared Fraction. (Chaudhary et al. 2015).

Figure 3 highlights (in colour) those aspects of the impact pathway, defined for land use impacts on biodiversity, covered by this approach. Among the different reasons why the experts recommended the model by Chaudhary et al. 2015 we would like to stand out that this method takes into consideration many of the important aspects identified by stakeholders over the past two years of work by the task force, including: it builds on species richness; incorporates the local effect of different land uses on biodiversity; links land use to species loss through the Countryside-SAR model; includes the relative scarcity of affected ecosystems; and includes the threat level of species (from IUCN lists, aggregating species vulnerability of specific habitats at the global level); it allows global coverage of six major land use types, thus enabling the consideration of biodiversity impacts across most products’ life cycles. However, there are some aspects of the model not yet mature enough to merit full recommendation. Based on its limitations, at this moment, the Initiative can only recommend it provisionally until additional land use types (i.e. intensities) are added for agricultural systems and sufficient case studies are undertaken to test the robustness and ability of the model to identify potential biodiversity impacts.

Because the recommendation includes the restriction of the use of the method to hotspot identification only, additional guidance for practitioners and/or environmental managers to follow up on hotspot investigation was provided:

If a potential hotspot is identified in the foreground system:

1. Specify the ecoregion where the process occurs to increase accuracy in your results and review the regional characterization factors for further insights into the main drivers of the hotspot.
2. Determine the local land use type and management characteristics/regime
3. Use more geographically specific or sector-specific biodiversity assessment methods, possibly including those that identify the conditions for maintained biodiversity (Michelsen 2008, Lindqvist et al. 2016); identify the criteria for responsible sourcing from that region or within a certain sector (e.g., LEAP (2015) guidelines for the livestock sector).
4. Take appropriate environmental management actions based on additional information.

If the potential hotspot is detected in the background system: try to increase the accuracy of the results (country specific CFs instead of default values) using expert judgment to understand the relevance. If relevant, start at 3 above and follow the same steps.

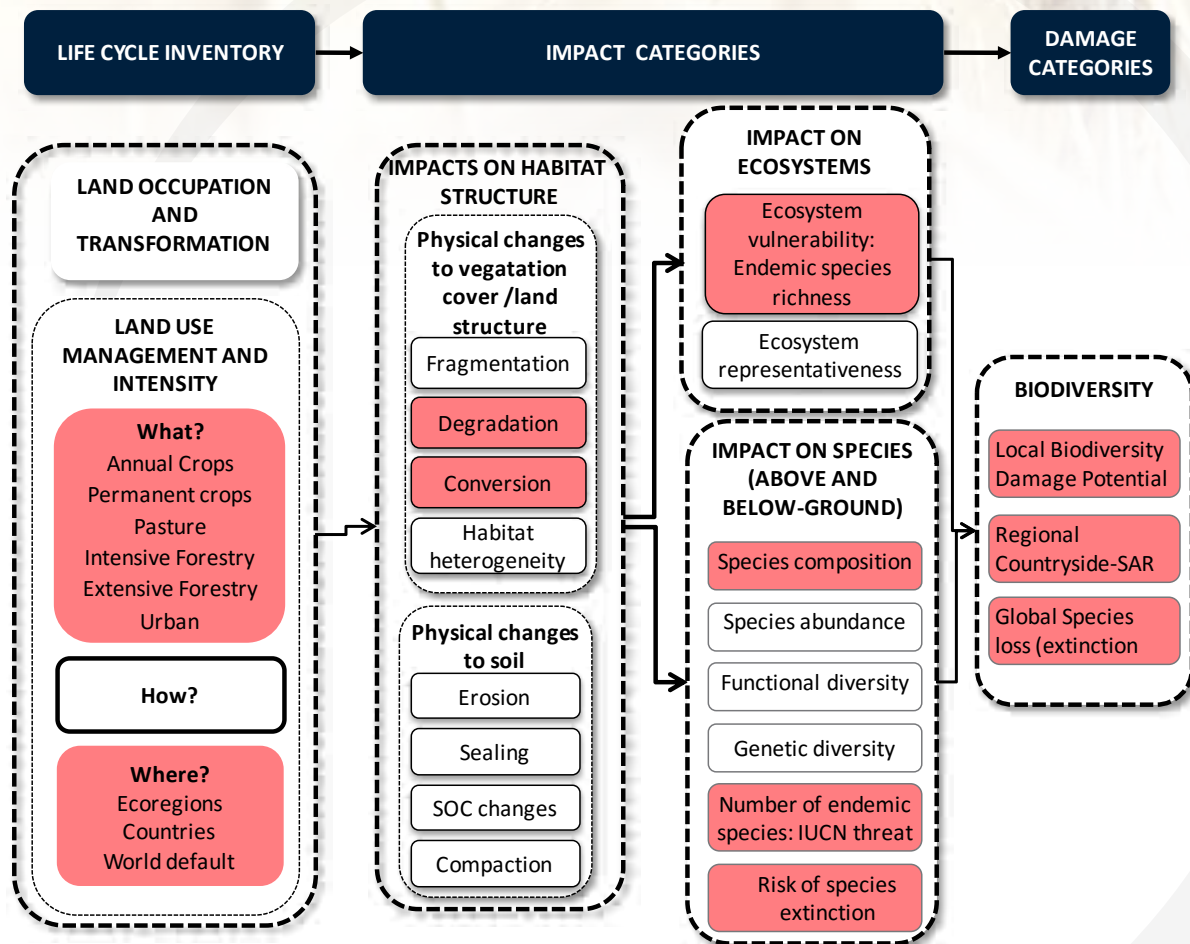


Figure 3: Aspects of the Impact Pathway for land use impacts on biodiversity covered by the Chadhaury et al. 2015 method.

4. Conclusions

The life cycle impact assessment guidance flagship project of the UNEP-SETAC Life Cycle Initiative recommends the Potential Species Loss from Land Use (Regional and Global - PSLreg and PSLglo) as a consensual indicator for assessing land use impacts on biodiversity. It provisionally recommends the global average CFs based on the method developed by Chaudhary et al. (2015) as suitable to assess impacts on biodiversity due to land use and land use change as hotspot analysis in LCA. Recommending the use for hotspots analyses only means that these CFs should not be used for comparative assertions and product labelling. Conditions required to move from a provisional recommendation to a full recommendation include adding additional land use types (i.e. intensities) for agricultural systems and undertake sufficient case studies to test the robustness and ability of the model.

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139. Defining the reference situation for biodiversity in Life Cycle Assessments: Review and recommendations

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ABSTRACT

Biodiversity – crucial for ecosystem health and its products and services – is being lost at an alarming rate. Several models that aim to assess biodiversity impacts in Life Cycle Assessments have been proposed, but there are still some obstacles before these models become biologically realistic and feasible. One of the major challenges to be addressed is the identification of an appropriate reference situation for biodiversity, to serve as a point of comparison for assessing changes over time. However, this reference could be a point in the past, present, or future, and its choice will imply different interpretations of land use and land transformation impacts. Here we provide a short introduction to this topic and outline the main challenges as well as possible solutions to properly incorporate biodiversity reference situations in Life Cycle Assessments.

Keywords: life cycle impact assessment, LCA, biodiversity impact, land use, reference situation

1. Introduction

The fundamental goals of LCA are the compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system (covering both goods and services) across the whole life cycle, to aid in decision-making, choice of environmental performance indicators and support market claims (ISO., 2006) often aiming at improvement in the system (Baumann & Tillman, 2004). Since LCA-models are powerful decision aids, they can be helpful, but also harmful if a model's foundational assumptions and/or its results are misinterpreted. This stresses the need for standardized approaches and justification of methodological choices made so that studies are not only transparent and relevant for decision-making but also reproducible and comparable.

Within the framework of the UNEP-SETAC International Life Cycle Initiative, a partnership formed by the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC), an international panel of LCA experts has completed the Land Use Life Cycle Impact Assessment (LULCIA) project (Koellner et al., 2013). Following on some other pioneers, the LULCIA project established preliminary methods for incorporating land use impacts in LCA. One crucial variable in this framework is the reference situation, that is the baseline for biodiversity which to compare actual land use to. Its definition is a key element in the calculation of impacts in the temporal-spatial model and must be established as a point of comparison to assess changes in land quality over time (Koellner et al., 2013). However, there is no consensus on how to best establish this reference situation. This reference could be a point in the past, present or future (de Baan et al., 2013 (1)) and either located in the region of production or nearby.

The need for a reference situation for biodiversity is however not just specific to LCA, but to conservation assessment, monitoring and planning in general. From a policy and management perspective, every ecosystem or biodiversity indicator must be linked to a reference level to

provide a context so that changes in indicator values can be interpreted relative to a desired state (Pitcher, 2001; Rice, 2003).

The aim of this paper is to assess the main challenges as well as identify possible solutions to properly incorporate biodiversity reference situations in Life Cycle Assessments. First, a literature review was carried out to find out which reference situations for biodiversity are proposed in impact assessment literature (i.e. reference situation in theory) and how they are applied (i.e. reference situations in practice). Findings from the literature review were then used to discuss the relevance and applicability of different reference situations from a biodiversity perspective. Finally, an evaluation matrix for biodiversity reference states is proposed.

2. On reference situations

There are a number of different kinds of reference situations for use in various assessments. Jennings and Dulvy (2005) suggest the following five categories: (a) baseline, which is a reference derived from time periods or locations free from human pressures, (b) normative reference levels, which is a reference defined based on what is socially acceptable, i.e., according to norms, (c) reference points, which are precise values of indicators used to provide context for the current status of an indicator, (d) limit reference which, if exceeded, indicate that the system or object will be subject to serious or irreversible harm and (e) target reference, which is a reference level that signals a desired state.

In principle, a baseline reference is derived from time periods or locations free from human pressures. The term can thus be used for an ecosystem state prior to (substantial) human impact, during some 'baseline' time period, but also inside areas protected from human impacts or remote geographic locations subject to minimal human pressures. Note that the three options described above (before any human impact, during some reference 'point' in time or in protected/set-aside locations) are three different types of baseline situations and recognizing the differences between them is crucial to avoid the shifting baselines syndrome (Pauly, 1995). The shifting baseline syndrome describes how our perception of an earlier (desirable or 'natural') situation shifts, when baselines are set with a short-term perspective and thus represent an increasingly impacted state over time. In other words, if nature changes and we only remember the near past - we may confuse this near past with the 'original past'. This might mainly be a risk when using reference situations as current climax state or (semi)-natural land.

3. Review of reference situation applied in LCA

Several potential reference situations are proposed in the existing LCA literature, some of which are presented in Table 1. In the UNEP-SETAC guideline on global land use impact assessment in LCA, Koellner et al. (2013) identify three options for establishing a reference situation. One is the so called Potential Natural Vegetation (PNV), described as the expected natural state of the ecosystem in the absence of human intervention. The concept of PNV, however, has been interpreted both as a future hypothetical state after all human interventions had stopped, and as a pre-anthropogenic disturbance state, i.e. some kind of "climax" natural state (Chiarucci et al., 2010). With both of these interpretations at hand there are several studies using PNV as a baseline included in Table 1. The second option, recommended by Koellner et al. (2013) for assessment of land use impacts on a global scale, is to use as reference situation "the (quasi-) natural land cover in each biome/ecoregion, i.e. the natural

mix of forests, wetlands, shrubland, grassland, bare area, snow and ice, lakes and rivers.” This is applied in studies 10 to 18 (in Table 1). In this group we also include de Baan et al. (2013 (2)) who choose to use “the current, late-succession habitat stages as reference, which are widely used as target for restoration ecology and serve as a proxy for the Potential Natural Vegetation.” A nearly-natural land state is described as “(semi)-natural” to illustrate that the reference state may not precisely match the (hypothetical) pre-human habitat due to the different degrees of human disturbance in the past (e.g. certain European landscapes where essentially no undisturbed regions exist). Here are also included studies that apply some kind of relaxation potential as reference, as this implies the return to a “natural state”. The third option is to use the present land use mix (i.e. current mosaic of land cover in a region), as recommended by studies 19 to 22 (Table 1).

Many other proposals have been made, fueling confusion within the topic and often precluding a practical implementation of the concept by policy-makers and industry (Table 1).

Table 1: main models proposed for including biodiversity in LCA, included in the literature review

No	Paper	Reference situation in theory	Reference situation in practice
1	Blonk et al. (1997)	Pre-anthropogenic disturbance state	Not specified
2	Brenttrup et al. (2002)	Climax state	No human influence, e.g. Untouched rocky, peat bog and tundra regions
3	de Baan et al. (2015)	Pre-anthropogenic disturbance state and Current regional state	Not specified
4	de Schryver et al. (2010)	Pre-anthropogenic disturbance state	Natural forest in region
5	Elshout et al. (2014)	Pre-anthropogenic disturbance state	Represented by: (semi-) natural land in 1: same region, 2 same ecoregion, 3 same biome
6	García-Quijano et al. (2007)	Climax vegetation	Site-specific ecosystem phase with the highest exergy content and the highest exergy flow dissipation capacity. Hypothetical extreme which assumed the conservation of the original climax vegetation cover
7	Scholes & Biggs (2005)	Pre-anthropogenic disturbance state	Represented by: contemporary populations in large protected areas or baseline year with records or reliable memory
8	De Souza et al. (2013)	Potential natural vegetation	Represented by (semi-) natural land, I.e. Primary forest
9	Wagendorp et al. (2006)	Climax vegetation/undisturbed nature	Site-specific ecosystem phase with the highest exergy content and the highest exergy flow dissipation capacity
10	de Baan et al. (2013 (1))	(semi-) natural land	Current, late-succession habitat stages

11	de Baan et al. (2013 (2))	(semi-) natural land	Current, late-succession habitat stages
12	Coelho & Michelsen (2014)	Natural state (relaxation potential)	Not specified
13	Michelsen (2008)	Natural state (relaxation potential)	Not specified
14	Penman et al. (2010)	Natural state	Perceived natural form in broad sense, not otherwise specified
15	Peter et al. (1998)	Natural forest	Non-managed forest
16	Schmidt (2008)	Natural state (relaxation potential)	Not specified
17	Toffoletto et al. (2006)	Natural state (relaxation potential)	Not specified
18	Weidema & Lindeijer (2001)	Natural state (relaxation potential)	Late -succession habitat stages
19	Baitz et al. (2000)	Present land use (Reference point)	State immediately before studied activity
20	Geyer et al. (2010)	Present land use (Reference point)	None/initial land use scenario
21	Vogtländer et al. (2004)	Present land use (Reference point)	Quality immediately before studied activity
22	Bare et al. (2003)	Present state (Reference point)	Current density of threatened and endangered (T&E) species in a specific area
23	Koellner & Scholz (2007, 2008)	Current regional status	Regional average land use mix
24	Milà I Canals et al. (2007)	Dynamic reference situation	Depends on scope of LCA
25	Lindqvist et al. (2015)	Desired state of biodiversity as defined in national strategy documents	A hypothetical maximum quality of biodiversity, including states of above natural biodiversity due to management of the landscape in line with, e.g. cultural heritage

4. Applicability of reference situation

Whereas Life-Cycle Impact Assessment (LCIA) models for land use and land use change mainly focused on baseline reference situations such as pre-anthropogenic disturbance state, (semi) natural state and relaxation potential (Table 1), some fairly well established reference levels in biodiversity management practice are defined differently. Examples include target population sizes for recovery of endangered species (Gerber & Hatch, 2002), the harvest rate corresponding to maximum sustainable yield in a fishery (Walters & Martell, 2004), the critical level of nutrient input beyond which a clear freshwater lake becomes turbid (Schindler, 1974) and, acceptable concentrations of toxic contaminants in water bodies (Suter, 2001). Also in Life Cycle Impact Assessment models limit and/or target reference levels have already been successfully applied. In the calculation of acidification and eutrophication potentials critical loads serve as limit reference, while toxicity potentials have been calculated based on the ratio of Predicted (No) Effect Concentration, PEC/PNEC (Huijbregts, 1999).

In order to be societally useful, LCIA models aiming to include biodiversity impacts need to address such biodiversity impacts that are applicable for current conservation strategies and policies. Several of the proposed reference situations listed in Table 1, especially those aiming at pristine nature (baseline reference situations as pre-anthropogenic state, climax state, natural state, relaxation potential), differ considerably from conservation targets as, for example, the global Convention on biodiversity (CBD) which is based on the vision "Living in Harmony with Nature" where "By 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people." (UN CBD, 2010). The European Species and Habitats Directive (Council Directive 92/43 EEC) aims at maintaining or restoring European protected habitats and species listed in the Annexes at a favorable conservation status – many of the listed habitats are anthropogenic and their species dependent on continued land-use (The Council of the European Communities, 2013). So far, LCIA models have largely been based on baseline reference situations rarely used in the society's conservation frameworks as that of the CBD and The European Species and Habitats Directive. An exception to this is the reference state applied in the biodiversity impact assessment method proposed by Lindner et al. (2014), and tested by Lindqvist et al. (2015), which is linked to policy targets.

Distinctions between different baselines are not clear when comparing how reference situations have been chosen in LCA studies, see Table 1. It seems to be a struggle to find a reference that is both meaningful and measurable. So are reference situations as potential natural vegetation, pre-anthropogenic disturbance state as well as relaxation potential all approximated with data of (semi-) natural land use in the region (Table 1). As semi-natural land is globally considered (though a common definition does not exist) as any habitat where human induced changes can be detected or that is human managed, but which still seems a natural habitat in what species diversity and species interrelation complexity refers (thus without considerable cultivation of plants or use of fertilizers), semi-natural conditions cannot deliberately be used as a proxy for the natural (IPCC, 2002). At the same time proxies for reference states are chosen differently, probably due to poor data availability and/or lack of methodological consensus.

A step in the right direction towards standardization of reference situations might be the acknowledgment of two levels of reference situations; on a first higher level the 'theoretical' reference situation which provide the theoretical foundation for the second level a 'methodological' reference situation, which is used as the proxy. The first, theoretical level is dealing with more or less ethic, moral (or dogmatic) ideas about what nature/biodiversity "should be like". The pristine nature-ideal could be held from an ethical standpoint, regardless of whether it is a realistic scenario or not. The second, methodological level is pragmatic, not aiming at one single reference situation for nature, but acknowledging that (a) human influence will persist, (b) reference situations may be context-dependent, i.e. differ from place to place and time to time, (c) the reference should be possible to reach, and (d) data is not always sufficiently available. The 'dynamic reference situation' (Milà I Canals et al., 2007), currently based on the density of threatened and endangered species or semi-natural situations, represents this practical level of reference setting. According to their definition, reference situations are typically set differently in different ecosystems or socio-economic regions and are not represented by one single reference situation for all ecosystems.

This practical level, the pragmatic reference-setting, could be brought up to the theoretic/ethic level if we want. Although it is rarely expressed, there are ethical foundations

behind the conservation pragmatism, i.e. as used by the Convention on Biological Diversity (UN CBD, 2010); (a) we should facilitate sustainable human living from biodiversity and live in harmony with nature (b) we should enhance the benefits to all people from biodiversity and ecosystem services (c) we should preserve the Earth's biodiversity to the maximum possible extent. This does not only apply to genes which persist in pristine environments, but includes all socio-economically as well as culturally valuable species in man-made environments.

5. Assessment of reference situations

To be able to understand the full potential of the use of target references in LCA of biodiversity, further work will entail a thorough evaluation of reference situations described in this paper. For this purpose, a set of criteria was developed, based on generally accepted fundamental requirements for LCIA models (EC-JRC, 2010) as presented in the Life Cycle Data System (ILCD).

The set of criteria and recommendations developed by the ILCD against which models and indicators for use in LCIA should be evaluated, covers both required scientific qualities and aspects that influence their acceptability to stakeholders. The scientific criteria focus on the completeness of scope, environmental relevance with respect to the mechanism describing the environmental cause-effect chain, scientific robustness and certainty, applicability, documentation, transparency and reproducibility. The stakeholder acceptance criteria relate to the understandability and the link to current policy. Being a part of the LCIA model, the characteristics of the reference situation should not violate the established ILCD criteria and criteria evaluating the reference situation are thus to be developed on a symbiotic level; herewith meaning that the set of new developed criteria for the reference situation and the ILCD criteria support each other. This applies not only to the scientific criteria, but involves also a critical examination of the consistency of reference situations with current conservation strategies, as pointed out in section 4. The proposed evaluation criteria were developed with respect to the two levels of reference setting and divided in sub-criteria which are relevant for reference situations;

A reference situation at the theoretical level should:

- Be environmentally relevant; critical parts of the environmental mechanism describing the cause-effect chain of biodiversity loss should be reflected in the reference situation
- Fit within the overall LCA framework
- Satisfy stakeholder acceptance criteria
 - The reference state is easily understood and interpretable
 - The principles of the reference state are easily understood by non-experts
 - The reference state is relevant with current policy of the European Commission or similar authoritative bodies

A reference situation at the practical level should:

- Reflect the theoretical reference situation
- Be based on scientifically robust and certain assessments
 - Reference state can be verified

- Underlying data are available for use by LCA practitioners
- The reference state allows for use of different indicators of biodiversity (for marginal and average impacts as well as for different levels for biodiversity)
- Underlying data have a good potential for being consistently improved and further developed

6. Conclusion

A literature review was carried out to find out which reference situations for biodiversity have been proposed in the LCA literature and how they are applied. Many different reference situations have been proposed, most of them are so-called baseline reference situations which reflect a time period or location free from human pressures. Despite the variety of reference situations proposed, only few authors give guidance on how to apply the proposed reference situations in practice. This leads to poor approximations of reference situations and stresses the need for standardized approaches and justification of the choice of reference situations so that studies are not only transparent and relevant for decision making, but also reproducible and comparable. Based on these findings, we propose the following to foster the standardization of biodiversity reference situations in LCA:

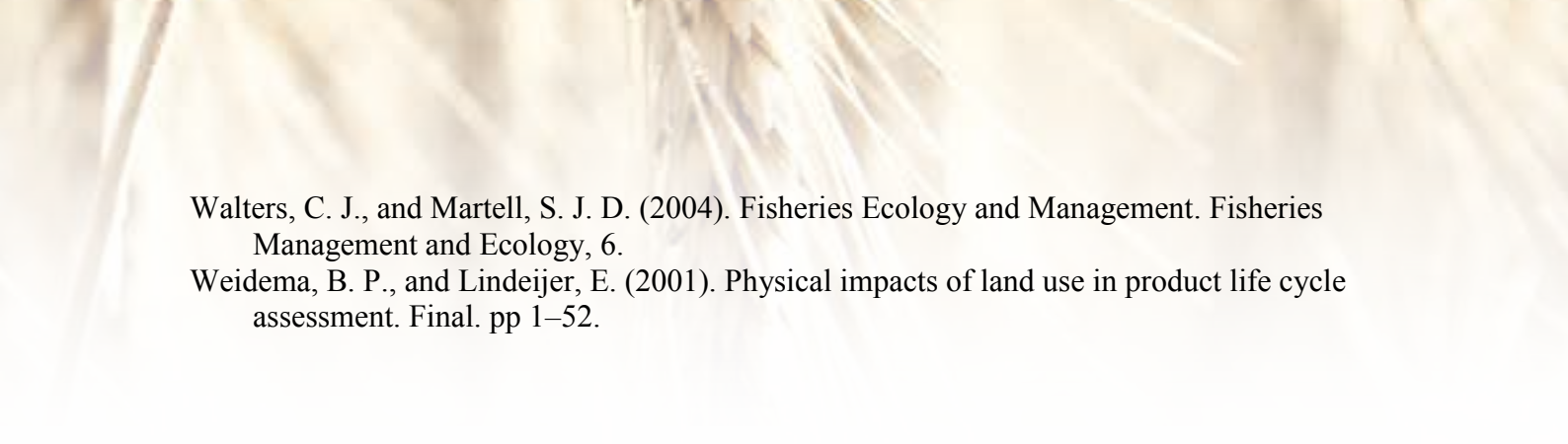
- A distinction between two levels of reference situations; the ‘theoretical’ level on the first, higher level, which provide the theoretical foundation for the second level, a ‘methodological’ reference situation, which is pragmatic and used as the proxy.
- We defined target reference situations as a promising alternative for currently applied reference situations. Further research is needed to identify the full potential and applicability of target reference situations in biodiversity assessments in LCA.
- Identification of a set of evaluation criteria for biodiversity reference situations to be used in LCA.

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186. The Need to Integrate Landscape Structure in Land Use Impact Assessment on Biodiversity: Making Transparent Trade-offs between Agricultural Production and Biodiversity Conservation

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ABSTRACT

We modeled the impact of different production intensities for several agricultural crops within different landscape contexts to analyze potential trade-offs between agricultural production and biodiversity conservation goals. The land use impact assessment method for biodiversity applied integrates cause-effect relationships between landscape structure, agricultural land use intensity, and species diversity within a landscape. A high percentage of semi-natural habitats within a landscape buffers negative impacts on biodiversity and, therefore, allows higher production intensity on the utilized agricultural area. Biodiversity in landscapes with low percentage of semi-natural habitats can be partly conserved by reducing production intensity. However, in both cases there is a competition for land and, therefore, a trade-off between agricultural production and biodiversity conservation that needs to be balanced. As this trade-off only becomes transparent when relating impacts to area, the area- and the product-related perspective need to be taken into account to adapt agricultural production intensity to the local biodiversity resources.

Keywords: farmland biodiversity, landscape management, interaction between agricultural intensity and landscape structure.

1. Introduction

As almost 40 per cent of the world's terrestrial surface are agriculturally used (Foley *et al.*, 2005) farmland plays an important role for global biodiversity and its conservation. Indeed, cultural landscapes that have been formed over long periods by human interventions may offer a wide variety of habitats which in turn generate high species diversity (Walz and Syrbe, 2013). If agricultural landscapes provide the matrix through which species can move farmland provide a high potential for biodiversity conservation (Perfecto and Vandermeer, 2008, 2010). Moreover, species diversity within agricultural landscapes are a prerequisite for ecosystem services supply (Schneiders *et al.*, 2012). The ecosystem services provided by a high biodiversity within agricultural landscapes are of importance for agricultural production itself and are seen as an important step to achieve productivity increases within sustainable farming and food systems (Maraux *et al.*, 2013; Secretariat of the Convention on Biological Diversity, 2014).

However, contemporary analyses on the global state of biodiversity estimate that drivers linked to agriculture account for 70 per cent of the projected loss of terrestrial biodiversity (Secretariat of the Convention on Biological Diversity, 2014). Various studies on the relationship between agriculture and biodiversity show that agricultural intensification is the key driver for the loss of farmland biodiversity (Benton *et al.*, 2003; Billeter *et al.*, 2008). Therefore, it is evident that impacts on biodiversity especially related to agricultural land use need to be included in LCAs of agricultural products. Including agricultural land use related impacts on biodiversity in LCA becomes even more important if LCAs are used to analyze and compare the environmental sustainability of different farming systems (Meier *et al.*, 2015). However, an important prerequisite for conclusive assessments on the environmental sustainability of different farming systems is that an LCIA method for land use related impacts on biodiversity is able to differentiate between different farming intensities.

There has been tremendous progress in the development of LCIA methods for land use related impacts on biodiversity in the past years. Besides, the UNEP/SETAC Life Cycle Initiative has initiated a process in building consensus on a shared modeling framework to highlight best-practice and guide model application for practitioners (Curran *et al.*, 2016). There are impact assessment methods available that differentiate impacts on biodiversity for different land use types (e.g. annual crops, permanent crops, pasture, forestry and urban) and of which the method proposed by Chaudhary *et al.* (2015) is the most elaborated one. However, the differentiation of farming systems on the level of land use types remains difficult. The main reason is that the spatial data on land use types used in

these methods contain no information on the matrix of structural elements within the respective land use type. However, it is the characteristics of this matrix in a landscape that in combination with agricultural land use intensity determines the number of species found within a farmed landscape (Billetter *et al.*, 2008).

Cause-effect relationships between landscape structure, land use intensity, and species diversity within a landscape served as the basis for the development of the LCIA method by Meier *et al.* (in prep.). As the matrix of semi-natural habitats within a landscape is important for the diversity of farmland species there is a potential trade-off between production and biodiversity conservation within agricultural landscapes. Here we applied the method by Meier *et al.* (in prep.) in different scenarios using inventories of several crops from the ecoinvent 3 database reflecting different production intensities to investigate the impact on species diversity and agricultural production in landscapes with different percentages of semi-natural habitats.

2. Methods

We applied the LCIA method for land use related impacts on biodiversity by Meier *et al.* (in prep.), which allows to differentiate the impact of agricultural production intensities in a specific structural landscape context. The impact assessment method is a bottom-up approach combining land use intensity (N input and crop diversity) and landscape structure parameters (percentage of semi-natural habitats within a landscape) to assess the impact on species diversity on landscape level for the taxa vascular plants, birds, spiders, carabids, syrphids, and wild bees. Landscape here is defined as a square of 4 by 4 km. The model equations were derived by regression analysis from empiric data collected from 25 agricultural landscapes in middle Europe (Herzog *et al.*, 2006; Billeter *et al.*, 2008). The data and therefore, also the regression equations derived from it are representative for the biome temperate broadleaf and mixed forests. To apply the model within another biogeographic context correction factors based on critical nitrogen loads (Clark *et al.*, 2013) were derived that account for the different vulnerability of ecosystems in another biogeographic region.

In general, species richness on landscape level of species group i is calculated as follows:

$$S_i = \alpha \times LI + \beta \times LS + i$$

S_i = species richness on landscape level of species group i

LI = land use intensity parameter

LS = landscape structure parameter

β = slope

i = intercept

The percentage of semi-natural habitats (%SNH), which comprises woods, permanent grasslands, fallow land, hedges, verges, literal zones, etc. (see Bailey *et al.* (2007) for a complete list) are integrated in the model via GIS processed satellite images from Google Earth. The impact on species diversity of a specific agricultural production intensity within a specific landscape structure is referenced to the maximum species diversity as given by the regression equations for a landscape with maximum structure and minimal land use intensity. By that the resulting number expressing the impact on biodiversity is always damage oriented. As a dimensionless index normalized between 0 and 1 it expresses the biodiversity depletion potential (BDP) on landscape level. The general formula for calculating the BDP for species group i is the following:

$$BDP_i = \frac{(S_{i_{max}} - S_{ij})}{(S_{i_{max}} - S_{i_{min}})} \times \frac{F_{LS}}{F_x} \times \frac{LI_{F_x}}{LI_{LS}}$$

BDP_i = biodiversity depletion for species group i on landscape level due to land use intensity LI_{F_x} on area F_x

$S_{i_{max}}$ = maximum species richness of group i in landscape LS

$S_{i_{min}}$ = minimum species richness of group i in landscape LS

$S_{i,j}$ = species richness of group i in landscape LS under land use intensity and landscape structure given a point in time j
 F_{LS} = landscape area
 F_x = utilized agricultural area of interest within landscape LS
 LI_{LS} = average land use intensity in landscape LS
 LI_{F_x} = land use intensity on F_x

To analyze the impact of different production intensities on species diversity on landscape level within landscapes of different percentages of semi-natural habitats we calculated the BDP per ha and per kg for the three crops barley, wheat, and potatoes as listed in Table 1. Fertilizer N-input and yields were taken from inventories out of the ecoinvent database version 3 (Weidema *et al.*, 2013) representing the production intensities organic, integrated production, and conventional (Table 1). Regarding the percentage of semi-natural habitats within the landscape we assumed three different levels: 10 %, 38 % (representing a real landscape in the Swiss Plateau), and 50 %. For each of these three levels of %SNH we assumed the same average land use intensity across the landscape and calculated the BDP per ha and per kg for each crop and production intensity. Across all landscapes irrespective of the production system it was assumed that 6 % of the agriculturally used area within the landscape consisted of permanent grassland and 94 % of crop land.

Table 1: Crop processes used within the scenarios.

Ecoinvent 3 process	Yield [kg/ha]	Fertilizer N-input [kg N /ha]
Barley grain, organic [CH]	4 153	93
Barley grain, Swiss integrated production [CH], intensive	6 828	126
Barley grain [DE]	7 500	135
Wheat grain, organic [CH]	4 069	114
Wheat grain, Swiss integrated production [CH], intensive	6 425	146
Wheat grain [DE]	7 567	173
Potatoes, organic [CH]	22 908	83
Potatoes, Swiss integrated production [CH], intensive	37 770	127

3. Results

Looking at the biodiversity depletion potential per ha, our results show that the lowest impact on biodiversity for all farming systems occurs in rich structured landscapes (i.e. %SNH = 50 %) whereas highest impacts occur in landscapes with a low percentage of semi-natural habitats (i.e. %SNH = 10 %) (Figure 1). Increasing the area of semi-natural habitats within the landscape from 10 to 50 % causes a reduction in the BDP per ha of 35 to 40 % across all crops within the same agricultural intensity (100 % basis = 10 % SNH). Reducing agricultural intensity from conventional to organic in a landscape with an area of 10 % semi-natural habitats leads to a reduction in BDP per ha by 34 to 45 % across all crops (100 % basis = conventional). In a landscape with an area of 50 % semi-natural habitats the same reduction in production intensity leads to a reduction in BDP per ha by 35 to 48 % across all crops.

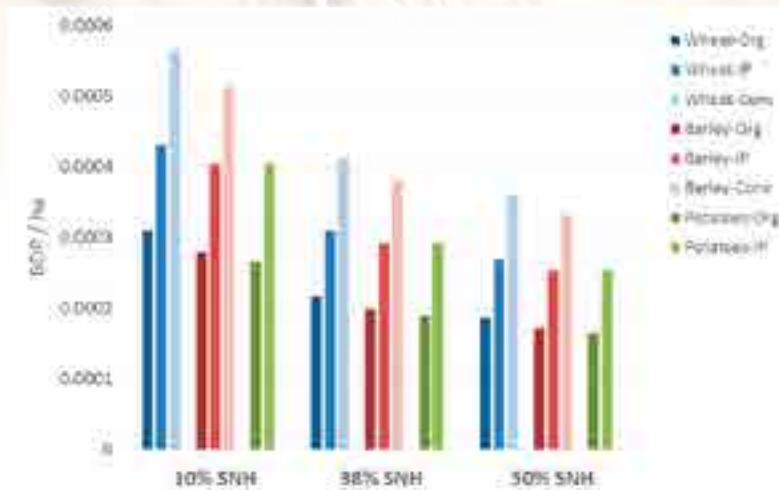


Figure 1: Biodiversity depletion potential (BDP) per ha for organic (Org), integrated (IP), and conventional (Conv) production for different crops in landscapes with a varying percentage of semi-natural habitats.

When considering the biodiversity depletion potential per kg, again, the lowest impact on biodiversity caused by a specific production intensity is found in rich structured landscapes (i.e. %SNH = 50 %) whereas highest impacts occur in landscapes with a low percentage of semi-natural habitats (i.e. %SNH = 10 %) (Figure 2). The differences in DBP per kg between the farming systems for the same crop within a landscape of the same structure are between 1.5 and 8 %. Whereas the analyzed crops produced in the organic farming system always had the lowest BDP per ha (Figure 1), in the integrated production the lowest BDP per kg was found (Figure 2). Organic farming resulted in the highest BDP per kg for potatoes in all landscapes and for wheat in the landscape with 10 % SNH (Figure 2).

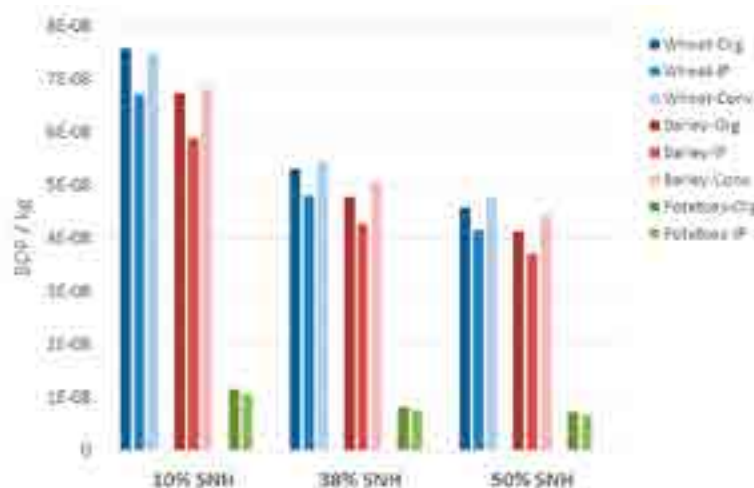


Figure 2: Biodiversity depletion potential (BDP) per kg yield for organic (Org), integrated (IP), and conventional (Conv) production for different crops in landscapes with a varying percentage of semi-natural habitats.

4. Discussion

Overall and irrespective of the farming system, land use related impacts on biodiversity due to agriculture are considerably lower in landscapes with a high percentage of semi-natural habitats (Figure 1). Obviously, semi-natural habitats are able to buffer and compensate negative impacts of agricultural land use on biodiversity to a certain degree. Figure 1 shows the reduction potential for

agricultural land use related biodiversity impacts by reducing production intensity which is higher in absolute terms in low structured than in rich structured landscapes. This model result is in accordance with empiric studies (reviewed in: Tscharrntke *et al.*, 2005; Tuck *et al.*, 2014).

The model results show that agricultural production and farmland biodiversity compete for land area in a double sense: Rich structured landscapes in principle allow for higher intensity of agricultural production until similar levels of BDP per ha are reached as in landscapes with low structure and lower production intensity. For example, in the case of wheat and barley the BDP per ha of conventional production intensity in the landscape with 50 % SNH is only 14 to 16 % higher than the BDP/ha of organic production in the landscape with 10 % SNH (Figure 1). In case of integrated production intensity in the landscape with 50 % SNH the BDP/ha is (depending on the crop) 5 to 15 % lower compared to the BDP per ha of organic production in the landscape with 10 % SNH. These relatively low impacts on biodiversity by a rather intensive agricultural production in the landscape with 50 % SNH are only achieved because enough area of compensating semi-natural habitats is available. However, this area of semi-natural habitats does not contribute directly to agricultural output. Therefore, the overall output of agricultural products from a rich structured landscape might be lower despite a high production intensity compared to the output from a landscape with low structure (i.e. small area of semi-natural habitats) and low production intensity. In contrast, biodiversity conservation goals for low structured landscapes might be achieved by reducing production intensity. However, this will also reduce the output of agricultural products from this landscape. To compensate for this reduction in yield additional land in other landscapes need to be farmed.

Interestingly, the above discussed trade-off between land for biodiversity conservation within agricultural landscapes and agricultural production becomes evident when taking an area-related perspective only. By just considering the BDP per kg to draw conclusions on biodiversity conservation there is the risk to decide for an agricultural production intensity that is actually still too intensive in a specific landscape context even though on a product related perspective impacts on biodiversity are minimal compared to production systems with lower intensity. In addition, differences in impacts on biodiversity due to different agricultural production intensities are much less pronounced in the product related than in the area related perspective.

In order to balance agricultural productivity and biodiversity conservation goals within a specific agricultural landscape production intensity needs to be adjusted to the regional biodiversity quality and quantity. Therefore, biodiversity impact assessment within LCA needs to take the regional context into account to make transparent potential trade-offs between production intensity and biodiversity conservation and allow for spatial planning. However, this requires defining a functional unit where both, biodiversity conservation and agricultural production goals, are reflected. One possibility could be to define an upper limit on biodiversity depletion tolerated per area within a specific landscape based on its vulnerability and then identify the agricultural production system with the lowest BDP per kg within the given limits.

5. Conclusions

As agricultural land use intensity and landscape structure both influence farmland biodiversity there is a trade-off between land for agricultural production and land for semi-natural habitats within agricultural landscapes. Land use impact assessment methods for biodiversity need to include the landscape matrix and the interactions with land use intensity in order to make this trade-off transparent and allow balancing agricultural production and biodiversity conservation goals. However, the trade-off between agricultural production and biodiversity conservation does not become evident by just considering the product-related perspective. The area- and the product-related perspective need to be taken into account to define on a landscape level, which agricultural production intensity can be applied in order to meet both the production and the biodiversity conservation goals. In this sense the production intensity needs to be tailored to the landscape context leading to a production intensity that is adapted to the local biodiversity resources.

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99. Combining environmental performance evaluation using LCA approach and biodiversity indicators for French dairy farms

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ABSTRACT

In order to respond to increasing societal demands for better environmental protection, stakeholders and scientists associated with cow milk production in Brittany and Massif Central French regions worked together, to assess current practices and identify improvement options for more sustainable farming practices. Fourteen dairy farms in low land Brittany (Finistere) and 15 dairy farms in mountain Massif Central (Cantal) were analysed and compared, using LCA for environmental impacts assessment and a score based tool (IBEA) for biodiversity indicators assessment. Per 1 ton of FPCM, Finistere mean impacts were higher than Cantal for eutrophication and terrestrial toxicity; Cantal mean impacts values were higher than Finistere for acidification, climate change, non-renewable energy use and land occupation. Per ha of land occupied, Finistere mean impacts values were highest than Cantal for all the categories considered, although very close for non-renewable energy use. For domestic biodiversity indicators, Finistere farms performed better than Cantal whereas for wild biodiversity, Cantal farms had highest performances. Results were presented to farmers and discussed in both groups. Priority actions were identified such as reducing input use, increasing self-sufficiency in animal concentrate feeds, increasing the variety of plant and animal species, varieties and breeds.

Keywords: Dairy farm, impact assessment, biodiversity, indicators, tools

1. Introduction

Total EU consumption of meat and dairy products is responsible for 20–50%, of aquatic ecotoxicity, acidification, eutrophication and nature occupation (EEA, 2014). In France, agriculture has got an important share in environmental impacts. Agriculture is at the forefront for CH₄ and N₂O emissions in 2013, 66% and 87% respectively, and its contribution to CO₂ equivalent was 16% of French total GHG emissions (CITEPA, 2015). In 2011, nitrate concentrations, mainly from agricultural diffuse pollution, were high (between 40 and 50 mg / l) to very high (> 50 mg / l) for 11% of French Metropolitan groundwater, both categories combined (Lacouture L., 2013). Finally, threats to biodiversity are caused by a number of factors, mainly intensive farming activities and the mass use of pesticides. At the end of 2013, 20% of all species evaluated were considered to be threatened (French Ministry of Ecology, Sustainable Development and Energy, 2014). In 2015, the hierarchy of environmental concerns of French citizens was: climate change (26%), air pollution (25%), natural disasters (18%) and water pollution (10%) (Pautard E., 2015).

This context exerts on the agricultural sector in general and dairy farmers in particular, an increasing pressure from society and authorities, asking them to adopt more environmentally friendly practices and to be aware of the impact of their practices and to adapt them. The QUALENVIC project (2013 – 2016 : <http://www.groupe-esa.com/qualenvic>) made a significant contribution to this goal by developing and using multi-criteria assessment methods and tools to identify improvement options. This project involved many stakeholders associated with cow milk production in Brittany and Massif Central French regions: dairy farmers, organisations involved in dairy farm development and technical advice to farmers, agricultural engineering schools and agronomic research. The aims of the project are to provide professionals of the dairy farming sector: (i) methodological guidance that can

lead to simplified tools in view to assessing current practices and developing scenarios for improved processes; (ii) technical references acquired on farms networks to help them adapt their practices.

2. Methods

All the stakeholders were involved in the project implementation through various committees that were set up from the beginning, including a technical committee of end users bringing together local actors of the dairy sector in each region (farmers participating in the project, technicians and professional representatives). These local committees were mobilized throughout the project, in particular for: (i) providing farm needed data, (ii) giving their opinion on methodological issues and the relevance of the results to the concerns and expectations of stakeholders, as well as (iii) facilitating deployment and adoption of the methods and tools by the profession.

LCA approach, from cradle to farm-gate, was used to assess the environmental impacts of milk production for 29 dairy farms in two highly contrasted regions: 14 mainly semi-intensive farms in lowland and intensive livestock region from western France (Brittany, department of Finistere), and 15 extensive farms in mountain and grassland area of the Center of France (Massif Central, department of Cantal). Among the Finistere farms, 3 were organic and 2 were farming system experiments set up in the experimental station of the Brittany extension service. Among the Cantal farms, 2 were organic and 2 were farming system experiments conducted at the experimental farm of the INRA Herbipôle Unit. The choice of farms was based on the diversity of production systems and practices without aiming at representativeness of the samples.

LCA calculations were performed with EDEN, a Microsoft® Excel-based tool (van der Werf et al., 2009). EDEN estimates farm emissions of CH₄, CO₂, NH₃, N₂O, NO, NO₂, NO_x, SO₂, NO₃, PO₄, Cd, Cu, Ni, Pb, Zn, non-renewable energy use and land occupation. Estimated CO₂ emissions presented in this paper do not include dynamics of soil carbon stocks. EDEN calculates potential impacts for eutrophication (EU, kg PO₄ eq.), acidification (AC, kg SO₂ eq.), climate change (CC, 100 years, kg CO₂ eq.), terrestrial toxicity (TT, kg 1,4-DCB eq.), non-renewable energy use (NRE, GJ), and land occupation (LO, m².year). EDEN distinguishes “direct” impacts originating on the farm site, from “indirect” (off-farm) impacts associated with the production and transport of inputs to the farm. Among the key inputs and farm operations considered were agricultural machinery, fuel and electricity, chemical fertilizers, fodder and concentrates feed purchased, emissions in stable and during storage of waste (manure/slurry), emissions during spreading of manure, emissions from animals during grazing period. Temporal coverage was a period of one year.

Impacts were compared using two functional units (FU): (i) 1 ton of fat and protein corrected milk sold (FPCM, Thomassen and de Boer, 2005) and (ii) on-farm plus estimated off-farm hectares utilised (Salou et al., 2016). For the functional unit 1 ton of FPCM sold, total sales data for milk and livestock are used to perform an economic allocation, which estimates the proportion of total impacts due to each of these two animal products. Processes associated with sold crop products were excluded from the system because they were not used for animal production (van der Werf et al., 2009).

Non-agricultural parts of the farm such as farm buildings and woodlands are not included in the system, nor are chemicals and veterinary products. For pesticides, non-renewable energy use and impacts associated with production and supply are considered, but impacts associated with the use of pesticides are not considered, due to lack of appropriate characterisation factors. EDEN uses data readily available on-farm, and thus can be performed to assess a large number of farms at a reasonable cost.

EDEN was developed within the project “Evaluation of the sustainability of dairy farming systems in Bretagne”, carried out from 2003 to 2006 by Agrotransfert Bretagne, a structure which formalised collaborative projects between the French National Institute for Agronomic Research (INRA) and the Bretagne Chamber of Agriculture (CAB) (van der Werf et al., 2009).

One of the objectives of QUALENVIC project was to update and improve EDEN, which was achieved, among other things, by incorporating a biophysical allocation method based on the metabolic energy required to produce each co-product (Koch P., Salou T. 2013), beside the original economic allocation.

We assessed biodiversity indicators on the same dairy farms samples. As no clear consensus exists yet on the use of specific impact indicator(s) to quantify land use impacts on biodiversity within LCA (Milà i Canals L. et al., 2014), we used a score based tool diagnosing the impact of agricultural practices on domestic and wild biodiversity. IBEA (Impact des Pratiques Agricoles sur la Biodiversité, designed by France Nature Environnement (FNE), INRA, Museum of Natural History and others, 2011) was chosen, because it is a synthetic tool suitable for both specialized dairy farms and mixed farming, at the whole farm scale. It consists of 33 generic indicators addressing most of the components of biodiversity. IBEA uses data readily available on-farm. No field measures or species counting are needed. It therefore allows a rapid diagnosis, which takes about an hour for data collection on the farm. IBEA does not address the species level but only biodiversity in general through an “environment quality” and a “genetic diversity” components (Bockstaller C. et al., 2011).

The assessment of wild and domestic biodiversity is estimated by an impact class from 1 to 5: very negative impact, negative, neutral, positive or very positive. Domestic biodiversity is evaluated by counting the number of productions on the farm, the number of species and varieties / breeds of plants and animals. For wild biodiversity, IBEA assesses impacts of farmer's practices on quality habitats distinguishing cultivated, semi-natural and natural areas. It assesses the variety of resources available on the farm for wildlife but also their spatial and temporal continuity (concept of ecological corridor).

In order to better clarify the impact of grazing and mowing on biodiversity, adjustments to the IBEA questionnaire were made specifically for Qualenvic project. We also opted for a lesser weight to the quality of woodland managed by the farmer in the note of natural environments quality, as we consider this criterion as less relevant to the objective of the project.

3. Results

Characteristics of dairy farms

Examined dairy farms differed with respect to mean values for farm structure, input use and output level (Table 1). Relative to Finistere farms, Cantal farms had a smaller usable agricultural area (UAA), smaller stocking density and smaller pasture residence time. Cantal farms had larger mean value for the percentage of fodder crops and grasslands in UAA. The differences in this category of farm structure and management were weak, between 10 and 15%, except for stocking density with 29% difference. There were more marked differences in input category, between 52% and 90%: Cantal farms had much larger mean value for purchased concentrate feed use per kg of FPCM, whereas Finistere farms had much higher mean value for total N input, as well as for diesel, electricity and pesticide use. Differences were also very high for the surplus of N and P farm-gate balance in the output category, where Finistere farms had much higher mean values than Cantal farms, 54% and 93% respectively. Finistere farms had also higher milk production per cow (22%). For milk fat and protein content, both had similar mean values. The proportion of milk sales in total farm animal product sales was slightly higher for Cantal farms.

Table 1: Mean values for characteristics of dairy farms (cash crops excluded) in Brittany (Finistere, n = 14), and Massif Central (Cantal, n = 15).

Characteristic	Dimension	Finistere	Cantal
Farm structure and management			
Useable Agricultural Area (UAA)	ha	80	68
Fodder Crops and Grass (FCG) in UAA	%	83	92
Stocking density	LU ha ⁻¹ FCG	1.4	1.0
Pasture residence time	days year ⁻¹	211	188
Inputs			
Purchased concentrate feed use	g kg FPCM ⁻¹	58	147
Total N input	kg ha ⁻¹ UAA yr ⁻¹	182	71
Diesel use	kg ha ⁻¹ UAA yr ⁻¹	110	53
Electricity use	kWh ha ⁻¹ UAA yr ⁻¹	672	286
Pesticide use (active ingredients)	g ha ⁻¹ UAA yr ⁻¹	223	22
Output			
Milk production	kg FPCM cow ⁻¹ yr ⁻¹	7005	5468
Milk fat content	%	4.15	4.07
Milk protein content	%	3.29	3.36
Milk-sales portion of total sales	%	80	86
Surplus of N farm-gate balance	kg ha ⁻¹ UAA yr ⁻¹	98	45
Surplus of P farm-gate balance	kg ha ⁻¹ UAA yr ⁻¹	40	2.7

Impacts of dairy farms

When expressed per 1 ton of FPCM, Finistere farms mean impacts were 14% higher than Cantal for EU and 58% higher for TT; Cantal farms mean values were higher than Finistere farms for the following impacts: AC (26%), CC (17%), NRE (39%) and LO (44%) (Table 2).

When expressed per ha of land occupied, Finistere farms mean impacts values were highest than Cantal farms for all the categories considered: 49% higher for EU, 26% for AC, 34% for CC, and 77% for TT and 3% higher for NRE (Table 2).

Table 2: Mean impacts (1) per 1 ton fat and protein corrected milk (FPCM) and (2) per ha of land occupied for dairy farms in Brittany (Finistere, n = 14), and Massif Central (Cantal, n = 15). Values in brackets are standard deviations.

Potential impact	Units	Per 1000 kg FCPM		Per ha of land occupied	
		Finistère	Cantal	Finistère	Cantal
Eutrophication (EU)	kg-eq. PO ₄	7.9 (2.6)	6.9 (1.3)	47 (13)	24 (8)
Acidification (AC)	kg-eq. SO ₂	7.9 (1.5)	10.8 (1.6)	50 (16)	37 (11)
Climate change (CC: 100 yr)	kg-eq. CO ₂	1090 (148)	1308 (178)	6824 (887)	4516 (1234)
Terrestrial toxicity (TT)	kg-eq.1,4-DCB	1.8 (1.7)	0.8 (0.6)	12 (12)	2.9 (2.3)
Non ren. energy use (NRE)	GJ	2.9 (1,0)	4.7 (1.6)	17.1(6.7)	16.7 (7.9)
Land occupation (LO)	m ² yr ⁻¹	1756 (643)	3144 (1049)		

Biodiversity assessment

Finistere farms performed much better than Cantal farms for domestic biodiversity indicators (Table 3). Conversely, for wild biodiversity indicators, Cantal farms had globally highest score than Finistere farms. Cantal farms mean values were highest for the proportion of natural and semi-natural areas on the farm, the quality of the cultivated areas (when they exist), the quality of the natural, semi natural and forests present on the farm and the preservation of key habitats (Table 3). For the other criteria, the performances were similar between the two sets of farms (diversity of resources offered, connectivity between natural and semi-natural areas, overtime continuity of plant cover).

Table 3: Mean values for domestic and wild biodiversity indicators for dairy farms in Brittany (Finistere, n = 14), and Massif Central (Cantal, n = 15).

Indicator	Finistere	Cantal
Domestic biodiversity		
Diversity of productions	5	2
Wild biodiversity		
Proportion of natural and semi-natural areas on the farm	3	4
Quality of the cultivated areas of the farm	3	4
Quality of other areas of the farm	3	4
Preservation of key habitats of the farm	1	2
Diversity of resources offered by the farm	2	2
Connectivity between natural and semi-natural areas	3	3
Overtime continuity of plant cover	3	3

4. Discussion

The discrepancies in the results for both the environmental impacts and biodiversity indicators can largely be explained by differences found in the main characteristics of the two dairy farm samples, especially those related to farm structure and management, input use and output (Table 1).

Cantal dairy farms, mainly with 100% permanent grassland do not produce feed for their livestock; hence their higher level in use of purchased concentrate feed compared to Finistere farms. This lack of autonomy in concentrate feeds entails procurement and transportation over long distances, which largely explain their highest level in non-renewable energy use and associated impacts when expressed per 1 ton of FPCM. More autonomy in concentrate feeds for Finistere farms has its own inconveniences such as pesticides use, chemical nitrogen and phosphate fertilizers use, which yields much higher level in surplus of N and P farm-gate balance. The terrestrial toxicity potential (mainly due to heavy metal accumulation in soil) expressed per ha of land, is much higher for Finistere farms due to the spreading of pig slurry brought from outside for most of the farms. This factor also explains much of the gap with Cantal farms for potential eutrophication impact. Our results also allow highlighting some ambiguity to analyze the data expressed per 1 ton of FPCM or per ha of land. We have demonstrated indeed that the classification differs widely between our two sets of farms according the used unit. Doing this, we follow Salou et al. (2016) that recommend the use of both mass-based and area-based functional units in life cycle assessments of agricultural goods in order to give a more balanced view of the impacts (including climate change) of the systems. To compare such contrasting production systems, it would also have been interesting to use alternative functional units such as energy content and protein content.

Regarding the domestic biodiversity indicators, Finistere dairy farms obtain a very high score because they are mixed farming systems that combine crops such as cereals and legumes intercrops, temporary grasslands and permanent grasslands. They managed thus a diversity of plant species and a wide range of varieties for the same species especially within the temporary grasslands (genetic diversity). In addition, some of those farms have got a certain diversity of breeds and animal species

(cows and a few pigs). By contrast, dairy farms in Cantal have a low value for domestic diversity. They are indeed specialized system with 100% of permanent grasslands and only a few farms own Cantal heritage breeds. Conversely, when we examine this data from the wild biodiversity point of view, Cantal farms have high performances because they manage large areas of semi-natural grasslands associated with unproductive natural habitats (hedges, woodlands, wetlands...), and contribute to the preservation of remarkable habitats. Furthermore, farmers implement practices using minimum inputs on their cultivated and grasslands areas, have small animal stocking density, and use late mowing management.

5. Conclusions

This study allowed us to update and improve a tool specifically designed for life cycle assessment of milk production (EDEN). It was implemented on 29 contrasted milk production systems in lowland and mountain areas. To take into account the multi-functionality of dairy farms, we also evaluated the impact of agricultural practices on biodiversity using IBEA tool, at a minimum extra cost because a large part of the data collected served for both assessments (LCA and biodiversity). Results were presented to each farmer participating in the project as a "4 pages" showing the potential environmental impacts and biodiversity indicators of the farm. Analysis of the results showed the hotspots at the farm and at the group level. Priority actions to ensure more sustainable farming practices were identified such as reducing input use, increasing self-sufficiency in animal concentrate feeds, increasing the variety of plant and animal species, varieties and breeds.

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83. Livestock Environmental Assessment and Performance (LEAP) Partnership: Building global methodological consensus for improved environmental management

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⁸ European Commission, DG Environment

⁹ Unione Nazionale Industria Conciaria

¹⁰ World Alliance of Mobile Indigenous Peoples

¹¹ Government of New Zealand

¹² International Organization for Standardization

¹³ Instituto Nacional de Carnes, International Meat Secretariat

¹⁴ Government of France

¹⁵ Beef and Lamb New Zealand, International Meat Secretariat

¹⁶ International Meat Secretariat

¹⁷ International Dairy Federation

¹⁸ Government of Switzerland

¹⁹ United Nations Environment Programme

²⁰ International Egg Commission

²¹ International Union for Conservation of Nature

²² European Feed Manufacturers' Federation, International Feed Industry Federation

²³ World Wide Fund for Nature

²⁴ Government of Ireland

²⁵ University of California, Davis

²⁶ International Poultry Council

²⁷ International Planning Committee for World Food Sovereignty

²⁸ Government of Uruguay

²⁹ International Council of Tanners

³⁰ Government of the Netherlands

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³³ Teagasc, Government of Ireland

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ABSTRACT

Research is evolving fast in environmental assessment. Accordingly, Life Cycle Assessment studies often deliver contradictory messages when it comes to agriculture and livestock supply chains. High is hence the risk to mislead policy making, and to create unjustified market distortions. While science continues to evolve, FAO, governments, private sector, non-government organizations and civil society organizations engaged in the Livestock Environmental Assessment and Performance (LEAP) Partnership to build global methodological consensus for sound environmental assessments of livestock supply chains. LEAP guidelines are developed by Technical Advisory Groups composed of scholars and technical experts from a wide array of regions. LEAP guidelines are designed to support the environmental benchmarking of livestock supply chains without focusing on a specific production practice. This paper provides an overview of the major achievements of the LEAP Partnership phase 1 (2012-2015) as well as presents the technical challenges that LEAP will tackle in its phase 2 until 2018. Special emphasis is on the ongoing technical activities on water footprinting, modelling of nitrogen and phosphorus flows, soil carbon stock changes, and road testing of LEAP1 guidelines.

Keywords: eco-efficiency; consensus building; GHG emissions; water footprinting; nutrients modelling; carbon stock changes; biodiversity; road testing; livestock

1. Introduction

Population growth, income gains and urbanization are translating into increasing demand for livestock products, particularly in developing countries. While the livestock sector makes an important contribution to global food supply and economic development, it also uses significant amounts of natural resources and can lead to significant environmental impacts. With global resource scarcity and environmental degradation presenting growth challenges for the sector, along with related market and regulatory pressures, the sector is facing a need to think more strategically about the sustainability of production (Gerber et al., 2013).

As the question of the sustainability of future food system takes root, there is a growing recognition of the need for comparative and standardized methods and indicators to assess the sector's environmental performance and progress towards sustainability. These methods and indicators are required not only to evaluate the environmental performance but also to identify areas where benefits are greatest as well as provide important information for the design of more efficient processes, improving resource use and environmental impacts.

To support this process, the Livestock and Environmental Assessment and Performance (LEAP) Partnership, which is a multi-stakeholder initiative involving governments, private sector, academia, non-government organizations, civil societies, and other organizations around the world, was launched in 2012. LEAP Partnership aims to build credible and robust accounting methods and indicators that serve as a foundation to address the sustainability challenges faced by livestock supply chains. The Partnership's leading goal is to improve the environmental performance of the livestock sector, while considering economic and social viability.

Among the assessment tools currently available, LEAP relies on, but is not restricted to, the Life Cycle Assessment (LCA) framework. Since the LCA framework was conceived and developed for industrial processes, several methodological challenges need to be addressed to ensure accurate and transparent application to food systems. Since its inception, LEAP Partnership has built technical consensus on data and guidelines on livestock supply chains, here referred as LEAP products.

LEAP products are essential tools to support the design of effective policies and improvement interventions that can contribute to the achievement of the UN's Sustainability Development Goals (SDGs) such as SDG2: "End hunger, achieve food security and improved nutrition and promote sustainable agriculture".

This paper provides an overview of the major achievements of the LEAP Partnership phase 1 (2012-2015) as well as presents the technical challenges that LEAP will tackle in its phase 2 until 2018. Special emphasis is on the on-going technical activities on water footprinting, modelling of nitrogen and phosphorus flows, soil carbon stock changes, and road testing of LEAP1 guidelines.

2. LEAP Phase 1: results

To pursue the leading goal of the Partnership, LEAP has initially shaped a 3 year work programme, called LEAP1, which ended in December 2015. LEAP1 consisted of few components ranging from the development of databases on GHG emissions to consensus building on LCA guidelines to account for the environmental impacts of livestock supply chains by taking into account the wide diversity of livestock production systems as well as natural resources availability around the globe. Six Technical Advisory Groups (TAGs) were formed and more than three hundred experts from all over the world have been involved in the development of the technical guidance documents. These TAGs developed (a) 5 technical guidance documents focusing on the feed and livestock supply chains, (b) 2 technical documents related to the assessment of the impacts of livestock on biodiversity, and (c) methodological notes. To ensure soundness, LEAP products go through a multi-step review process. This include an internal review by the LEAP Secretariat and the Steering Committee, followed by an external review from a group of experts and a public review. A short description of these documents is provided below.

1. The technical document “*Environmental performance of animal feeds supply chains: Guidelines for assessment*” by the LEAP Feed TAG focuses on the accounting of GHG emissions and fossil energy use and associated environmental impacts including climate change and fossil resources depletion. Moreover, this document provides with additional guidance to account for other impacts such as acidification, eutrophication, and land occupation (FAO, 2016b).
2. The technical document “*Greenhouse gas emissions and fossil energy use from Poultry supply chains: Guidelines for assessment*” by the LEAP Poultry TAG focuses on different poultry species such as chicken, turkeys, guinea fowl, geese, quails, ducks, and pigeons and provides guidance for environmental accounting of GHG emissions and fossil energy use as well as associated impacts of climate change and fossil resources depletion (FAO, 2016c).
3. The technical document “*Greenhouse gas emissions and fossil energy use from Small Ruminant supply chains: Guidelines for assessment*” by the LEAP Small Ruminant TAG focuses on accounting of GHG emissions and fossil energy use as well as associated impacts of climate change and fossil resources depletion for goats and sheep supply chains (FAO, 2016d).
4. The technical document “*Environmental Performance of Large Ruminant Supply Chains: Guidelines for assessment*” by the LEAP Large Ruminant TAG focuses on the accounting of GHG emissions and fossil energy use and covers several impact categories such as climate change, fossil resources depletion, water footprint, acidification, eutrophication, land occupation, and biodiversity (FAO, 2016g).

5. The technical document “*Environmental Performance of Pig Supply Chains: Guidelines for assessment*” by the LEAP Pig TAG focuses on the environmental accounting of GHG emissions and fossil energy use. Moreover, it provides guidance on the impact categories such as climate change, fossil resources depletion, water footprint, acidification, eutrophication, land occupation, and biodiversity (FAO, 2016h).
6. The *LEAP Database on Feed Crops*, which was developed by an ad hoc task force, provides detailed information on GHG emissions of the five major crops including barley, cassava, maize, soybeans and wheat. The LEAP database delivers results at a regional and country level. Parameters such as production system (i.e. rain fed, irrigated), production practice (i.e. no tillage, minimal tillage, conventional) can be set and results are delivered accordingly in CSV format. For transparency, the default values of the database model are documented (FAO 2015a,b,c).
7. The documents “*Principles for the assessment of livestock impact on biodiversity*” and “*A review of indicators and methods to assess biodiversity*” by the LEAP Biodiversity TAG are an important step to adequately capture impacts on biodiversity in environmental assessments of livestock supply chains. These documents describe different methods and approaches that are relevant to a variety of stakeholders. The principles were shaped to guarantee a minimum level of soundness, transparency, scientific relevance, and completeness (FAO, 2016e,f).
8. The *LEAP methodological notes* summarize the approach used to reach technical consensus, point out the principal features and limitations of LEAP1 guidelines, and includes a commentary on the complementarity of LCA data modelling approaches in different application contexts (FAO, 2016a).

3. LEAP Phase 2: on-going activities

The LEAP Partnership work programme 2016-2018, also known as LEAP+, aims at maximizing the value and application of LEAP1, but it also broadens its scope with the inclusion of additional environmental issues. Road testing is a necessary step to evaluate to what extent the technical documents produced are clear and provide sufficient guidance for application at various scales (field, farm, country, region and globe). Building on the results of the testing as well as on the additional outcomes of LEAP+, LEAP guidelines will be revised and consolidated.

Phase 1 of the LEAP work programme has largely focused on the harmonization of methodologies for the quantification of greenhouse gas emissions from livestock supply chains. However, measurement of GHG emissions are only partial metrics, and can lead to misleading policy signals if not placed within the proper context of the wider relationship between livestock and the environment. Part of the mandate of the LEAP Partnership is to develop methods and indicators that can be applied to measure the wider environmental performance of livestock. LEAP1 initiated some of this work through the development of biodiversity principles as well as the work initiated on nutrient use efficiency in livestock supply chains. In 2016 three more TAGs have been launched to discuss on critical topics including Nutrient Cycles Accounting and Impact Assessment, Water Footprinting and Soil Carbon Stock Changes.

3.1 Nutrient Cycles Accounting and Impact Assessment

Environmental assessment of livestock supply chains are challenged by methodological issues related to the accounting of nutrients such as e.g. nitrogen (N) and phosphorus (P). Nutrients either mineral or organic are valuable resources used in agricultural systems to produce feed, food, biofuel and timber. However, nutrients use can result in environmental pollution of water, soil and air. Depending on the environmental assessment framework concerned (e.g. LCA, resource-use efficiency metrics), pollution can be expressed in terms of potential impact on climate change, acidification, eutrophication, degradation of water quality or air quality, or in terms of nutrient pressure (losses or surpluses) per unit of land, product, farm or system. While industrially producing N can be energy intensive (Galloway et al., 2003), P is a scarce resource, which, in the form of phosphate rock, is classified as critical raw material by some countries (Cordell et al., 2009). Several models for nutrient accounting and impact assessment methods exist, and assessment results are often questioned.

For example, many nutrients accounting models do not take into account, amongst others, both regional and temporal variability aspects and changes in soil nutrient stocks. Such accounting models seemingly do not distinguish animal categories and require additional accounting layers when it comes to assessment of e.g. integrated production systems (e.g. crop-livestock systems, intra-annual crop rotation, silvo-pastoral systems). Impact assessment methods in LCA often lack acceptance from stakeholders because the “pressure-to-impact” framework lack details and local data. As result, LCAs of livestock products are seemingly struggling to deliver plausible results, which are representative for the geographical areas concerned in the assessments. In addition, nitrogen footprinting approaches and other indicators to evaluate nutrient use efficiency over the life-cycle of livestock products are being developed and fine-tuned. All these assessment frameworks are emerging as addition to the nutrients budget, which are already widely used in many countries in support of both environmental management and monitoring agricultural policies.

LEAP members called for sound recommendations on nutrient accounting and impact assessment for inclusion into the LEAP guidelines. To achieve global consensus in the field, a TAG was formed to build common ground by facilitating technical dialogue between the relevant scientific communities, practitioners, and LEAP stakeholders.

Some of the questions that the TAG has been answering include the following:

- What are the key features of the different nutrients assessment frameworks? How and to what extent do they differ from each other? What are their key strengths and limitations? Which contexts these assessment frameworks are normally applied in? Do the different frameworks compete in application or can complement each other bringing additional, useful information for informed decision making?
- Which assessment frameworks are suited for benchmarking the environmental performance of livestock supply chains? Which indicators are relevant for assessments conducted at regional or global scale?
- Which accounting method is the most suited to estimate the amount of nutrient losses into the environment?
- How to account for soil nutrients stock changes? How to deal with nutrients carried over from previous productions? How to take into account biological N fixation? How to frame the accounting systems when land use change took place? How to allocate emissions from up-stream activities (e.g. mineral fertilizer production) to integrated crop-livestock production systems or crop rotation?
- What criteria should be considered to characterize and evaluate the scientific soundness of life cycle impact assessment methods for eutrophication and

acidification? To what extent the “pressure-to-impact” models can be tailored to take into account local geographical conditions?

- Which indicators are proposed in the literature to assess phosphorus as critical resource?

3.2 Water Footprinting

Water is an essential production input for feed and livestock supply chains. In several geographical areas water is an increasingly scarce resource, whose availability varies widely over temporal and spatial scales. In addition, other challenges such as climate change and increasing competition with other users (e.g. other agriculture sectors, household, industry, tourism, etc.) is exacerbating water scarcity. Efficient management of this resource is essential to ensure food security and viability of livestock supply chains and better future for next generations.

The Water Footprint Network (WFN) spearheaded the development of water footprint indicators. However, the assessment framework introduced by the WFN has often been questioned in scientific literature and alternative approaches have been proposed. Recent progress has been made for instance through the development of the ISO 14046:2014 that highlights the principles and life cycle approach for the calculation of product water footprints. In order to complement the ISO assessment framework with blue water assessment methods, the UNEP SETAC Life Cycle Initiative (WULCA project) has developed a set of blue water footprint indicator(s) and related characterization factors. At the sectoral level, the International Dairy Federation (IDF) recently announced release of its guidelines on water footprinting of dairy systems. The reduction of the amount of water use per unit of animal product can reduce the pressure of current practices on this scarce resource especially in the area where water stress indexes are higher. Therefore, the development of clear guidelines on water footprinting can support water management solutions through the identification of hotspot of water use in livestock supply chains.

LEAP members called for sound recommendations on water footprinting that adequately capture the specificities of livestock production systems. Building on existing standards and methods, the Water TAG has been building global consensus on water footprinting of livestock supply chains. This is deemed necessary to build confidence in the assessment results that water footprinting studies deliver and to expand the scope of existing LEAP guidelines.

Some of the questions the Water TAG has been answering include the following:

- Which water footprinting approaches and impact assessment methods are currently recommended for applications at various scales and in different application contexts (e.g. environmental benchmarking at the level of farms, regions and countries; product water footprinting; environmental assessments of technology alternatives)? What are their key features, application contexts, strengths and limitations?
- How plausible water footprints look alike when the mainstream approaches and methods are applied for assessments of livestock supply chains?
- Which accounting rules and models shall be incorporated into the LEAP guidelines in order to obtain fair results that also reflect performance in water use efficiency? What livestock water requirements can be recommended to estimate water flow volumes whenever measurements are either not taken or cannot be taken?

- How to capture soil water retention changes due to e.g. deposition of manure and soil compaction by livestock?

3.3. Soil Carbon Stock Changes

Livestock have a significant contribution to anthropogenic GHG emissions and measuring these emissions in a harmonized way is a first key step towards mitigation. Soil carbon storage in grassland offers an important potential to compensate GHG emissions from livestock; however, the lack of consensus on methodologies to account for soil carbon stock changes hinders fair assessments and evidence-based policy dialogue. Within LEAP, a technical advisory group of experts has been recently formed to build this consensus. As an initial step, LEAP conducted a scoping analysis to prepare the work of the TAG on soil carbon stock changes. In particular, the scoping analysis was conducted to:

- Review the main approaches for modelling soil carbon stock changes. Special emphasis will be on grasslands and rangelands.
- Identify and describe those contentious issues in modelling approaches (with specific emphasis on LCA and national GHG emission accounting frameworks) where consensus is to be built.
- Come up with the technical boundaries of the TAG regarding, but not restricted to:
 - The definition of grasslands (e.g. whether to include savannahs, cut grasslands) and rangelands;
 - Whether to include other land use types, e.g. cropland, which are relevant for feed;
 - How to define a reference state for land use and land use change;
 - Whether and how to consider carbon stock changes during land occupation;
 - How to consider carbon stock changes after a land use transformation, e.g. forest to grassland, grassland to cropland;
 - Existing sources of data for soil carbon storage and which model they follow;
 - The type of biomass: below ground only (and until what depth) or above ground as well;
 - Where to cut off the carbon cycle;
 - Whether implications on soil fertility or quality should be addressed;
 - An overview of the main research groups and international task forces working on soil carbon storage and related issues.

The tasks of the Soil Carbon TAG are being synchronized with on-going activities of the Global Soil Partnership (GSP) in view of the revision of IPCC guidelines.

3.4. LEAP Public Review 2016 and Road Testing of LEAP1 guidelines

Feedback on the following was collected in the LEAP public review that lasted from April to August 2016: (a) *Environmental performance of pig supply chains: Draft guidelines for assessment*, and (b) Draft Replies to selected comments from *Cowspiracy*, a documentary that is playing a major role in the public debate on the livestock sector sustainability.

Regarding road testing of LEAP1 guidelines, it has started in 2016 and will continue over the next years:

- To evaluate the suitability of LEAP1 guidelines for tracking environmental performance improvement in different (policy) application contexts
- To evaluate the comprehensiveness of LEAP1 guidelines as well as to what extent they fit for all feed and livestock production systems in different geographical areas
- To get feedback on the clarity of recommendations
- To spot differences with competing, prominent standards and guidelines, and pave the way for alignment and adoption
- To identify gaps in recommendations and any barriers preventing application and endorsement
- To shape the agenda of the future meetings of LEAP TAGs envisaged to consolidate LEAP guidelines.

4. Conclusions and perspectives

To fully achieve the general objective of the Partnership, LEAP+ will address several issues in order to make fair and sound environmental assessment of livestock supply chains. While in 2016 special focus is devoted to nutrients modelling and water footprinting, the work on soil carbon stock changes will continue over 2017. Amongst others, the LCA scientific community will be asked to provide feedback on the LEAP 2016 deliverables. A second call for road testing of LEAP1 guidelines will be launched soon. Over 2017-2018, LEAP+ technical activities will also include the consolidation of the guidelines delivered by LEAP1; the development of guidelines on feed additives or on change-oriented assessment approaches, the principles and guidance on chemical dispersion models, in relation to the use of chemicals for feed production, the guidance on biodiversity indicators as well as the guidelines for integrated sustainability assessments.

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16. New characterization factors based on direct measures of plant species richness in Europe to estimate land use impacts on biodiversity in LCA

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Introduction and objective of the work

Biodiversity are seldom included in life cycle assessments (LCA) due to methodological limitations. When assessing organic agricultural products the omission of biodiversity in LCA is problematic, since organic systems are characterized by a higher species richness compared to conventional systems. The aim of the study was to estimate characterization factors for land use impacts on plant species richness in the ‘Temperate Deciduous Forest’ biome, that are able to distinguish between different land use types and between organic and conventional farming practices.

Methodology

The new characterization factors are based on a unique dataset derived from field recording of plant species diversity in farmland across six European countries in the ‘Temperate Deciduous Forest’ biome from the BioBio project (<http://www.biobio-indicator.org/>). Based on the number of plant species recorded in conventional and organic arable land and pastures, the plant species richness and characterization factors expressing the Potentially Disappeared Fraction (PDF) were calculated using the same method as De Schryver et al. (2010).

Results

The calculated characterization factors clearly distinguished the different land use types of pastures (monocotyledons or mixed), arable land and hedgerows and were also able to distinguish management practices (organic or conventional) (Fig. 1)

Implications and conclusions

The new characterization factors can be used to supplement or validate current CFs or they can be applied directly in LCAs of agricultural products to assess land use impacts on species richness in the biome ‘Temperate Deciduous Forest’, that represent a major part of the North, West and Central European agricultural land.

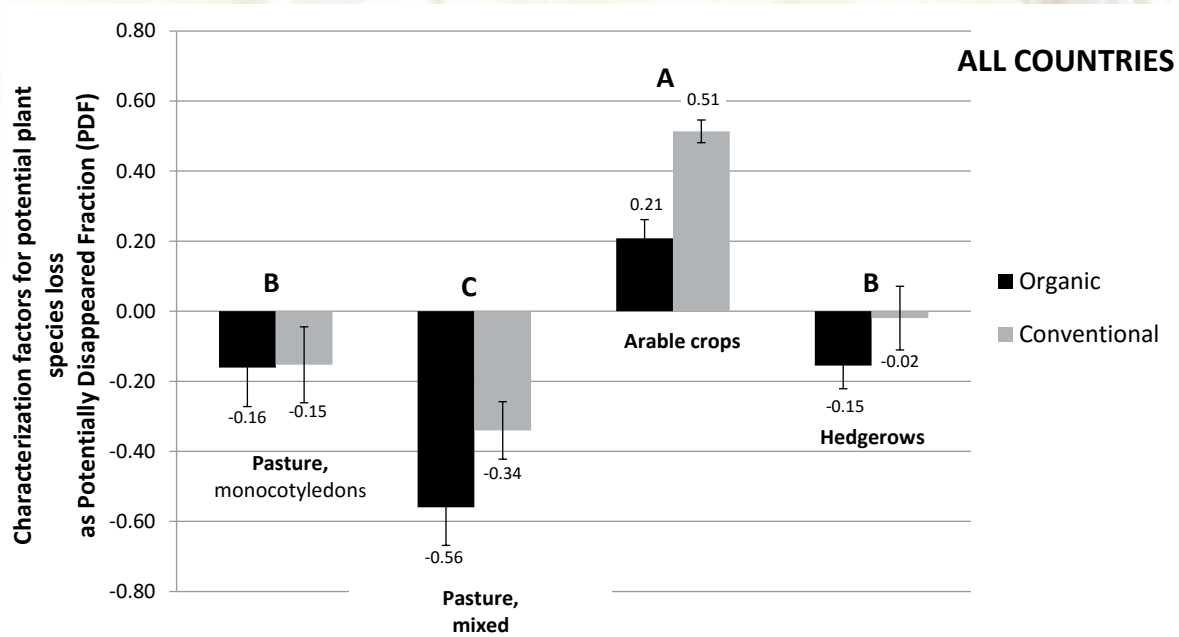


Figure 1. LCA biodiversity characterization factors for the different land use types and management practices. The uppercase letters represent significant differences between the four overall land use types ('Pasture, monocotyledons', 'Pasture, mixed', 'Arable crops' and 'Hedgerows').

Reference

De Schryver, A.M., Goedkoop, M.J., Leuven, R.S.E.W., Huijbregts, M.A.J., 2010. Uncertainties in the application of the species area relationship for characterisation factors of land occupation in life cycle assessment. *Int. J. Life Cycle Assess.*, 15, 682–691.

ORAL PAPERS

SEAFOOD AND AQUACULTURE

10. Seafood and Aquaculture

7. Challenges and best practices of seafood supply chains LCA

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ABSTRACT

Due to the rapid growth of the seafood sector, it is relevant to study the environmental and ecological impacts associated with current and future seafood supply chains of aquatic products aimed for direct or indirect human consumption, including fish, molluscs, crustaceans and algae. Despite efforts, certain challenges remain in seafood LCA. Based on these concerns, this work suggests best practices (including recent methodological developments and novel methods) addressing challenges for the application of LCA to study seafood supply chains, and promoting more holistic and robust outcomes. A literature review was performed, targeting recent reviews, methodological papers, case studies and guidelines for LCAs of seafood-based supply chains. Best practices were identified based on their capacity to complete, complement and support the interpretation of LCAs, their practical demonstration, and our expert judgement. The adoption of these best practices (which address the inclusion of fisheries management concerns, goal and scope decisions, and data availability and management) guarantees solid LCA studies with adequate data and uncertainty management, inclusion of seafood-specific impact categories, and coherent study design.

Keywords: fisheries, aquaculture, fish processing, fish supply chains, life cycle assessment

1 Introduction

Given the increasing global demand for fish and other aquatic products for human and animal consumption, and the fact that wild caught fisheries —supplying inputs for the food and feed industries— have stagnated over the past 15 years (FAO 2014), it is highly relevant to study the environmental and ecological impacts associated with current and future seafood supply chains. These seafood supply chains encompass aquatic products aimed for direct or indirect human consumption, including fish, molluscs, crustaceans and algae. The production of aquatic biomass for non-food uses, such as microalgae as feedstock for biodiesel production, could also be considered as part of these supply chains, because they share some common production processes and thus exert similar pressures on the environment. Topics addressed by seafood supply chain research include harvesting practices, processing, life cycle assessment (LCA), eco-efficiency, waste management, distribution and consumption, total energy costs, and conservation of resources and biodiversity (Ayer et al. 2009).

An outstanding number of studies have, to date, applied LCA to seafood production systems, many of which have applied novel impact categories and tools designed to account for the ecological impacts of removing biomass from ecosystems, disrupting benthic ecosystems, etc. Despite these efforts, certain challenges remain in seafood LCA. Based on these concerns, this work presents best practices (including recent methodological developments and novel methods) addressing challenges for the

application of LCA to study seafood supply chains, and promoting more holistic and robust outcomes.

2 Methods

A documentary review was performed, targeting recent reviews (Avadí and Fréon 2013; Henriksson et al. 2012; Vázquez-Rowe et al. 2012), methodological papers (Avadí and Fréon 2015; Ayer et al. 2007; Patrik Henriksson et al. 2015; Vázquez-Rowe et al. 2010; Ziegler et al. 2015), case studies (Almeida et al. 2015; Avadí et al. 2015; Henriksson et al. 2014) and guidelines (BSI 2012; EPD 2014; Hognes 2014) for LCAs of seafood-based supply chains. From these, suggestions are given for best practices and more homogenised methods for LCA of seafood systems. Best practices were identified based on criteria such as a) their capacity to complete, complement and support the interpretation of life cycle inventory analysis and life cycle impact assessment results; b) their demonstration in literature beyond methodological proposal; c) our expert judgement based on an extensive contribution to the field by the co-authors. For instance, preferred fisheries-specific indicators complement conventional LCA by addressing ecological impacts and are easy to calculate (e.g. they rely on easily obtainable data), while preferred uncertainty management approaches have been demonstrated in fisheries and aquaculture case studies, and contribute to more robust interpretation of results.

3 Results

Across the reviewed studies, there was a strong focus on salmonids aquaculture in Europe and North America. Most studies that evaluated Asian aquaculture looked at *Pangasius* in Vietnam, a commodity mainly exported to the EU and the US. Carp farming in China, however, has been sparsely explored despite being the largest source of farmed fish. As for supporting data, many studies relied upon generic processes for feed resources from LCI databases. This was deemed concerning in some cases since the major LCI database, ecoinvent, mainly covers European agricultural production. Especially concerning was the use of fishmeal from the consequential LCAFood database (<http://www.lcafood.dk/>), since this process is incompatible with attributional LCA data and only describes fishmeal from sandeel in Denmark, a marginal source of fishmeal on global markets. LCA studies on fisheries have largely focused on industrial fleets targeting small and large pelagics, cephalopods and demersal fish.

Additional challenges identified, the following are of great relevance to improve the utility of LCA in the management of this industry: a) inclusion of fisheries management concerns and related impact categories (e.g. discards, by-catch, seafloor damage, biotic resource use, biomass removal impacts on the ecosystem and species); b) general LCA challenges in the specific context of seafood supply chains, such as the selection of functional units, the delimitation of system boundaries (e.g. inclusion of capital goods, end-of-life scenarios), cut-off criteria, allocation strategy, and selection of impact categories; c) data availability and data management; and d) the relation between LCA and seafood certifications. Seafood LCA guidelines were found to have either failed to include all relevant concerns or are yet to be widely applied by the industry, as noticeable from the documentary review (i.e. a consolidated set of practices is not widely applied by practitioners). Best practices were identified to address each challenge (**Error! Reference source not found.**).

Table 1 Challenges and identified best practices for seafood LCAs

Challenges		Best practices
Inclusion of fisheries management concerns	Capture data	Account for landings, discards, by-catch and on-board process losses (Vázquez-Rowe et al. 2012)
	Seafloor damage	Account for at least distance trawled per functional unit (Nilsson and Ziegler 2007)
	Biomass removal impacts	Prefer less data-intensive indicators (e.g. Langlois et al. 2014)
	Biotic resource use (BRU) ^a	Calculate BRU per functional unit, including all wild caught and agriculture-derived inputs to processes assessed (applies also to aquaculture and seafood processing)
	Management-related indicators	Include indicators derived from and informing fisheries management (e.g. Shin et al. 2010)
Methodological LCA challenges in the seafood context	Selection of functional units ^a	<ul style="list-style-type: none"> • Fisheries: volume of whole landed fish • Aquaculture: volume of whole live fish at farm-gate • Seafood processing: volume of final product
	Delimitation of system boundaries ^a	<ul style="list-style-type: none"> • Include capital goods (infrastructure, fishing vessels) • Include end-of-life in terms of material recycling and land use change • Model fate of by-products (e.g. fish processing residues, process water, excess heat) considering any raw materials they substitute in their receiving treatment/valorisation process (e.g. fish residues may partially substitute fresh whole fish in the fishmeal industry)
	Cut-off criteria	Include <i>ad-minima</i> inventories (Fréon et al. 2014; Henriksson et al. 2012; Vázquez-Rowe et al. 2012)
	Allocation strategy	Contrast mass-, economic- and gross energy content-based allocation, but use each consistently throughout the LCI; alternatively, treat it as choice uncertainty
	Selection of impact categories	<ul style="list-style-type: none"> • Select <i>ad-minima</i> lists of impact categories (Avadí and Fréon 2013; EC 2013; Henriksson et al. 2012; Vázquez-Rowe et al. 2012) • Include seafood-specific impact categories (BRU, biomass removal, etc.)
	Direct emissions	Aquaculture: nutrient budget modelling by means of mass balances (including weight gain, feed, faeces and not consumed feed, mortalities) to estimate direct emissions (Aubin et al. 2006)

Data availability and data management	Data gaps	<ul style="list-style-type: none"> • Reconstruction of missing data (e.g. fuel use) data from economic data (Fréon et al. 2014) • Approximate missing values within a dataset by multiple linear regression (Fréon et al. 2014)
	Uncertainty management	<ul style="list-style-type: none"> • Data uncertainty: Horizontal averaging of unit process data including estimates for uncertainty (Henriksson et al. 2013). For comparative purposes, perform dependent sampling and pair-wise comparisons (Henriksson et al. 2015a, 2015b) • Data and choice uncertainty: Statistical or pseudo-statistical methods for joint treatment (Andrianandraina et al. 2015; Mendoza et al. 2015)
Relation between LCA and seafood certifications		Use full-fledged LCAs to provide environmental indicators for and complement seafood certifications (Jonell et al. 2013)

^a Anchoveta Supply Chains project (<http://anchoveta-sc.wikispaces.com>)

4 Discussion

Much inventory data relevant to seafood LCAs have been collected. However, a great deal of it has unfortunately gone unreported and there is an overrepresentation of intensive systems in Western countries. Future efforts should therefore aim at collecting data on a more diverse set of countries and systems, and report these properly. Fishmeal and fish oil production also need to be better describe in literature as they often stand out as environmental hot-spots (see Fréon et al. (2016), in this conference, for a detailed study on the Peruvian fishmeal industry).

A large variety of indicators have been proposed by different research groups to cover seafood specific environmental impacts. The most relevant ones, useful to comparatively assess the status of exploited marine ecosystems, were compiled by the IndiSeas project (Shin and Shannon 2009). These indicators complement the environmental impact indicators informed by LCA. Moreover, additional key indicators pertinent to exploited marine ecosystems and fisheries have been proposed and used by environmental assessment practitioners, including those presented in **Error! Reference source not found.** Alternatives to these indicators, such as the fish-in fish-out ratio (Jackson 2009; Tacon et al. 2011) as an alternative to BRU, or the Lost Potential Yield (Emanuelsson et al. 2014) as an alternative to the impacts on the Biotic Natural Resource (Langlois et al. 2014), were not retained in our list due to additional complexity, refinement specific to certain supply chains but in our view not general enough, and reliance on not easily accessible data. Other indicators were excluded because they are indices based on more common indicators, such as the energy return on investment, which is the ratio of the energy contained in a seafood product and the industrial energy required for its production (e.g. gross energy or protein energy content per cumulative energy demand) (Tyedmers 2000; Vázquez-Rowe et al. 2014).

Key methodological, choice and study design challenges in LCA include the selection of functional units, delimitation of system boundaries, cut-off criteria, allocation strategies, selection of impact categories and estimation of direct emissions. Our retained best

practices are mainly based on our own experience applying LCA to fisheries, marine and freshwater aquaculture, and seafood processing. We believe the suggested approaches allow delivering more robust and objective results. In the case of allocation, for instance, the use of contrasting allocation keys prevents criticism of the results based on contrasting opinions and preferences by the research community (given that the ISO 14040 standard is subject of dissimilar and even contradictory interpretations).

Data and specially uncertainty management address critical elements determining the results of LCA studies. The quality of the life cycle inventories and an adequate propagation and incorporation of uncertainty into impact assessment results contribute to the robustness of the latter, and facilitate their interpretation. The approaches retained are relatively easy to implement and, in the case of the highlighted uncertainty management methods, they successfully address two of the main sources of uncertainty in LCA, namely data and choices. Addressing the uncertainty due to missing, inaccurate or imprecise characterisation factors is beyond the scope of these recommendations, yet we recommend using the latest and more complete impact assessment methods, models and characterisation sets available, and to clearly identify uncharacterised substances (e.g. antifouling molecules).

Impact assessment results from different studies should not be compared, because they may rely on different assumptions and methodological choices. Key inventory items such as fuel, water and chemicals use, on the other hand, can and should be contrasted per equivalent functional units for different studies, because interpretation of LCI outcomes also contributes with results and elements of interpretation on the studied system.

5 Conclusions

The adoption of these best practices for seafood LCAs promotes solid LCA studies with adequate data and uncertainty management, inclusion of seafood-specific impact categories, and coherent study design (elements of goal and scope). The wide adoption of best practices will thus contribute to improve the soundness of future LCA studies on seafood and other aquatic supply chains, including fish, algae, crustaceans and molluscs for direct or indirect human consumption.

Further methodological developments and consensus on how to address these challenges should ultimately contribute to make LCA a useful tool as a decision support tool for managers to visualize a wider scope of environmental indicators.

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120. Advancement of the marine biotic resource use metric in seafood LCAs: a case study of Norwegian salmon feed

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Seafood Life Cycle Assessment (LCA) studies have adopted the primary production required (PPR) measure to account for the impact of these production systems (e.g. capture fisheries or aquaculture) on the ecosystems they harvest wild inputs from. However, current practice often does not consider species- and ecosystem-specific factors and there is large diversity in the application of PPR.

We define and apply a refined method to a case study of average Norwegian salmon feed in 2012. This refined method incorporates species-specific fishmeal and oil yield rates, source ecosystem-specific transfer efficiency and the %PPR out of total ecosystem primary production. These results were compared to the most common previously applied method that employs global averages rather than ecosystem-specific factors and species-specific fishmeal and oil yield rates. Furthermore, we performed a Monte Carlo analysis to consider uncertainty and natural variability inherent to trophic level of species.

The proposed method presents results that are more refined based on current understanding than those employing an estimated global transfer efficiency, and average fishmeal and oil yield rates. While some sources of fishmeal and fish oil had very similar results irrespective of method used, species such as Capelin (*Mallotus villosus*) from ecosystems with low transfer efficiencies had a large divergence of PPR (Figure 1). This can be explained by the degree of divergence of input parameters relative to the averages. Overall, the refined method led to PPR results for Norwegian salmon feed in 2012 that were three times higher than the currently employed method signaling that previous LCA research may have substantially underestimated the marine biotic impacts of fisheries.

While the refined method further broadens diversity of assessment methods within seafood LCAs, this method should be adopted for future LCA studies to be more specific to the context of the study. This is an important step for seafood LCA to consider biological factors influencing production system environmental outcomes and to reduce diversity of practice for greater comparability of results.

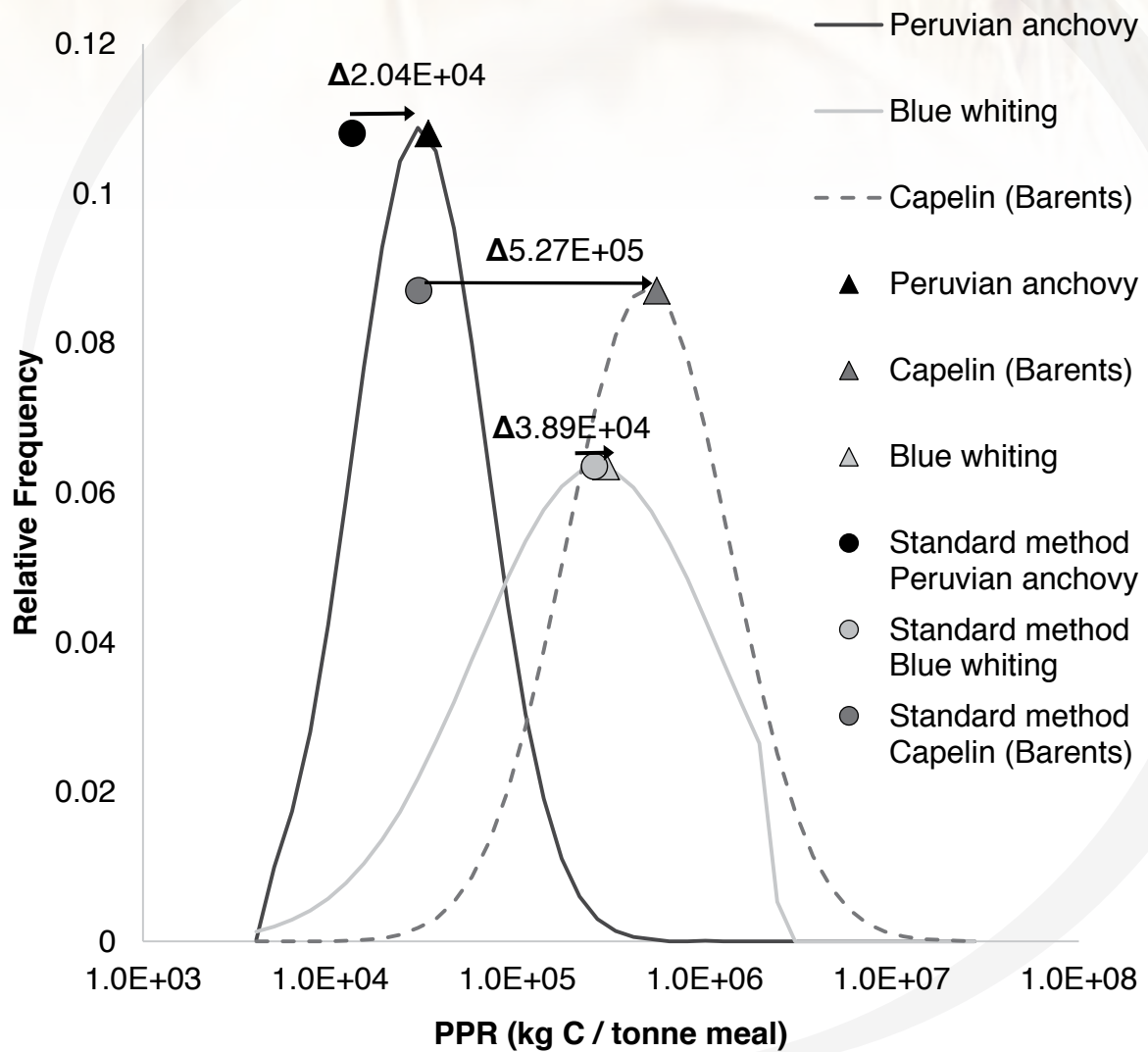


Figure 1. Primary production required (PPR; kg C/ tonne meal) on a logarithmic scale (X -axis) of blue whiting (*Micromesistius poutassou*), capelin (*Mallotus villosus*; Barents Sea), and Peruvian anchovy (*Engraulis ringens*). The curves represent the Monte Carlo distribution of results along with their relative frequency (Y -axis) of occurrence in the Monte Carlo analysis. The triangles represent the results of the refined method, which are irrespective of relative frequency. The circles represent the standard method results, which are irrespective of relative frequency. The arrows demonstrate the difference between the standard method and the refined method, with the difference between them indicated above.

119. Same stock, different management: Quantifying the sustainability of Skagerrak shrimp fisheries from a product perspective

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Background: The northern shrimp (*Pandalus borealis* L.) stock in the Skagerrak is shared by Sweden, Norway and Denmark. Although the fishery is regulated by an annual agreement between the EU and Norway, there are also national regulations as well as differences in fleet composition and shrimp markets. In early 2014, the World Wildlife Fund (WWF) gave all Skagerrak shrimp a red light in their seafood consumer guide, which led to an extensive debate, especially in Sweden, about the sustainability of the Skagerrak shrimp fishery.

Objective: The aim of this study was to quantify a set of indicators to provide a broad picture of the sustainability of the three different fisheries as a basis for a discussion on needed measures.

Method: This study takes a system approach and a product perspective, but is not a formal LCA. We defined relevant indicators of environmental, economic and social impacts that were quantified per tonne of shrimp landed by each country in 2012.

Main results: The Danish fishery was most efficient in terms of the environmental and economic indicators, while the Swedish fishery provided most employment per tonne of shrimp landed. Fuel use in all fisheries was high in all fisheries, and highly variable between vessels, stable over time (Fig. 1). Smaller vessels were more fuel efficient than larger ones in Sweden and Norway, with the opposite trend in Denmark (Fig. 2). The

study also demonstrated major data gaps and differences between the countries in how data is collected and made available.

Conclusions and Implications: With our relatively simple approach, we could demonstrate major differences and inconsistencies in data collection between the three fisheries on the same stock in the same area. Overcapacity in the Swedish fleet probably explains its inefficiency both in terms of fuel use, discarding and the trends between vessel sizes. Product-oriented studies of this kind could be useful in order to follow up performance of fisheries over time and to identify how to maximize the societal benefits generated while minimizing environmental impacts of a fishery. This could involve evaluating innovative solutions in terms of technology and management, based on current and future scenarios.

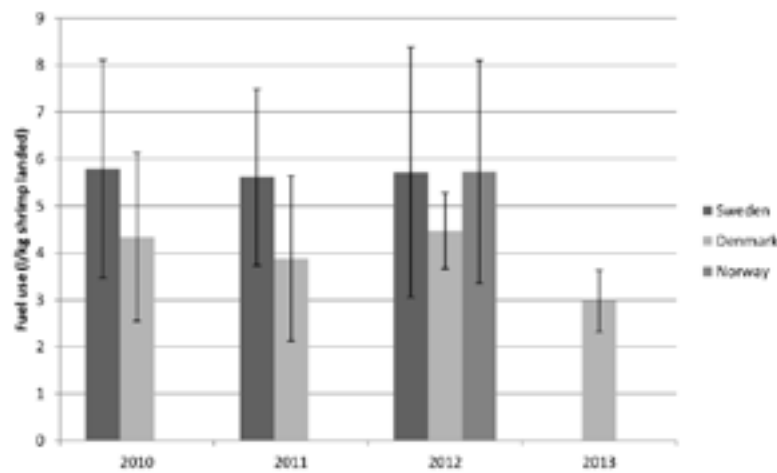


Figure 1. Fuel use over time in the three fisheries is relatively stable. Combinations of countries and years are missing due to lack of data.

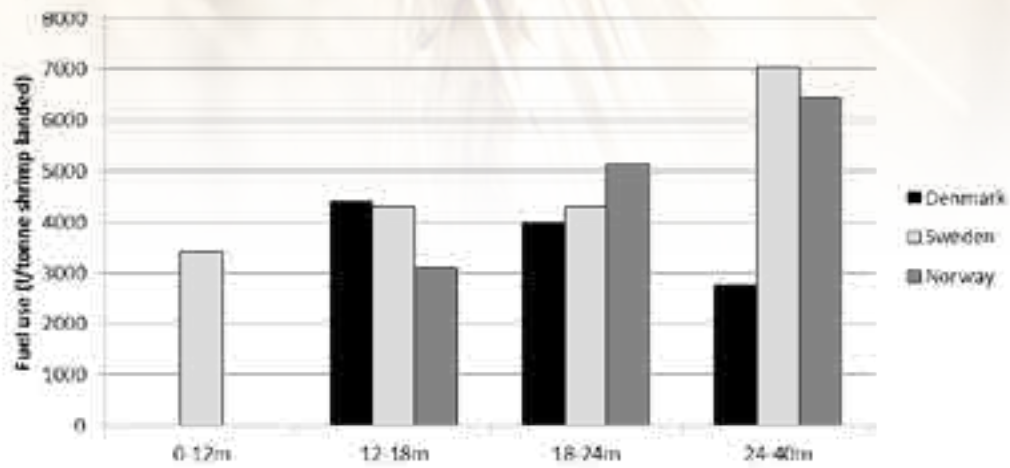


Figure 2. Fuel use in the three fisheries in 2012 per vessel size segment, showing opposite trends for Denmark than for Sweden and Norway. Note that data in some combinations of size and country is based on very few vessels and therefore is uncertain.

For more detail: Ziegler et al. in press ICES Journal of Marine Science

118. Life cycle environmental impacts of a northeast Atlantic trawler on a fishing trip basis, including a novel approach to assess biotic impacts of fishing

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Background and objective: Capture fisheries is the only industrial-scale harvesting of a wild resource for food. Temporal variability in environmental performance of fisheries has only recently begun to be explored, but only between years, not within a year. Our aim was to better understand the causes of temporal variability within and between years and to identify improvement options through management at a company level and in fisheries management.

Method: We analyzed the variability in broad environmental impacts of a demersal freeze trawler targeting cod, haddock, saithe, and shrimp, mainly in the Norwegian Sea and in the Barents Sea. The analysis was based on daily data for fishing activities between 2011 and 2014 and the functional unit was a kilo of landing from one fishing trip. We used biological indicators in a novel hierarchic approach, depending on data availability, to quantify biotic impacts. Landings were categorized as target (having defined target reference points) or bycatch species (classified as threatened or as data-limited). Indicators for target and bycatch impacts were quantified for each fishing trip, as was the seafloor area swept.

Main results: No significant difference in fuel use was found between years, but variability was considerable within a year, i.e., between fishing trips. Trips targeting shrimp were more fuel intensive than those targeting fish, due to a lower catch rate. Steaming to and from port was less important for fuel efficiency than steaming between fishing locations. A tradeoff was identified between biotic and abiotic impacts (Figure 1). Landings classified as main target species generally followed the maximum sustainable

yield (MSY) framework, and proportions of threatened species were low, while proportions of data-limited bycatch were larger. This improved considerably when reference points were defined for saithe in 2014.

Conclusions and Implications: The large variability between fishing trips shows that there is room for improvement through management. Fuel use per landing was strongly influenced by target species, fishing pattern, and fisheries management. Increased awareness about the importance of onboard decision-making can lead to improved performance. This approach could serve to document performance over time helping fishing companies to better understand the effect of their daily and more long-term decision-making on the environmental performance of their products. Fishing companies should document their resource use and production on a detailed level. Fuel use should be monitored as part of the management system and managing authorities should ensure that sufficient data is available to evaluate the sustainability of exploitation levels of all harvested species.

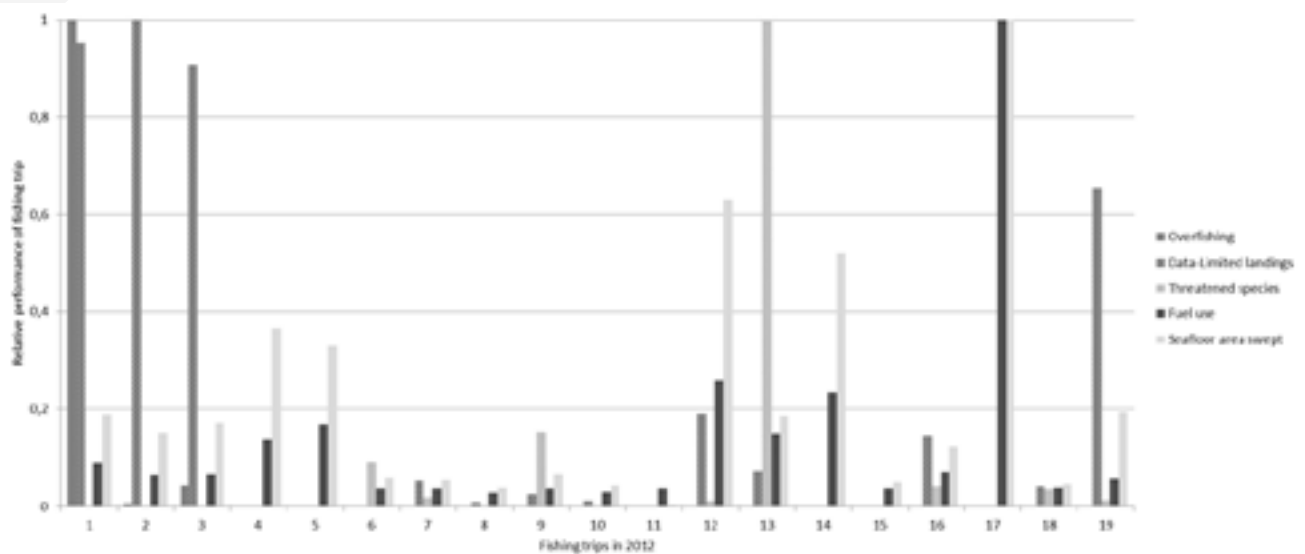


Figure 1. Relative results for the 19 fishing trips in 2012 for the four biological indicators quantified and fuel use demonstrating trade-offs.

For more details: Ziegler et al. 2015 International Journal of LCA

145. Environmental assessment of sea cage aquaculture in Tunisia using Life Cycle Assessment (LCA)

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ABSTRACT

Life Cycle Assessment (LCA) was performed to characterize the potential environmental impacts of the aquaculture systems in Tunisia. A comparative LCA of the production of 1 ton of fish was conducted on 18 sea cage aquaculture farms of European seabass and gilthead seabream, followed by a classification of the farms using the Hierarchical Clustering on Principal Components (HCPC) based on the rearing systems characteristics and practices. The estimated impacts from the LCA were associated as illustrative variables. Then, we established a distinctive LCA for each farm group to further characterize the aquaculture industry. Based on the HCPC, we divided the aquaculture farms into 6 different types principally according to the annual production, the water depth, and the feed conversion ratio (FCR). Results show that for all impact categories, feed and fry production are the main contributors to acidification and climate change which are directly related to the production of fish meal and fish oil as feed ingredients, and to the fact that they are fully imported. Eutrophication is mainly linked to the fish growing phase, which strongly depends on the FCR, due to the uneaten and the undigested fraction of feed. Results highlight the importance of adapting the farming techniques and the feeding practices (stock management, feed distribution, and accurate ration calculation).

Keywords: Life Cycle Assessment (LCA), fish farming, environmental impact.

1. Introduction

Fish farming is considered as one of the fastest growing food production sectors worldwide (FAO, 2012). Indeed, aquatic products play an increasing role in human nutrition. World demand of food fish increased from 9.9 kg per capita in the 1960s to 18.6 kg per capita in 2010 (FAO, 2012). However this increase of demand is no longer sustained by the fishing activity, which remained stable over the last three decades against an important growth of the aquaculture production with an average of 8.8% per year between 1980 and 2010 (FAO, 2012). At present, approximately 40% of consumed fish are farmed, and this percentage is predicted to reach 60% by 2030 (FAO, 2014).

Aquaculture bears risks of negative impacts upon environment through the use of natural resources (Naylor et al., 2000) and the emissions of pollutants and effluents (Read and Fernandes, 2003) with the ability to affect and change the ecosystems and the biodiversity (Tovar et al., 2000).

Tunisia is occupying a central place in the Mediterranean, and it has more than 1300 km of coastline. Fisheries and aquaculture play an important role in socio-economic terms and as a source of food. Aquaculture activity is mainly marine oriented, with an annual production of almost 10000 tons in 2014 (DGPA, 2014). The most important species in terms of farming value are the European seabass (*Dicentrarchus labrax*) and the gilthead seabream (*Sparus aurata*). In the light of current social, economic and environmental constraints, it becomes necessary that aquaculture production systems in Tunisia evolve in the context of sustainable development, in order to assess and control the negative impacts of this activity. The present study aims to

evaluate and compare the environmental impacts of 18 different sea cage aquaculture farms of seabass and seabream using Life Cycle Assessment (LCA).

2. Methods

2.1. Goal and Scope

A comparative cradle-to-farm-gate LCA was conducted to assess environmental impacts of the production of 1 ton of fish on 18 sea cage aquaculture farms of European seabass and gilthead seabream. Then, a Hierarchical Clustering on Principal Components (HCPC) was applied to characterize and get an efficient classification of the farms. We considered farms characteristics and practices as principal variables:

- total surface of the studied farms: varies between 24 and 84 hectares.
- annual production: varies between 480 and 3000 ton.
- number of cages: 15 cages in small farms and a maximum of 90 cages in big farms.
- diameters of the cages: only three different diameters 22m, 25m or 29m.
- Feed Conversion Ratio (FCR) calculated by dividing the total feed intake by the net production, reflecting the efficiency of the feeding strategy and the cost-effectiveness of using a particular feed. FCR values vary between 1.4 and 2.3 for seabream and between 1.6 to 3 for seabass.
- length of the production cycle: periods fluctuates between 10 and 17 month.
- water depth: varies between 20 m and 40 m of depth.
- fish feed: the annual fish feed quantity necessary for the total production varies between 300 and 20000 ton.
- fry: fingerlings quantity required by the farms in Tunisia varies between 960000 and 6540000 individual.

The estimated impacts from the LCA were considered as illustrative variables (eutrophication, climate change and Acidification). Afterwards, a distinctive LCA was performed for each farms category. The functional unit chosen was "1 ton" of fish over one normal year of production. System boundary includes the rearing stage, transportation and importation of larvae and feed. However, distribution, marketing and use (consumption, processing, conditioning, etc) are not included in the study (Figure 1).

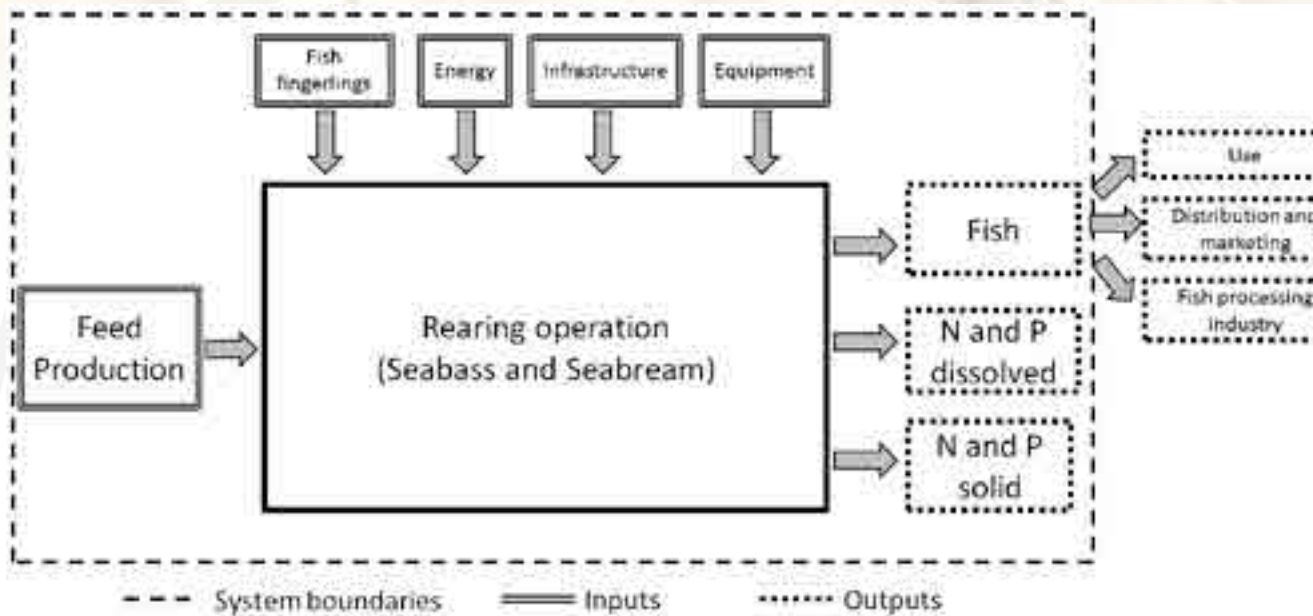


Figure 1: Diagram representing stages of the fish production system studied

2.2. Life Cycle Inventory (LCI)

Inventory data were obtained from the Fisheries and Aquaculture Department (DGPA) statistics and from farms records. Some field surveys were conducted in order to gather more detailed information to validate the previously collected data and to complete the inventories. Data referring to the fish feed chemical and ingredients composition were determined based on the commercial labels.

In terms of outputs, the amounts of nutrients emission associated with the fish growth phase were calculated based on a mass-balance model (Cho and Kaushik, 1990). The solid and dissolved fractions of the emitted Nitrogen (N) and Phosphorus (P) were estimated based on the difference between the amounts of nutrients provided to fish and the amounts assimilated as fish weight-gain, and the calculations took into account the nutrients digestibility of the feed, the fish body composition and the non-ingested feed estimations. This modelling approach has been previously adapted and validated for several fish species (Bureau et al., 2003; Kaushik, 1998; Lemarié et al., 1998; Mallekh et al., 1999) and it has also been used to put in place the Life Cycle Inventory (LCI) of fish-production systems (Aubin et al., 2009; Jerbi et al., 2012; Mungkung et al., 2013).

2.3. Life cycle impact assessment (LCIA)

The inventory data (consumptions and emissions) were aggregated into impact categories and reported to one ton of fish produced. The environmental impact assessment was conducted following the CML2 Baseline 2000 method using Simapro®8.0 and Ecoinvent V3.0 database for background data. We investigated three impact categories selected based on previous guidelines in aquaculture LCA (Aubin et al., 2009; Jerbi et al., 2012; Mungkung et al., 2013; Pelletier et al., 2007):

- Acidification: refers to the negative effects generated by the release of compounds capable of increasing the acidity of water and soils, which may results in acid rains. It is

expressed in kg SO₂-equivalent and it was calculated using the average European acidification potential factors following Huijbregts (1999).

- Eutrophication: refers to the negative impact of the presence of an excess amount of nutrients in the environment, it is expressed in kg PO₄-equivalent and it was calculated using the factors found in Guinée et al.(2002).

- Climate change: represents the potential impact of gaseous emissions on heat-radiation absorption in the atmosphere, it is expressed in kg CO₂-equivalent and it was calculated according to the climate change potential 100 factor (GWP100) used by the IPCC (International Panel on Climate Change).

3. Results

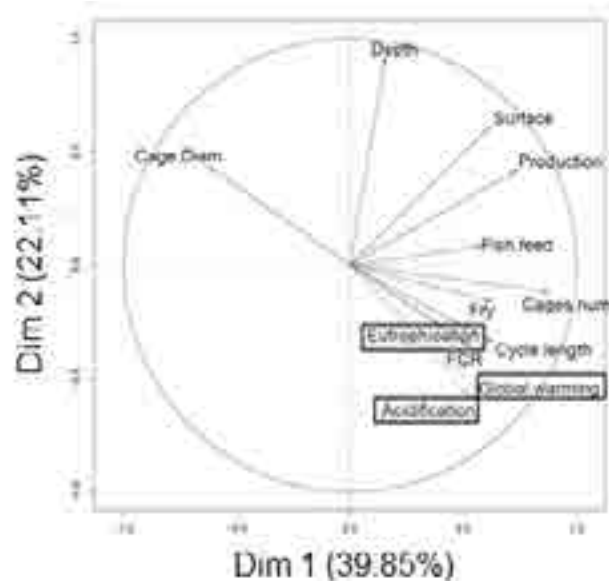


Figure 2: Variables factor map of the PCA, the three illustrative variables obtained from the LCA (Eutrophication, Climate change and Acidification) are in frames.

The correlation circle of the HCPC shows the correlated variables and their contribution to the axis. The first principal component explains 40% of the total variance and it reflects mainly the number of cages in the farms and the length of the rearing cycle in the positive side and the diameter of the cages in the negative side. The second axis explains 22% of the total variance and it is principally represented by the sea depth (Figure 2).

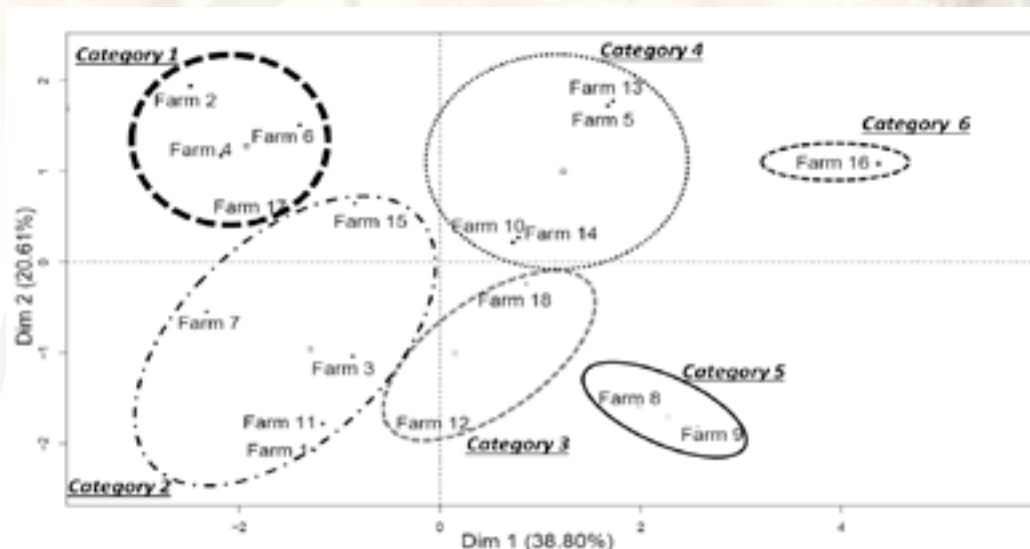


Figure 3: Classification of aquaculture farms in Tunisia using HCPC method

Table 1: The characteristics of the 6 farm types according to the HCPC.

	Category 1	Category 2	Category 3	Category 4	Category 5	Category 6
Characteristics of the category	Small farms with low FCR and important sea depth	Small farms with low FCR and shallow sea depth	Small farms with high FCR and important sea depth	Large farms with low FCR and important sea depth	Large farms with high FCR and shallow sea depth	Large farms with high FCR and important sea depth

The individuals map shows that small farms with low FCR and important sea depth (category 1) and large farms with high FCR and shallow sea depth (category 5) are totally opposite on the diameter of the cages and the sea depth. Small farms with low FCR and important sea depth is also opposite to large farms with high FCR and shallow sea depth (category 6) (Figure 3).

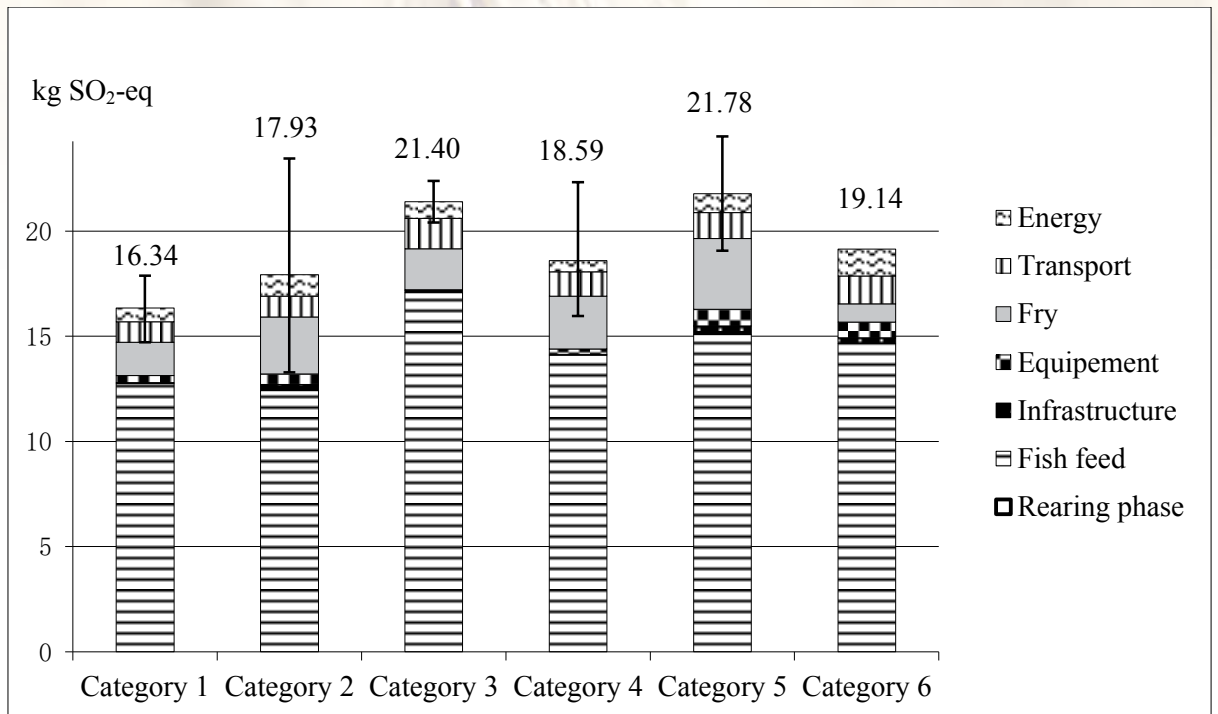


Figure 4: Potential acidification impact per ton of fish (seabream and seabass) and the contribution of the system components for each type of farms. Bars represent the variability inside each farm type (minimum/maximum).

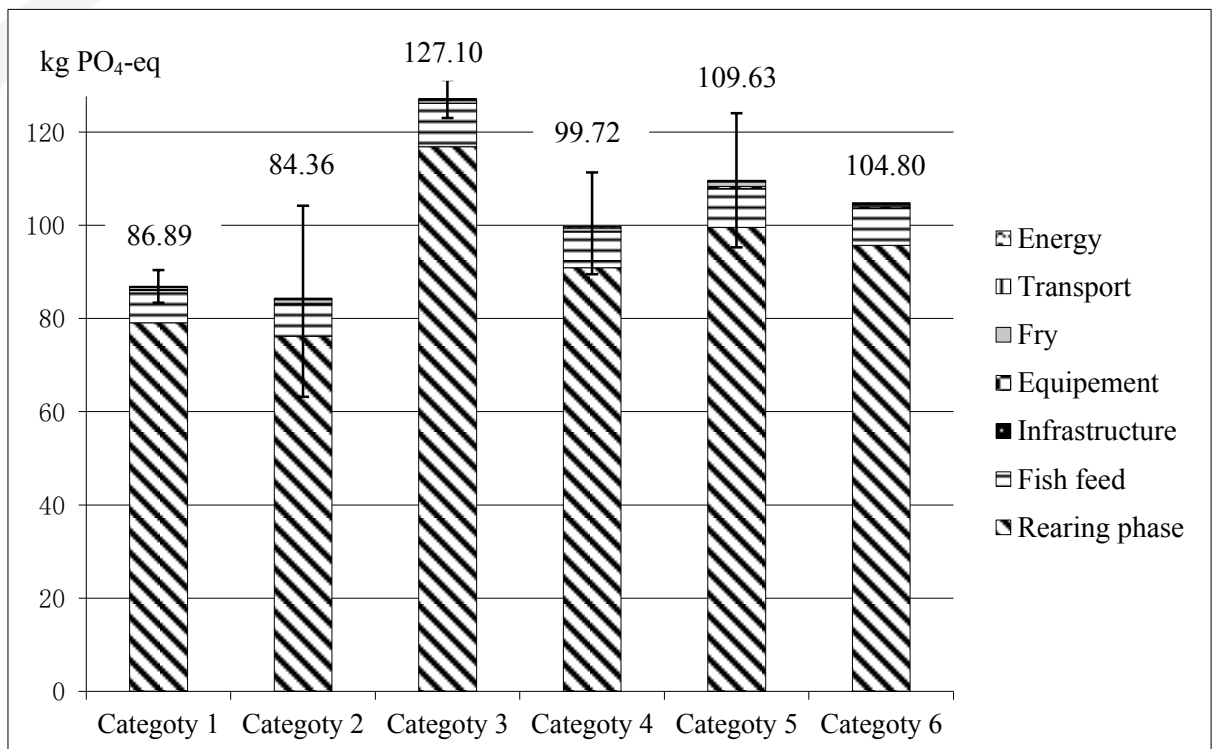


Figure 5: Potential eutrophication impact per ton of fish (seabream and seabass) and the contribution of system components for each type of farms. Bars represent the variability inside each farm type (minimum/maximum).

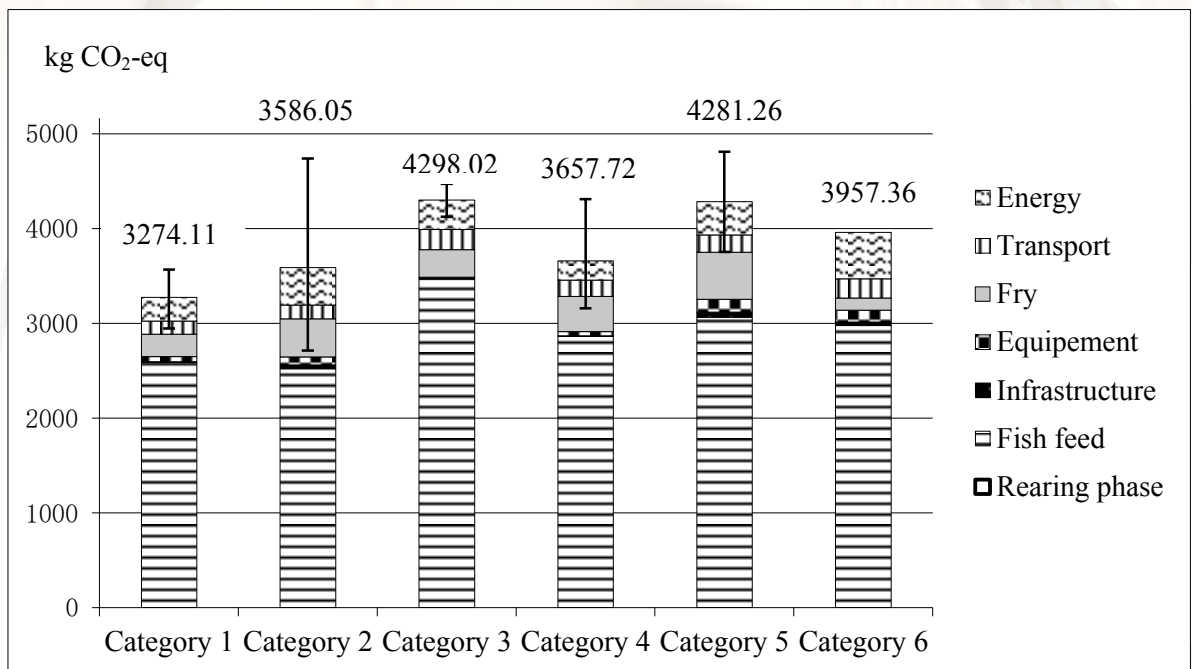


Figure 6: Potential climate change impact per ton of fish (seabream and seabass) and the contribution of the system components for each type of farms. Bars represent the variability inside each farm type (minimum/maximum).

Figure 4, Figure 5 and Figure 6 shows that small farms with high FCR and important sea depth and large farms with high FCR and shallow sea depth have the most important impact at the level of three impact categories. On the other hand, small farms with low FCR and important sea depth and small farms with low FCR and shallow sea depth are the less impactful. For all types of farms, the main contributor to acidification is the fish feed (Figure 4). The figure 5 shows that the rearing phase (farm operations necessary to produce the fish) is the principal contributor to the eutrophication, and the second contributor is the fish feed production (Figure 5). Fish feed is also the main contributor to the climate change followed by the fry production (Figure 6).

4. Discussion

Results of the LCA and the HCPC for the 6 farm types show that small farms with high FCR (2.35) and important sea depth and large farms with high FCR (2.10) and shallow sea depth have the most significant impact on the three impact categories studied. On the opposite level, small farms with low FCR and important sea depth and small farms with low FCR and shallow sea depth are the less impactful. We can conclude that a low FCR is really important in order to minimize the negative impact of aquaculture farm. The LCA shows also that the main source of acidification and climate change is feed-derived, more specifically uneaten and undigested feed residues and also excretion products. Owing the tight relation between the FCR and the feed related wastes, we confirm that fish farms with higher FCR (Small farms with high FCR and important sea depth and Large farms with high FCR and shallow sea depth) are the most important contributors to those impacts with a release of more than 21 kg SO₂-eq of which fish feed is responsible for 73%. The acidification impact in all Tunisian farms is lower than the acidification impact of Seabass culturing in sea cages in Greece (25 kg SO₂-eq) (Aubin et al., 2009), and in traditional and cascade raceway in Tunisia (54 and 70 kg SO₂-eq, respectively)

(Jerbi et al., 2012). On the other hand, both categories emitted more than 4281 kg CO₂-eq per year to produce 1 ton and fish feed contribution is around 75%, while Aubin et al. (2009) estimated an emission of 3601 kg CO₂-eq and 86% feed contribution to the climate change in seabass cage farms in Greece. Jerbi et al., (2012) estimated much higher impact of Seabass rearing in traditional and cascade raceway than all sea cages farms (11087 and 17500 kg CO₂-eq, respectively).

Eutrophication is mainly linked to the rearing operation due to nutrient emissions into the environment with a contribution that exceeds 90% and an emission of 127.1 kg PO₄-eq by small farms with high FCR and important sea depth and 109.63 kg PO₄-eq emitted by Large farms with high FCR and shallow sea depth. Both of these categories exhibit higher impact than Seabass rearing in sea cages in Greece (Aubin et al., 2009). However, the eutrophying emissions for all studied farms are lower than emissions of Seabass rearing in traditional and cascade raceways in Tunisia, with 180 kg PO₄-eq and 215 kg PO₄-eq, respectively (Jerbi et al., 2012). In fact, eutrophying emissions are directly related to the water depth under the cages and to the productivity level and the quantities of feed provided, so farms with high FCR have a higher impact at the eutrophication level.

This leads to the conclusion that the impact generated by fish farms is less pronounced when the sea depth under the cages is important. In fact, shallow depth induces the accumulation of organic matter and the degradation of the water quality and eventually the proliferation of pathogens, while high depth results in the increase of the degree of dispersion of wastes and consequently minimizing the negative impacts. This conclusions is in line with what has been proven by previous studies and models (Cromey et al., 2002; Gowen, 1994). This study shows the capital importance of adapting the farming and the feeding practices (stock management, feed distribution, accurate ration calculation) in order to reduce the FCR and consequently minimize the acidification and climate change impacts generated by the production. In addition, the important contribution of the fish feed is directly related to the production of fish meal and fish oil as feed ingredients, and to the fact that they are fully imported, so putting in place a facility to produce locally fish feed and avoid their import can be beneficial to reduce the environmental impact. Also looking for other alternative ingredients of the fish meal and fish oil could help improving the performance of fish farms.

5. Conclusions

This study aimed to evaluate the environmental performance of culturing seabass and seabream in sea cages, which constitutes the major aquaculture activity in Tunisia. The quantification was expressed for the production of 1 ton of fish over one normal year of production. According to our results, we concluded that fish feed is the main contributor to acidification and climate change, and that the rearing phase is responsible for the eutrophying emissions, this is directly related to the FCR which have a direct repercussion on all impact categories studied. Therefore, taking measures to optimize the feeding practices and the diet formulation would have a positive impact on the overall environmental performance of fish farms. Moreover, results gathered from this study show that the establishment of fish farms in areas with important sea depth would minimize the negative impact of the rearing activity. Those conclusions can help pinpoint the key elements that need to be reviewed and provides recommendations in order to place the Tunisian aquaculture activity in the context of sustainable development taking into account environmental objectives.

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264. Assessing alternative scenarios for the Indonesian aquaculture sector up to 2030 using exploratory LCA

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ABSTRACT

The Indonesian aquaculture industry is growing at an incredible rate, a rate that could speed-up or tackle off up to 2030, depending upon how the industry develops. In order to provide insight into the consequences of different development paths of the Indonesian aquaculture industry, the present study combines future projections with LCA. The future growth projections were generated using the Asiafish model, while the LCA studies were based upon primary data for ten different aquaculture systems in Indonesia. Global warming, acidification, eutrophication, land-use, freshwater- and energy use were evaluated across the different lifecycles, alongside wild fish used in feeds, employment at farms, monetary value at farmgate. The systems range from off-shore cage aquaculture of grouper, a carnivorous high-valued species, to pond aquaculture of *Clarias* catfish, a omnivorous low-valued species. The outcomes show that the Indonesian aquaculture industry primarily need to focus on developing freshwater aquaculture of omnivorous species to enable sustainable growth of the aquaculture industry. Indonesia also need to review local, more sustainable, feed resources.

Keywords: Indonesia, aquaculture, projection, fish, LCA

1. Introduction

Aquaculture production in Indonesia has increase ten-fold over the last ten years, largely as a response to collapsing capture fisheries (Figure 1). Aquatic plants dominate production, but these are of low value and generally cause limited environmental impacts. Apart from aquatic plants, the main species are Nile tilapia (23% of total production), Torpedo-shaped catfishes (23%), Milkfish (13%), Whiteleg shrimp (10%), Common carp (10%), Pangas catfishes (10%), and Giant tiger prawn (3%) (FAO 2016). Most of these species are omnivorous (tilapia, shrimp, milkfish and catfishes), while others produced at lower quantities (e.g. grouper (0.3%)) are carnivorous and require a high protein diet. The finfish and crustaceans are grown in a variety of systems, including freshwater cages and ponds, brackishwater ponds, and marine cages. The Indonesian government target is to further increase production with the goal of expanding the sector by 30% up to 2019.

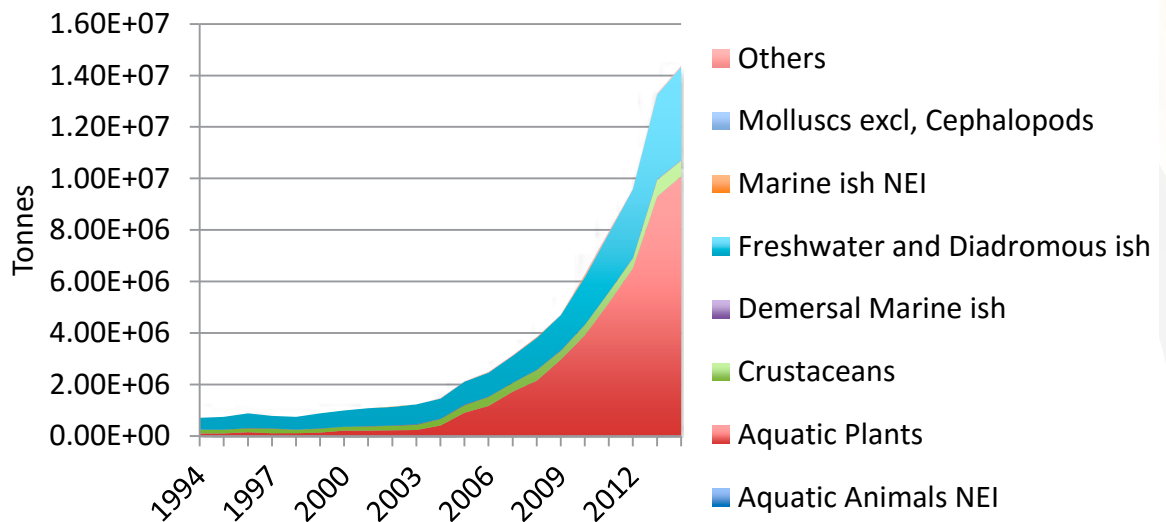


Figure 1: Indonesian aquaculture production between 1994 and 2014 by species group. Data from (FAO 2016)

With this rapid expansion, some serious environmental concerns have also arisen. For example, brackishwater pond aquaculture has been blamed for 49% of the mangrove forest loss in Indonesia since 2000 (Richards and Friess 2015). The provision of feed has also been associated with several environmental impacts, including global warming, acidification and resource use (Henriksson et al. 2012). The aquaculture farms also emit nutrients to surrounding environments, resulting in eutrophication and loss of biodiversity (Mungkung et al. 2013).

In order to identify potential environmental tragedies in Indonesia, as a result of a rapidly growing aquaculture industry, the present study use the AsiaFish economic model (Dey et al. 2005) to predict the future growth of the aquaculture industry, together with life cycle assessment (LCA), to identify sector-wide environmental impacts.

2. Methods (or Goal and Scope)

The present research was financed in part by the Moore Foundation and carried out in order to provide recommendations to the Indonesian department of Fisheries. The ambition was to benchmark the major production systems using LCA and evaluate their overall environmental impact. The LCAs were carried out using the protocol for horizontal averaging by Henriksson et al. (2014), CMLCA and ecoinvent v2.2. The functional unit was one tonne of whole fish at farm-gate. The impacts at farm-gate were later multiplied per species with the projected production in six different alternative future projections generated using the AsiaFish model. The future projections assumed: business-as-usual; stagnating capture fisheries; export-oriented aquaculture industry; domestic driven aquaculture; slow aquaculture growth; and disease struck aquaculture. For these future projections, six impact categories were evaluated (global warming, acidification, eutrophication, land-use, freshwater consumption and energy use), alongside wild fish use, employment and monetary value as socio-economic proxies.

Primary life cycle inventory (LCI) data were collected during 2014 on Java, Sumatra, Sulawesi and Lombok. The systems detailed were tilapia, Clarias and Pangasius in freshwater ponds, tilapia and carp in freshwater cages, shrimp and milkfish in brackish water ponds, and grouper in marine cages. Between three and six farms for each system were approached using questionnaires.

3. LCI

Among the different systems, milkfish and carps had the largest feed requirements in terms of pelleted feeds, while groupers consumed the largest quantities of low-value fish. However, the composition of the feeds varied greatly, with fishmeal inclusions ranging from 1% in milkfish feed to 15-25% in shrimp feeds, to 50% in grouper feeds. The energy distribution also varied among the systems with diesel and electricity used in the tilapia and carp farming systems, while the main energy consuming practice in grouper farming was gasoline for travels to and from the cages.

Table 2: Selected lifecycle inventory data for different farming systems in Indonesia.

	Feed pellets	Low-value fish	Fertilizer	Fuel/ electricity	Land use	Fresh-water use	Fin-fish	Crustaceans
Unit	kg	kg	kg	MJ	m ²	m ³	kg	kg
Tilapia, ponds	1660			2715	58	18.4	100	
Tilapia, cages (n=5)	1600			2414	0	0	100	
Carp, cages (n=6)	1810			1854	80	18.7	100	
Clarias, ponds (n=5)	1168				33	10.3	100	
Pangasius, ponds (n=3)	1400				195	61.8	100	
Pangasius, cages (n=6)	1502			324	0	0	100	
Milkfish, ponds (n=3)	1917		271,5	922	9048	2870	100	
Grouper, cages (n=5)	258	17966		62498	0	0	100	
Whiteleg shrimp, ponds (n=6)	1552		87	9214	2380	2850		1000
Tiger shrimp,	1625	571	2277	21902	3008	15600	954	1000

4. LCIA

The outcomes indicated that domestically and export driven aquaculture industries would be the most productive in terms of quantity of fish and monetary value. The export-oriented future, however, was expected to result in an eight-fold increase in freshwater use and wild fish use. This translates to quantities of fish used in feeds that would exceed the expected total capture fisheries' landings of Indonesia before 2030. As for the domestic oriented future, the requirements of land were largest, raising concerns about more loss of biodiversity across the Indonesian archipelago. The slow aquaculture growth and disease struck aquaculture future projections, in the meantime, had the most modest increases in terms of both fish and monetary value.

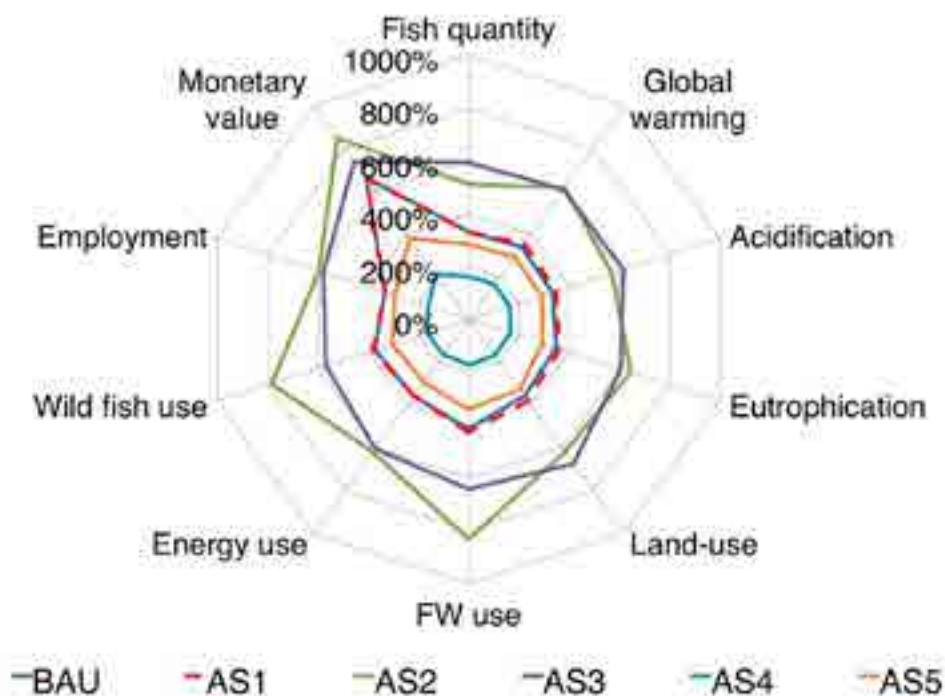


Figure 2: Relative change of environmental impacts and socio-economic indicators between 2012 and 2030 for: BAU =business-as-usual; AS1 = Stagnating capture fisheries; AS2 = Export-oriented aquaculture industry; AS3 = Domestic driven aquaculture; AS4 = Slow aquaculture growth; and AS5 = disease struck aquaculture

5. Conclusions (or Interpretation)

Different aquaculture systems result in distinctly different environmental impacts. Grouper farming performed the worst in terms of global warming, acidification, eutrophication, energy use and wild fish use. However, it also required the least amount of land and freshwater, given that the groupers are farmed in marine cages using mainly wild fish. Groupers, moreover, generated the largest amounts of monetary value, followed by shrimp. The freshwater fish, in the meantime, had lower requirements in terms of wild fish and energy, while causing lower global warming, acidification and eutrophication impacts. Especially pangasius and clarias were efficient at producing fish protein at low environmental costs. These were unfortunately also the species of lowest monetary value.

Our analysis clearly shows that the Indonesian government's targets are next to impossible to meet, especially if higher valued species are to be prioritized. Instead, we recommend a stronger focus on developing freshwater aquaculture at a more modest rate. We also promote research into developing alternative feed resources that could be sourced locally, which would secure production and lower environmental impacts. This would help feed Indonesia's growing population while limiting environmental impacts. The value of these species should, however, be improved through, for example, value additions.

The indicators labor and generated monetary value generated at grow-out were included as proxies for social and economic impacts. These indicators could in the future also be elaborated and applied to the whole lifecycle chain, as a step towards lifecycle sustainability analyses

(Guinée 2015). However, in the process of doing so, one need to consider issues such as profits, rather than sales, and working conditions.

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8. Preliminary LCA of three Peruvian fishmeal plants

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ABSTRACT

Fishmeal and fish oil are largely used as input to several animal feed industries, but there is a lack of LCAs on Peruvian fishmeal plants, despite their predominance in the global supply. Preliminary LCAs were performed on three different types of Peruvian fishmeal plants with the objective of comparing them and suggesting ways of limiting their impacts. Two system boundaries were used: one including the fishery and another excluding it in order to enable others to use our dataset. We used the SimaPro software, the ecoinvent 2.2 database and the ReCiPe method. Despite the predominant impact of the use phase, in particular consumption of fossil energy, the construction and maintenance phases contribute significantly when fishing is excluded from the system boundaries. Furthermore, existing screening LCAs of the use phase underestimate significantly its environmental impacts. The environmental benefit of using natural gas instead of heavy fuel as energy source is quantified. The comparison of environmental impacts between different qualities of fishmeal shows higher impacts of residual fish meal, intermediate impact of standard fishmeal and lower impacts prime fishmeal. Future studies on other fishmeal and residual fishmeal plants should take into account the construction and maintenance phases, and more items in the use phase than in historical screenings. There is room to decrease the environmental impact of this industry in Peru.

Keywords: Animal feed; Cleaner production; Fisheries; Fishmeal; Fish oil; Peruvian anchovy

1. Introduction

1.1. Rationale

Intensive or semi-intensive farming of livestock and aquatic animals requires feed with high protein and lipid contents, some of which must be of animal origin to supply essential amino and fatty acids. Those two ingredients are found in fishmeal and fish oil (FMFO) respectively. Although the substitution of those two commodities by cheaper products of vegetal and animal origin is increasing, the increase in farming of livestock and aquatic animals counterbalances these substitutions and the FMFO demand for animal feed is still growing. There is also a growing demand of fish oil for human consumption (omega-3).

In the aquaculture sector, LCAs demonstrated that feed provision accounts for a large share in many of the environmental impacts in this sector (Henriksson et al., 2012, 2015). FMFO contribution within fish feed environmental impacts is substantial and usually ranks first in fish feed of carnivorous species such as salmon and trout (e.g. Pelletier et al. 2009; Avadí et al., 2015). Moreover, feeds for farmed herbivore fish often include small amounts of FMFO, thus representing a large aggregated consumption due to the large share of these families in the worlds' aquaculture output (Chiu et al., 2013; Henriksson et al., 2014). Nonetheless the precision of FMFO impacts in most studies is hindered by the lack of a life cycle inventory (LCI) of the FMFO production process. As far as we know, only Denmark benefits from a rough LCI of fishmeal plants, whereas Peru and Norway only benefit from an even more superficial screening. The Danish fishmeal plant LCI, available at <http://www.lcafood.dk/>, was performed in 2000 and most its data were used as proxies for the other LCIs, in addition to few generic data for freshwater use and waste water (FAO, 1986; COWI, 2000). According to Henriksson et al. (2014) fishmeal environmental impacts could differ with two orders of magnitude depending upon its origin.

1.2. The Peruvian FMFO sector

The Peruvian FMFO sector produces in average (2006-2015) 1.183 million t of fishmeal and 230 000 t of fish oil per year, which represent 24% and 23% of the global production, respectively. Peru exports most of this production which relies on the extremely high abundance of the Peruvian anchovy (*Engraulis ringens*), commonly referred to as 'anchoveta'. This species is also characterized by its high variation in abundance at different time scales (seasonal, inter-annual and inter decadal).

The production of FMFO is supplied by the Peruvian industrial fleet of purse-seiners, which by law consists of vessels whose holding capacities are over 32.6 m³ and land their catches exclusively for reduction into FMFO. This fleet subdivides into two major segments: steel vessels and wooden hull vessels (Fréon et al., 2014b). There is also a Peruvian wooden small- and medium-scale (SMS) fleet of purse-seiners with holding capacity under 32.6 m³. SMS vessels are allowed by legislation to land anchoveta exclusively for direct human consumption (DHC), but from 2012, 10% of the small-scale anchoveta landings and 40% of the medium-scale one can be legally redirected to reduction under certain conditions. Up to 2008 the industrial fishery was regulated by a single quota whereas the SMS fishery benefited from a full open access. From 2009, an Individual Vessel Quotas (IVQs) system was fully implemented for the industrial fleet. Illegal, unreported, and unregulated (IUU) fishing is a recurrent problem in Peru (although improving), and in the SMS fleets operations it reached 200% over the officially reported figures (Fréon et al., 2014b).

Three different categories of fishmeal were produced in Peru during the study period (2010-2013), where quality depends mainly on protein, lipid and salt content:

- 1) Standard fishmeal, also are referred to as "fair average quality" (FAQ), usually produced using direct hot air during the drying phase ("flame drying" or "direct-fire drying"),
- 2) Prime fishmeal,
- 3) Super prime fishmeal; for producing prime fishmeal and super prime fishmeal, special driers are needed; typically hot air is produced by circulation of steam inside the dryer ("indirect steam drying").

There is no clear definition of fish oil categories in Peru, except for the recent (2009) European sanitary regulation on fish oil importation. There are three main types of fishmeal plants operating in Peru:

- 1) Residual plants which, in principle, are only allowed to process fish residuals and unsuitable fish of different species aimed at DHC. In practice, most of these plants process 30-50% of IUU anchoveta;
- 2) Traditional FAQ plants, which use mostly anchoveta as raw material. Both residual and traditional plants are producing only FAQ fishmeal and consume mainly heavy fuel as energy source;
- 3) Modern steam plants, which produce both and prime and super prime quality fishmeal and also use mostly anchoveta as raw material. These plants consume both heavy fuel and natural gas when available.

All traditional and modern steam plants belong to fishing companies that operate their own steel vessels and, in addition, buy fish from the wooden industrial vessels. In the recent period, the quality of the Peruvian FMFO increased and the production of FAQ fishmeal remains only in small plants.

There is a total of 207 fishmeal plants constructed in Peru, including 37 with cancelled permits, which correspond to an impressive total processing capacity of 11 400 t per hour (9 350 excluding plants with cancelled permits).

These plants are located all along the Peruvian coast, with concentration close to the main fishing harbours, which generates social conflicts between the industry and the local population about the nuisances of the plants (odour nuisance and costal water contamination). One important

characteristics of nearly all the large plants is that they benefit from a floating transfer terminal located several hundred m offshore, where the fish is pumped from the holds of fishing vessels and sent directly to the plant by an underwater pipe.

2. Goal and Scope

2.1. LCA Goals

The intended applications of our results are: 1) to provide data and related recommendations for environmental protection in Peru in order to allow a future greening of the FMFO supply chain; and 2) to provide results of life cycle inventory (LCI), LCI analysis and life cycle impact assessment (LCIA) that can be used in LCAs of any supply chains where fishmeal or fish oil are key. The major limitations of this study are: 1) the limited number of sampled plants (one per category); 2) the usual inherent limitations of LCA when applied to fisheries¹; 3) the lack of characterisation of the impacts of certain substances released to the environment (oils, some antifouling substances, biological oxygen demand (BOD), etc.) including their odour nuisance; and 4) as usual in LCAs, impact categories and associated characterisation factors are often insufficient, subject to uncertainty and subjectivity in the weighting factors, and prone to biases and errors (Vázquez-Rowe et al., 2012; Avadí and Fréon, 2013).

2.2. Scope

The studied system consists in two major processes: 1) capturing fish at sea and delivering it to the terminal of a fishmeal plant, and 2) transforming this raw material into FMFO. Because process 1) is already fully documented (Avadí et al., 2014a,b; Fréon et al., 2014b), this work concentrates in process 2) and its sub-processes. The function of the system is the procurement of the two commodities, fishmeal and fish oil.

In order to reach our two intended applications, two different types of functional units (FUs) were used: an output-based one and a process-based one. The first type of FU is the delivery of one metric tonne (t) of each of the two commodities at the gate of the plant, using gross energy content for the allocation of impacts between those two coproducts, and considering separately three categories of commodities in the case of fishmeal: residual, FAQ and Prime or Super Prime. We do not consider any category of fish oil in the definition of the corresponding FU. These output-based FUs allow reaching our first intended application (greening the Peruvian supply chain of FMFO).

In order to reach our second intended application (providing generic results) process-based FUs were retained. They consist in the processing of 1 t of raw material entering at the floating terminal of the plant and used to produce the same three categories of fishmeal. The fact that our process-based FUs consider the raw material input rather than the outputs takes into account most of the consequences of changes in the conversion ratio according to the raw material, providing that the LCA practitioner that use this kind of FUs knows the actual value of the conversion ratio of his/her case study.

The reference flows are one t of Peruvian fish oil or Peruvian fishmeal of specified quality (out of three) for the output-based FUs. In the case of process-based FUs, the reference flow is one t of raw material as the major input of a plant aimed at producing two co-products (fish oil and fishmeal of a specified quality).

¹ The major limitations are the need for standardisation of fisheries LCA research (fisheries-specific impact categories, inventory details, normalisation references, etc.) and the weakness of some methodological assumptions, as discussed in Vázquez-Rowe et al. (2012) and Avadí and Fréon (2013).

Because our goals are mostly retrospective, accounting and descriptive ones, the retained LCI modelling framework is an attributional one. Although we address the consequences of a change of the main energy source from heavy fuel to natural gas, consequential LCA was not used because existing Peruvian data allows this comparison. The allocation approach was retained based on energy content as a physical relationship (Ayer et al., 2006).

The system boundary of the study for the output-based FUs is “from cradle to gate” and includes the extraction of the raw material (fishing), its delivery at the plant terminal, its processing and conditioning in the plant. In contrast, the boundary for the process-based FUs is “from gate to gate” (Figure 3). The following three life cycle stages of the fishmeal plants were retained: construction, use and maintenance. The factory infrastructures were considered, as well as the large storage area and the total land occupation, but the end of life (EoL) stage was ignored for the plant (not for the fishing vessels when using the output-based FUs). The main reason for the exclusion of the EoL phase was the lack of previous experience of full dismantlement in Peru. We made the assumption that EoL environmental impact is limited based on: 1) the large duration of life of the equipment (up to 40 years) allowed by an excellent maintenance; 2) the tradition in Peru to reuse equipment; 3) the results of other LCA studies of food production (Hall and Howe, 2012).

Inventory data were collected in the period 2010–2013 and encompass averaged fishery data from all the Peruvian anchoveta fleets and three fishmeal plants. These plants were numbered chronologically as Plant 1 for the traditional FAQ plant, Plant 2 for the modern steam plant producing prime fishmeal and Plant 3 for the residual plant (Table 3). The cut-off rules used for the fishing process are detailed in Fréon et al. (2014b,c). Regarding Plant 3, only a screening LCI was performed on the field. In order to allow LCIA comparisons between the three plants, this initial LCI was expanded by assigning to Plant 3 the rescaled Plant 1 LCI. Although the rescaling factors were very rough, the comparison makes sense mainly because the main LCI item (fuel use) was available.

Table 3: Major characteristics of the three sampled plants

Characteristic	Plant 1: traditional FAQ	Plant 2: Modern steam	Plant 3: Residual
Type of fishmeal produced	100% FAQ	100% Prime	100% FAQ
Type of fuel used for heating	Heavy fuel	98% gas converted in 100% by simulation in the LCA.	Heavy fuel
Number of production lines	2	3 (2 Prime fishmeal, 1 FAQ)	1
Average instantaneous processing yield (t/h)	88	114	5
Average processing yield per working hours* (t/h)	70	100	4
Average annual working hours (h)	700	1400	1900 (estimated)
Fresh fish processed (t)	48 430	155 535	9 600
Base years**	2008, <u>2009</u> , 2010	2007, 2008, <u>2009</u> , 2010	<u>2012</u>

* Taking into account daily maintenance (4 h per working day) and other delays. ** Major one underlined

The LCIA method ReCiPe v1.07 (Goedkoop et al. 2009) was used as available in the LCA software SimaPro v7.3, and LCI database ecoinvent v2.2 was used for background processes. The

egalitarian perspective of ReCiPe was retained because it is the most precautionary one (Goedkoop et al. 2009).

The major LCI datasets were provided by the major fishing companies for the fishing sub-system (details in Fréon et al., 2014c) and by a single fishing company (anonymous) regarding Plants 1 and 2. In both cases we had access to reliable data. Fossil energies, electricity mix and materials consumed by the fishing fleet or the plant, and combustion of fuels in industrial boilers were modelled specifically for Peru.

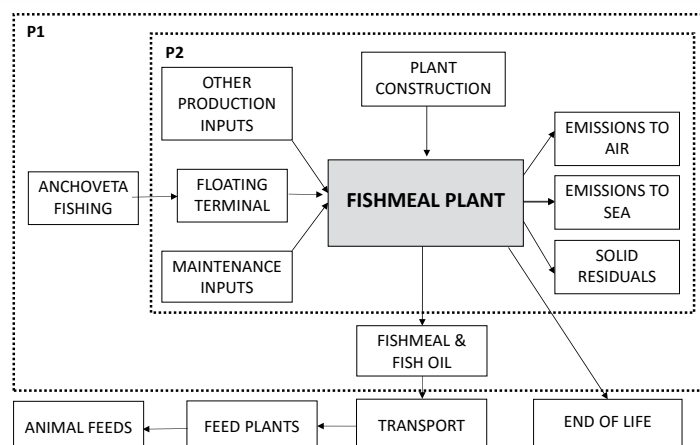


Figure 3: System boundaries according to the functional units (FUs): in P1 the FUs are the delivery one metric tonne (t) of fishmeal or fish oil at the gate of the plant; in P2 the FU is the processing of 1 t of raw material entering at the floating terminal of the plant

3. LCI

3.1. Life cycle inventories

The land occupation is quite large (e.g. > 34 000 m² for Plant 1) because, in addition to the settlement of the plant itself, the plant must have a large storage area, sometimes cemented (Plants 2 and 3) sometimes gravelled (Plant 1). The total number of items of the LCI is presented in

Table 4, whereas the major flows of material and energy in the plants are summarized in **Error! Reference source not found.**, using the value of 30 years for lifespan of the factories.

Table 4: Number of items in the LCI of Plants 1 and 2, per phases of the LCA, and corresponding number of entries in SimaPro

LCA phase	This work LCI items (n)	This work Entries in SimaPro (n)	Danish leafood LCI items (n)
Construction	258	25	0
Maintenance	100	20	0
Use	50	51	17
Total	409	90	17

3.2. LCI analysis

The annual quantities of raw material processed by the three plants are much lower than their potential processing capacity. Considering 240 potential working days at full time (that is 20 h of processing and 4 h of cleaning and preheating per day), Plants 1, 2 and 3 could have processed in

theory 422 400, 547 200 and 24 000 t per year, respectively (and the whole Peruvian industry 44.9 million tons in 2009, based on a 9 350 t per hour capacity). This means that Plants 1 and 2 used 11% and 28% of their potential full capacities, respectively, and the whole sector 13% of it, reflecting the large overcapacity of the Peruvian fishmeal industry. In contrast Plant 3 used 40% of its potential full capacity, which is a reasonable value due to the high variability in time and space of the resource. This good performance of Plant 3 is mostly due to more regular supply, both in fish residues from DHC plants and in fresh fish from IUU anchoveta. Overcapacity of the traditional FAQ and modern plants (the dominant ones) is mostly due to the race for fish when the industrial fisheries were managed by a single quota and the annual duration of the number of fishing days felt below 50 days (Fréon et al., 2008). IVQs resulted in an increase of this duration to around 150 days and in a slow decrease of the capacity of the fleets, but not of the plants. As a result, the race to fish is now replaced by the race to buy fish from the freelance industrial wooden fleet (Fréon et al., 2014a).

The large overcapacity of the plants largely increases the LCI expressed by FU, especially the construction phase (Table 3). The maintenance phase is also affected, although to a lower extent, whereas the use phase is only indirectly affected by likely lower daily processing rates, as detailed below.

Fishmeal and fish oil yield rates mostly influence the process-based FUs. Because these rates are fluctuating (especially the oil rate) according to the environmental condition experience by the anchoveta, the rates were based on average data for the period 2002–2011 for better representativeness. The resulting values were 21.3% and 4.3% for fishmeal and fish oil yield respectively. These figures are lower than other values recently reported for Peruvian and foreign FMFO industries (Péron et al., 2010), mostly because Peru produces its FMFO nearly exclusively from whole fish and because Péron's reference period was shorter.

The construction of the plants required huge quantities of infrastructure material (bricks, cement, concrete) and of metals, including those known for their high environmental impact (chromium steel and copper). When those quantities were prorated by FUs along the life cycle of the plants, they become quite low but still significant, due to the underuse of the plants (Table 3).

The use and maintenance phases of the plant required large quantities of chemical products, particularly for inside cleaning the different devices every 20 h of use. Different types of paint were used during these two phases, resulting in airborne emissions of diluents. The LCI of the use phases of the plants are dominated by energy consumption, as it is the case for the fishery use phase (Avadí et al., 2014b; Fréon et al., 2014b, 2014c). The major sources of energy for the plants themselves are fossil fuels mostly used for heating (cooking of raw material, drying of fishmeal, evaporation plant) whereas the share of electricity is low (4.7% for Plant 1, 2.5% for Plant 2 and 1.7% for Plant 3). Most of this electricity (Plant 1: 76%; Plant 2: 93%) comes from the Peruvian grid (dominated by hydroelectric generation), the rest being self-generated (Table 5).

Emissions to the ocean resulted mostly from the phase use and were dominated by the large quantities of suspended solids (mostly fish residues) and the associated Biological Oxygen Demand after five days (BOD₅). Nitrogen outputs, also linked to suspended solids, were estimated from the Danish plant.

Table 5: Abridged inventory table of fishmeal production in Peru per process-based and output-based FUs, compared with a Danish plant in the first case

Type of FU	Inputs/outputs	Main items	Unit	Plant 2	Plant 1	Plant 3	Danish plant
Process-oriented FU (1 t raw material)	Inputs	Fuel use ^a	MJ	1,498	1,913	2,406	1,523
		Electricity ^b	kWh	20.6	13.8	15.3 ^c	40.8
		Antioxidants	kg	0.17	0.25	0.10	0.08
		Concrete	L	13.7	1.97	2.54 ^c	NA

		Sodium hydroxide	kg	0.59	0.58	0.68 ^c	1.03
	Outputs	N	kg	0.35 ^d	0.35 ^d	0.35 ^d	0.35
		Suspended solids	kg	3.70	6.92	7.69 ^c	NA
		Oil and fat	kg	3.14	3.94	4.38 ^c	NA
		BOD ₅ or COD	kg	9.17 ^e	17.8 ^e	15.2 ^{ef}	0.12 ^g
Output-oriented FU (1 t fishmeal)	Additional inputs ^h	Fresh fish ⁱ	t	4.21	4.21	2.11	NA
		Fish residues ^j	t	0	0	2.75	NA
	Additional outputs ^h	Fish meal	t	1.00	1.00	1.00	NA
		Fish oil ^k	t	0.19	0.19	<0.19	NA

^a Heavy fuel oil (R500) or natural gas used for heating (excluding fuel use for self-generated electricity and fishing). ^b Excluding self-generated. ^c Estimated from Plant 1. ^d No proper LCI data, Danish data used as proxy. ^e BOD₅. ^f From Plant 1 data, rescaled by yield rate. ^g COD. ^h In addition to above inputs, that must be rescaled by fish input. ⁱ Fish caught by the industrial steel fleet (81%) and the industrial wooden fleet (19%) for Plants 1 and 2, and the small- and medium-scale fleets (100%) for Plant 3. ^j Considering a 43% inclusion of fresh fish coming from IUU landing for reduction (range 30-50%), which results in a 50:50 ratio in fresh fish and fish residue in the origin of FM given their different yields (4.21 vs 5.5). ^k Allocation factor fishmeal:fish oil (mass-weighted gross energy content): 73:27.

The comparison between our LCIs and others, beyond the fact that our inventory is much more detailed, shows quite similar values regarding the use of fossil energy, but quite different results regarding other items (**Error! Reference source not found.**). Electricity consumption from the grid is twice lower in our study and this is only partly explained by the used of self-generated electricity. The use of chemical products inventoried in the Peruvian plants is much lower than those inventoried in the Danish plant, but this is certainly due to the fact that other descaling agents are used (and inventoried) in those plant.

4. LCIA

4.1. Process-based FUs

Because the Peruvian fishmeal production is increasingly dominated by Prime and Super Prime fishmeal, results of Plant 2 will be more detailed than the results of the other plants. The dominant ReCiPe endpoints in the three Peruvian plants are by far human health and resources (not shown). As expected, most of the environmental impacts during the life span of fishmeal plants are due to the use phase. Nonetheless the construction and maintenance phases, largely ignored in other studies, contribute significantly. The average contribution of the use phase at the endpoint level is 87% in Plant 2, whereas the shares of the construction and maintenance phases are 10 and 2.5% respectively. Nonetheless, at the midpoint level, these contributions reach currently values of 10 to 40% in one or two of these two phases in (**Error! Reference source not found.**Figure 3). As a result, the remaining contribution of the use phase varies from values as low as 19 to 77% in ten midpoint impact categories of ReCiPe. This difference in the relative importance of the use phase is due to the weighting factors used in the ReCiPe midpoints.

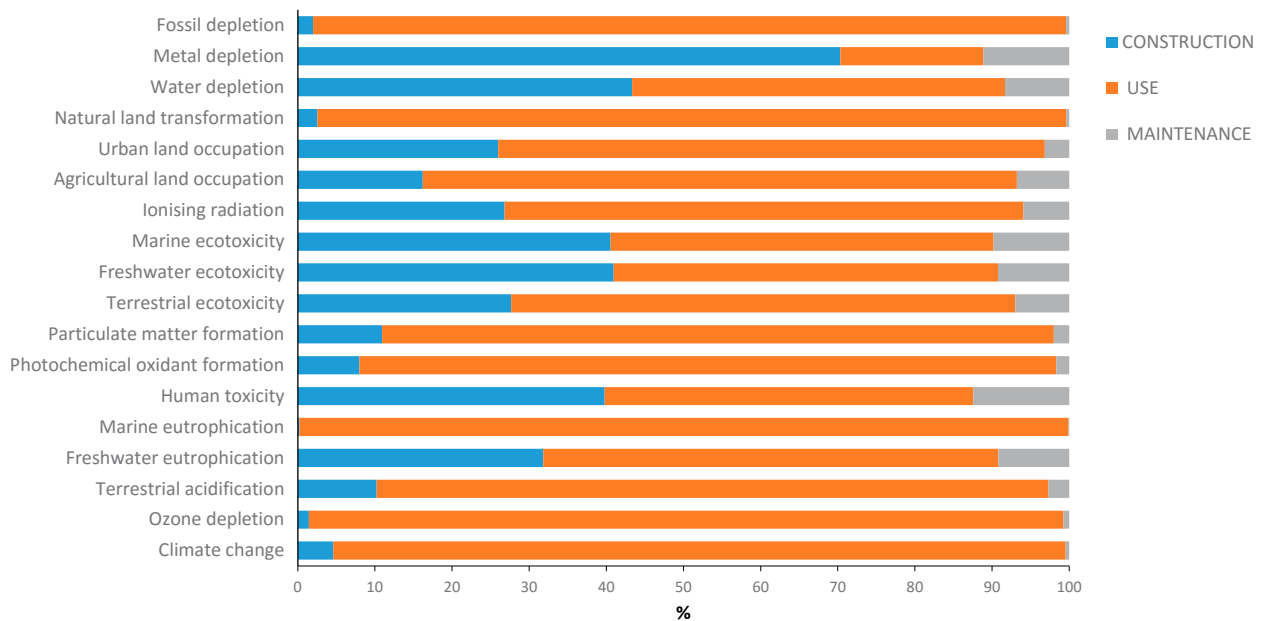


Figure 4: Fishmeal plant 2 LCA midpoint environmental impacts using the ReCiPe method. The functional unit is the processing of one t of raw material

Within the use phase, fuel use (mostly natural gas) dominates most of the midpoint impact categories, with the notable exceptions of marine eutrophication (where ocean waste dominates), freshwater eutrophication, agricultural land occupation and water depletion. The dominance of fuel use in industrial processes is a common finding (e.g. Hall and Howe, 2012). This dominance is even stronger in Plants 1 and 3 (not shown) than in Plant 2, due to the use of heavy fuel which is more impacting than natural gas. As a result, the relative importance of the use phase is higher in Plants 1 and 3 than in Plant 2.

The construction phase of Plant 2 is dominated in most midpoint categories (not shown) by the impact of concrete fabrication, the manufacturing of metals and the fabrication of unalloyed steel (cast iron) and chromium steel.

The maintenance phase is dominated by the impact of chemical products (not shown). Among them, those coming first in many midpoint impact categories are chlorine dioxide, epoxy paint and a variety of inorganic chemicals products used for cleaning. Copper also have a relatively strong impact.

The comparison of the environmental impacts of the three plants at the midpoint levels shows that Plant 2 is the cleanest in nearly all impact categories, Plant 3 the less environmental friendly, whereas Plant 1 falls in between (**Figure 5**:). The interpretation of these results is straightforward for most categories. First, Plant 2 benefits from the use of natural gas as its main energy source whereas, Plants 1 and 3 use heavy fuel. Second, Plant 2 average working hours per annum are double than those of Plant 1, which result in a lower impact per FU in the construction and maintenance phases, as explained earlier. Third, there are certainly economies of scale along the life cycle that benefit to Plant 2 and largely disadvantage Plant 3. In order to refine this comparison, we simulated the life cycle of Plant 1 using natural gas instead of heavy fuel. Because the requested changes in the capital goods are negligible, there were ignored. In all impact categories except metal depletion, the move to gas supply results in substantial or large decreases of impact. As a result, the impacts of the simulated Plant 1 falls most of the time in-between those of Plant 1 (original) and 3, or close to those of Plant 2.

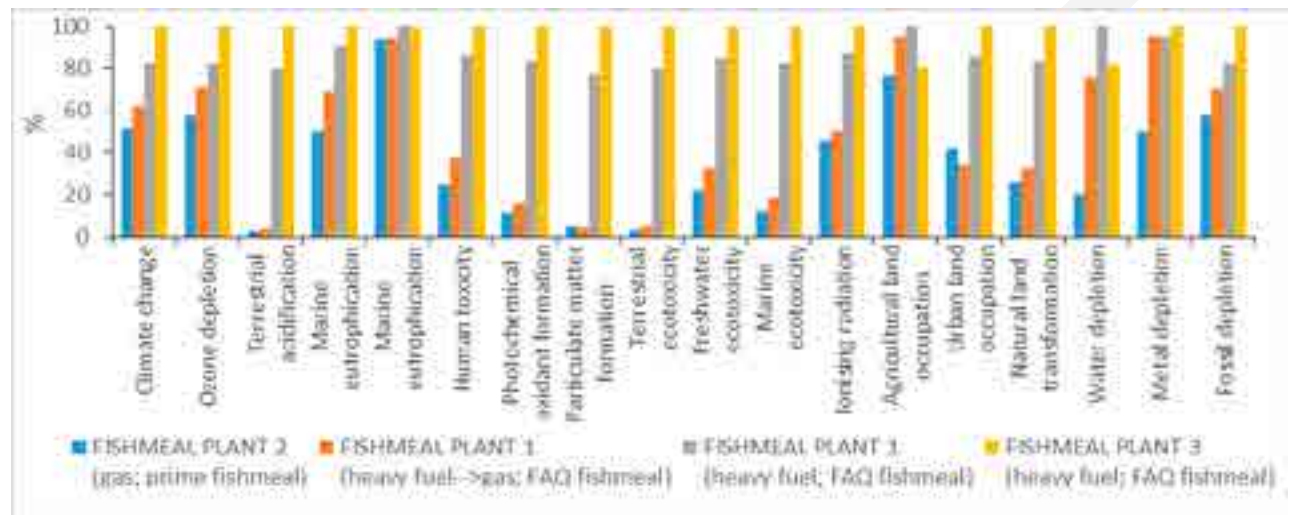


Figure 5: Comparison of LCA midpoint environmental impacts of the three fishmeal plants (with addition of a simulation of Plant 1 using natural gas) using the ReCiPe method. The functional unit is the processing of one t of raw material

The comparison between our LCIA and other work is hindered by large difference in the LCIs, mostly due to the use of different cut-off rules. The effects

of this difference in LCIs on the LCIA are evidenced by comparing Plant 2 current results with simulated results based on the same limited number of entries as the Danish LCI. The ReCiPe single score of Plant 2 is 20% higher when its LCI is detailed than when it is as coarse as the Danish one. This is partly due to the absence of the construction and maintenance phase in the latter case, but also to the lack of several items in the inventory of the use phase. It worth noting that at the midpoint level this comparison shows increases >100% in the categories human toxicity, freshwater ecotoxicity, marine ecotoxicity, urban land occupation and water depletion, and >470% in metal depletion (not shown).

Plant 2 LCIA results resulting from the simulation of a paucity of data were compared with the Danish plant results, assuming that its heat production uses natural gas. The ReCiPe single score of the Danish plant is 28% higher than the score of Plant 2 when using the same coarse LCI. This result is surprising because the two plants use similar quantities of fossil energy, the major source of impact in Plant 2. The LCIA of the Danish plant (not shown) shows that the share of electricity represents nearly 30% of the impact of its direct energy consumption, versus 9% for Plant 2. This is not only because the Danish plant use twice the amount of electricity than the Peruvian plant. It is also because the Danish electricity production is more impacting than the Peruvian one due to the relative contribution of coal-powered generation.

4.2. Output-based FUs

The share of anchoveta supply in the ReCiPe single score impact of Plant 2 life cycle is 49%. The dominant endpoints are by far human health and resources (not shown). As expected, the relative contribution of the construction and maintenance phases of the plant decreases substantially in most impact categories when considering the output-based FUs. At the midpoint level (not shown), all these contribution are lower than 15%, except for water depletion (20%), ionising radiation and human toxicity (16% each). As a result the remaining contribution of the use phase varies from 76 to 100%.

Within the use phase, the supply of raw material by the two industrial fleets dominated most of the midpoint impact categories in Plant 2 (not shown), followed by fuel use, with the notable exceptions of marine eutrophication. It is worth noting that fuel use impact also dominates in most categories of the supply of raw material (Avadí et al., 2014b; Fréon et al., 2014b, 2014c). As a result, fuel use is by far the most impacting issue in the output-based FUs.

The comparison of the relative environmental impacts of the Peruvian plants at the endpoint levels (not shown) are similar to those obtained using the process-based FU.

4.3. Towards a cleaner production

The environmental benefit of using natural gas instead of heavy fuel as energy source in Plant 1 can be quantified first by the single score of the process-based FUs that shows a decrease of 41%, and second at the midpoint level were all categories decreased by more than 24%, except metal

depletion, agricultural land occupation, marine eutrophication and ozone depletion (Comparison of LCA midpoint environmental impacts of the three fishmeal plants (with addition of a simulation of Plant 1 using natural gas) using the ReCiPe method. The functional unit is the processing of one t of raw material). Similarly, the benefit resulting from the production of Prime fishmeal instead of FAQ is obvious, although not precisely quantifiable from Comparison of LCA midpoint environmental impacts of the three fishmeal plants (with addition of a simulation of Plant 1 using natural gas) using the ReCiPe method. The functional unit is the processing of one t of raw material because, even after simulation of Plant 1 using natural gas as Plant 2, the production of these two commodities still comes from two different plants with different capacities, etc.

It is noteworthy that when a plant line works at its average processing rate, as it was mostly the case for Plants 2 and 3, but less true for Plant 1 in 2009, the fuel consumption is optimal. In contrast, when a line does not produce fishmeal but expect fish delivery for the next days, it carries on consuming fuel either for keeping warm its major equipment (cooker and drier) or for preheating them at the end of the daily 4-hour cleaning. Durand (2010) showed that when the actual daily processing rate increases from 60 to 138 t/h, fuel use decreases from 8.0 to 5.7 GJ per t of fishmeal produced. These results, although based only on 9 data points, show that the processing overcapacity combined with the increased fishing season (which generates difficulties to optimize daily processing rate), result in a substantial waste of energy.

Cleaner production and improved quality of final products can be obtained by chilling the fish on board when necessary. Oldest plants could benefit from renovation aimed at reducing energy lost by recycling the steam, eliminating steam leaking, and from increase descaling frequency to limit inhibition of heat transfer.

Finally, a better processing of blood water should result in reaching the legal maximum limits regarding the emissions of suspended solids, oil (result not shown) and BOD, which is not the case presently.

5. Recommendations and conclusion

This is, as far as we know, the first detailed LCA of fishmeal plants in the world, beyond existing screening LCAs. The LCIs of the construction and maintenance phases represented by far the heaviest work, although their corresponding environmental impacts were much lower than that of the use phase (87% of the ReCiPe single score, dominated by fuel use), which is a common finding in LCAs of industrial processes. The share of these two phases in the Peruvian case, particularly the construction one, is exacerbated by the processing overcapacity. As a result, these combined shares can reach 23 to 81% in some midpoint impact categories for Plant 2. Ideally, future studies on fishmeal and residual fishmeal plants should include not only a screening of the construction and maintenance phases, but also an improvement of the LCI of the use phase. According to our simulation, the Danish plant LCA screening, the most documented one available, is likely to have underestimated its environmental impact by more than 15% at the single score level, and by more than 100% in some midpoint impact categories.

There is room to decrease the environmental impact of this industry (use of natural gas instead of heavy fuel, reduction of overcapacity, modernisation of the oldest plants, production of higher fishmeal quality, improvement of sanitary condition, etc.). Because the use of natural gas instead of heavy fuel as the main source of energy results in large decreases of environmental impacts (Comparison of LCA midpoint environmental impacts of the three fishmeal plants (with addition of a simulation of Plant 1 using natural gas) using the ReCiPe method. The functional unit is the processing of one t of raw material), it is recommended to favour this move by extending the natural gas network all along the Peruvian coast. Presently this network covers only a fourth of the Peruvian coast line. Projects to extend this network exist but suffer from delays. Similarly, the move from the production of FAQ fishmeal to the production of Prime fishmeal,

already started, should continue to be encouraged by the legislation. These two measures are beneficial both from the environmental and economic points of view. Regarding overcapacity, if it was decreased by a factor two, the share of the construction phase would decrease by about the same amount. A final recommendation for the Peruvian industrial sector is to enforce the present policy regarding management and sanitary conditions in order to address “black fishing” and under-reporting issues, illegal and unregulated fishmeal plants in operation and the lack of compliance with environmental regulations (although recent progresses in these domains have been observed).

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117. What we feed matters: differences in life cycle greenhouse gas emissions and primary productivity requirements to sustain provision of fish meals and oils

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Abstract

In fed aquaculture production systems, provision of feed often accounts for a substantial portion of life cycle impacts up to the farm gate. Amongst common aquafeed inputs, fishmeals and fish oils (FMFO) can be some of the most impactful sources of proteins, lipids and other essential nutrients to diets. Importantly, however, FMFO are derived from many different sources, and these differences in terms of source species, ecosystem and fishery, can result in substantial differences in associated life cycle environmental and ecological impacts. In this research, we elucidate the divergent environmental and ecological impacts of a broad range of widely traded FMFO products that together represent over half of globally available FMFO.

Environmental and ecological impacts of these products were evaluated using two life cycle impact categories that are widely employed in seafood LCA research and are driven, respectively, by abiotic and biotic aspects of the FMFO production system: global warming potential (GWP in tonnes CO₂-e/tonne meal or oil) and primary production required (PPR in km² of marine ecosystem support/tonne meal or oil). Global warming potential was calculated as the sum of life cycle greenhouse gas (GHG) emissions from direct, source-specific energy inputs to both the fishing and processing stages. Primary production required was calculated following Pauly and Christensen (1995) and Parker and Tyedmers (2012), but modified to better reflect source marine ecosystem-specific characteristics as proposed by Cashion *et al.*, (2016).

Results indicate that amongst the 24 unique sources of FMFO analyzed, GWP varied by over one order of magnitude while PPR varied by over two orders of magnitude. Consistent with earlier research, GWP was driven primarily by fuel use during the fishing stage, with fuel use intensity and resulting GWP substantially influenced by the type of fishing gear used. In contrast, major sources of variance in PPR resulted from differences in source species trophic level, as well as differences in source ecosystem productivity and efficiency. Impacts of FMFO derived from by-products were often greater than when sourced from reduction fisheries reflecting typically higher fuel use and source trophic level.

Our results highlight the opportunities that exist to reduce life cycle impacts of fed culture systems through the selection of low impact FMFO and to the general importance of attending to the source-specific character of feed inputs from both marine and terrestrial origin.

Keywords: reduction fisheries, fishmeal and oil, carbon footprint, primary production required

1. Introduction

Globally, one-sixth of capture fisheries are destined for the production of fishmeal and fish oil (i.e. reduction fisheries) which are currently overwhelmingly utilized by fed aquaculture (FAO 2014a). In general, the fish targeted for reduction are small pelagics and serve as forage fish in their ecosystems for higher trophic level species (Tacon and Metian 2009). Many different sources are used globally for the production of fishmeal and fish oil (FMFO), but little concern has been given to the divergent environmental impacts of FMFO products based on the source: species, ecosystem, and fishing gear. The major impacts can be characterized broadly as biotic, impacting the populations and ecosystems from which they are harvested, and abiotic, negative changes to the immediate habitat or the broader environmental systems.

The current major concerns regarding reduction fisheries relate broadly to biotic impacts caused by their biomass being removed from ecosystems. The biotic impacts of reduction fisheries include decreased population of the target species, by-catch, and disruption of energetic flows within their source ecosystem leading to reduced food energy available for trophic levels above the target species including marine mammals and seabirds (Alder *et al* 2008; Naylor *et al* 2009; Cury *et al* 2011). While many indicators of biotic impact have been applied to seafood LCA studies, measuring the primary production required (PPR) to sustain the products is the most common (Avadí and Fréon 2013; Cashion *et al* 2016).

Abiotic impacts of FMFO production are diverse and include greenhouse gas (GHG) emissions from fuel use during the fishing stage and other activities along the supply chain including processing energy, and pollutants released from the fishing vessel including anti-foulant

paints and refrigerants. The fuel use for fishing and the processing energy have been previously found to contribute substantially to abiotic impacts of FMFO production in some settings (Pelletier and Tyedmers 2007; Pelletier et al 2009; Avadí and Fréon 2013). The aim of this study is to compare the cradle-to-gate abiotic and biotic impacts of the major FMFO products to demonstrate variance in environmental performance because of target species, source ecosystem, and gear used.

The abiotic and biotic impacts of FMFO production can be accounted for through two life cycle assessment impact categories: global warming potential and PPR. Primary production required serves as a coarse measure of the scale of dependence on the ecosystem as a human appropriation of limited photosynthetically captured energy available in these source ecosystems. The GWP is quantified through the cumulative life cycle emissions of GHGs. We aim to compare the major sources of FMFO through the use of both of these measures using a ‘cradle-to-gate’ life cycle perspective.

2. Methods

To form an overview of the major contemporary sources of FMFO, literature sources and expert opinions were sought for agreement and inter-reliability. We included all sources of FMFO that met the following criteria: i) at least an average of 100,000 tonnes of biomass of that species must be destined for reduction annually over the period of 2008-2012 (FAO 2014b); ii) general agreement among sources that this species, or a substantial portion of its landings are regularly destined for reduction over the period of 2008-2012; and iii) adequate information exists on (a) the source species, including source ecosystem, trophic level, and meal and oil yields, and (b) the fisheries, including fishing nations, gear employed, fuel use intensity (FUI), and annual landings. In addition to these reduction species, species whose processing by-products are now commonly used for FMFO production were included as well. These species are caught for DHC, but have large portions of by-products that can be and are converted into FMFO for use in other sectors, and accounted for an estimated 35% of fishmeal production in 2012 (FAO 2014a).

We used a functional unit of one tonne of meal or oil produced to compare individual FMFO products on the basis of their mass as these are the units that these commodities are typically traded in. Where the need arose to allocate resource inputs and subsequent environmental burdens among coproducts, the relative mass or energetic content of products was used depending on the stage of the production system concerned. The resulting environmental burdens were divided by the relative mass of co-products originating from DHC fisheries because of the physical relationship of mass to fuel use during the fishing stage. The environmental burdens arising from reduction fisheries and the processing stage were divided among the co-products of fishmeal and fish oil based on their gross nutritional energy content to reflect the primary use of these products. The meal energetic density was species or species-type specific (e.g. whitefish, herring type, anchovy type), whereas oil energy density was constant regardless of source (FAO 1986; Sauvans et al 2004; Parker and Tyedmers 2012).

The primary production required (PPR) to yield FMFO products was quantified following Cashion *et al.*, (2016) that refines the analytical approach originally established by Pauly and Christensen (1995; Equation 1)

$$P_x = \frac{M/C * (1/T_x)^{L-1}}{R_x}$$

Equation 1

Where P is PPR in km² per year, M is the mass of landings in tonnes, C is the ratio of wet weight biomass to carbon content of typical marine tissue (9:1 or 11.1% ; Pauly and Christensen 1995), T is transfer efficiency of ecosystem x, L is trophic level, R is the ecosystem primary production rate, expressed in tonnes carbon per km² per year, and x is the ecosystem under study. The

spatialized version of PPR was used to account for differences in source ecosystem primary production rates, and thus has greater specificity to the source ecosystem.

Ecosystem specific transfer efficiency values rather than the global average of 10% were used (Pauly and Christensen 1995) as these models demonstrate substantial variance from the global average of 10% (3.51% to 14.8% for ecosystems modeled), and these ecosystem-specific estimates were obtained from a summary of Ecopath with Ecosim models (Libralato et al 2008). Fish trophic levels were obtained from FishBase, and the same species harvested from different ecosystems was assumed to have the same trophic level value. Data on LME primary production were accessed from the Sea Around Us Project (2011). Lastly, species-specific fishmeal and oil yield rates were obtained from Cashion *et al.*, (2016).

The direct and indirect GHG emissions, or carbon footprint, of FMFO products were quantified through the summation of GHG emissions from fuel use of the fisheries stage and energy use during the processing stage of FMFO production. Greenhouse gas emissions from fossil fuel combustion also accounted for all associated upstream processes (e.g. extraction, refining, transport). The Ecoinvent 2.0 database from the Swiss Centre for Life Cycle Inventories was used for the GHG emissions of various fuel sources, and the carbon intensity of electricity were modeled based on country-specific electricity mixes (World Resources Institute 2011).

The FUI, in liters of fuel per tonne of round weight landings, for each fishery was estimated from available published sources and unpublished analyses by the authors (Parker and Tyedmers 2014). Original data sources include national and international energy analyses, fishery life cycle assessments, government and industry reports, and fishing vessel energy audits. Each fishery was matched to records of FUI by both target species and fishing gear. Where multiple reported FUI values were available for a single fishery, estimates were weighted on the basis of sample size.

Energy inputs to reduction plants were modeled based on previous studies to reflect species- and region-specific technologies where these data were available. Data regarding types and quantities of energy inputs to wet fish and invertebrate reduction processes were assembled from available public and private sources. For all sources of meal and oil for which direct reduction energy use data were not available, average values were applied per tonne of wet biomass processed based on similar reduction settings. On-board processing produces a minor amount of FMFO, and these were modeled based on previous studies of these specific FMFO products.

3. Results

Of all dedicated reduction fisheries that contribute to global FMFO availability, 18 discrete combinations of species, source ecosystem and gear met our inclusion criteria (Table 1). Together, they represent an estimated 52% of the 16.3 million tonnes of wet weight biomass landed by dedicated reduction fisheries in 2012. The majority of these species' landings originated from nine large marine ecosystems with concentrations in the North Atlantic region and the western coast of South America. All dedicated reduction fisheries studied used purse seine or pelagic trawl fishing gear, except for a bottom-trawl fishery targeting sandeel (*Ammodytes marinus*). The sources of FMFO were mainly small pelagic species that occupied a middle trophic level. However, some important sources of FMFO were derived from relatively low (e.g. Antarctic krill, *Euphausia superba*; L=2.2) and high trophic level organisms (e.g. blue whiting, *Micromesistius poutassou*; L=4.0). Furthermore, sandeel and blue whiting both inhabit benthic environments regularly, in contrast to most of the other species that occupy pelagic environments. An additional six combinations of FMFO sources from were included from three DHC fisheries with substantial by-product utilization rates targeting Alaska pollock (*Theragra chalcogramma*), Atlantic cod (*Gadus morhua*), and haddock (*Melanogrammus aeglefinus*). Thus, a total of 24 discrete sources of FMFO were compared in this study.

The average carbon footprint of assessed fisheries per tonne of fishmeal was 1.67 tonnes CO₂-e and ranged from 0.477 t CO₂-e to 5.57 t CO₂-e, while a tonne of fish oil had an average of 3.37 t

CO₂-e and ranged from 0.770 t CO₂-e to 11.9 t CO₂-e. The average PPR per tonne of fishmeal was 12.3 km² and ranged from 0.00693 km² to 167 km², while a tonne of fish oil had an average of 21.3 km² and ranged from 0.0143 km² to 356 km². In addition, three major patterns emerged when comparing the biotic and abiotic impacts of various FMFO products: i) the relative importance of fishing gear and source of processing energy on carbon footprint; ii) the effect of ecosystem transfer efficiency and trophic level on PPR; and iii) contrasting results of reduction fisheries and DHC fisheries.

The fisheries employing purse seine gear had generally lower direct fishing FUI, and resulting carbon footprints, though overlap exists with some fisheries using pelagic trawl gear of one form or another (Figure 1). Mixed gear fisheries for Atlantic cod and haddock performed the worst on this measure. The proportion of the carbon footprint attributed to the fishing stage varied substantially between sources of meal and oil. Not surprisingly, those sources with higher fuel inputs during the fishing stage have a much greater proportion of GHG emissions associated with that stage. The estimated aggregate energy to process a tonne of wet weight biomass is 1600-2200MJ. However, its associated carbon footprint varies more dramatically based on the source of thermal and electrical energy inputs and their relative proportions. The differences in amount and type of processing energy used are trivial in terms of the carbon footprint of fishmeal and oil products in all but the most fuel-efficient fisheries (<100L/tonne of fish landed).

The sources of meal and oil that performed well on the PPR measure were derived primarily from low trophic level animals (Figure 1). However, the influence of source ecosystem specific transfer efficiency and productivity also played a substantial role. For example, the Barents Sea had the lowest transfer efficiency modeled (transfer efficiency= 3.51%), as compared to the North Sea (transfer efficiency= 11.6%) and the Icelandic Shelf (transfer efficiency= 14%). Consequently, the low transfer efficiency coupled with the very high trophic level species harvested from it (Atlantic cod= 4.2, haddock= 4.1) results in very large levels of PPR for these two sources of meal and oil. Other ecosystems also had low transfer efficiencies (Humboldt current= 6.6 California current= 4.0%), however, target species from these ecosystems were of lower trophic levels resulting in a smaller PPR than would otherwise be the case.

Sources of meal and oil that had high yields generally performed well. If the inputs of energy into the fishing and processing stage were comparable to other species, the higher level of products produced reduced the relative impact per unit of meal or oil. Yields were much more variable for oils than for meals, and this aided in dividing the burdens among more products and reduced the impact of both products when oil yields were high. Thus species with high oil yields like Gulf menhaden (16% oil yields) performed particularly well, and species with low oil yields, like blue whiting with a typical oil yield of 1.9%, performed particularly poorly.

In general, FMFO from by-products of DHC fisheries perform worse on both the carbon footprint and PPR measures than did products from dedicated reduction fisheries (Figure 1). This is mainly attributable to the relatively fuel intensive gear types deployed (bottom trawls, gillnets and long-lines), and the high trophic level sources of meal and oil. The division of inputs and emissions of co-products of DHC fisheries based on their relative mass functionally sets the stage for this finding. Importantly, however, FMFO from DHC fisheries are not destined to perform poorly. This is illustrated by the performance of FMFO from Alaska Pollock, a species mainly caught with pelagic trawls in fisheries with low FUI that target a relatively low trophic level (3.5) animal. Fishmeal and oil from Alaska pollock not only performs better than the meals and oils from other DHC fisheries, it is just below the median value of all sources of FMFO assessed for both carbon footprint (median = 1.34 tonnes CO₂-e/tonne meal) and PPR (median = 0.129 km²/tonne meal). In contrast, blue whiting has a relatively high trophic level for a species largely dedicated to reduction (4.0), and is on the higher end for its carbon footprint performing worse than Alaska pollock on both measures. Thus, there are exceptions to the general finding that sources of meal and oil from DHC fisheries perform worse than dedicated reduction fisheries on these measures (Figure 1).

Table 1. Fishery characteristics and impacts per meal and oil sorted by carbon footprint of meal

Species ^a	FUI (L/tonne) (Gear) ^b	Fishing stage (%) ^c	Trophic Level	Yield (kg/tonne)		Thermal (Source)	Electricity (Country)	T (%)	PP (mg C m ⁻² day ⁻¹)	Size (km ²)
				Meal	Oil					
Anchovetta (HC)	18 (P)	33.2	2.7	240	50	1518 (NG)	74 (Peru)	6.6	876	2,619,386
Gulf Menh. (GM)	37 (P)	48.9	2.2	210	160	1486 (NG)	92 (USA)	9.7	570	1,530,387
Capelin (IS)	23 (P)	40.2	3.2	165	77	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (IS)	43 (P)	55.7	3.2	200	110	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (NS)	43 (P)	55.7	3.2	200	110	1486 (NG)	92 (Nor)	11.6	1115	690,041
A Menh. (NE)	29 (P)	42.8	2.3	240	50	1486 (NG)	92 (USA)	14.8	1536	308,544
Cal. Pil. (CC)	100 (P)	72.2	2.4	230*	180*	1486 (NG)	92 (Mex)	4	613	2,224,665
Euro. Pil. (CnC)	109 (P)	76.1	3.1	230	180	1486 (NG)	92 (Nor)	5	1196	1,120,439
Capelin (IS)	102 (P)	74.9	3.2	165	77	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (IS)	142 (T)	80.6	3.2	200	110	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (NS)	142 (T)	80.6	3.2	200	110	1486 (NG)	92 (Nor)	11.6	1115	690,041
B. Whit. (NS)	85 (P)	71.3	4.0	197	19	1486 (NG)	92 (Nor)	11.6	1115	690,041
B. Whit. (NWS)	85 (P)	71.3	4.0	197	19	1486 (NG)	92 (Nor)	3.51	491	1,109,613
Pollock (GA)	64 (T)	50.2	3.5	170*	17*	2212(FO)	0	14.2	906	1,491,252
Pollock (WBS)	64 (T)	50.2	3.5	170*	17*	2212(FO)	0	12.1	586	2,182,768
B. Whit. (NS)	111 (T)	76.4	4.0	197	19	1486 (NG)	92 (Nor)	11.6	1115	690,041
B. Whit. (NWS)	111 (T)	76.4	4.0	197	19	1486 (NG)	92 (Nor)	3.51	491	1,109,613
Sandeel (NS)	149 (B)	81.3	2.7	197	42.4	1486 (NG)	92 (Nor)	11.6	1115	690,041
Euro. Sprat (NS)	371 (T)	91.6	3.0	188	79	1486 (NG)	92 (Nor)	11.6	1115	690,041
Krill (AS)	141 (T)	72.8	2.2	160	0.8	418 (MDO)/1507 (IFO)	0	14	273	3,486,169
Cod (BS)	533 (M)	94.0	4.4	170	17	1486 (NG)	92 (Nor)	3.51	1910	396,838
Cod (IS)	533 (M)	94.0	4.4	170	17	1486 (NG)	92 (Nor)	14	551	521,237
Haddock (BS)	679 (M)	95.2	4.1	170*	17*	1486 (NG)	92 (Nor)	3.51	1910	396,838
Haddock (IS)	679 (M)	95.2	4.1	170*	17*	1486 (NG)	92 (Nor)	14	551	521,237

a. See naming convention for Figure 1. b. Fuel Use Intensity (FUI) with (P) indicating purse seine fisheries, (T) indicating pelagic trawl fisheries, (B) indicating bottom trawl fisheries, and (M) indicating mixed-gear. c. The percentage of the carbon footprint from the fishing stage. *Indicates non-species specific yields used, with Atlantic cod yield rates used for other whitefish species of haddock and Alaska pollock, and European pilchard used for California pilchard.

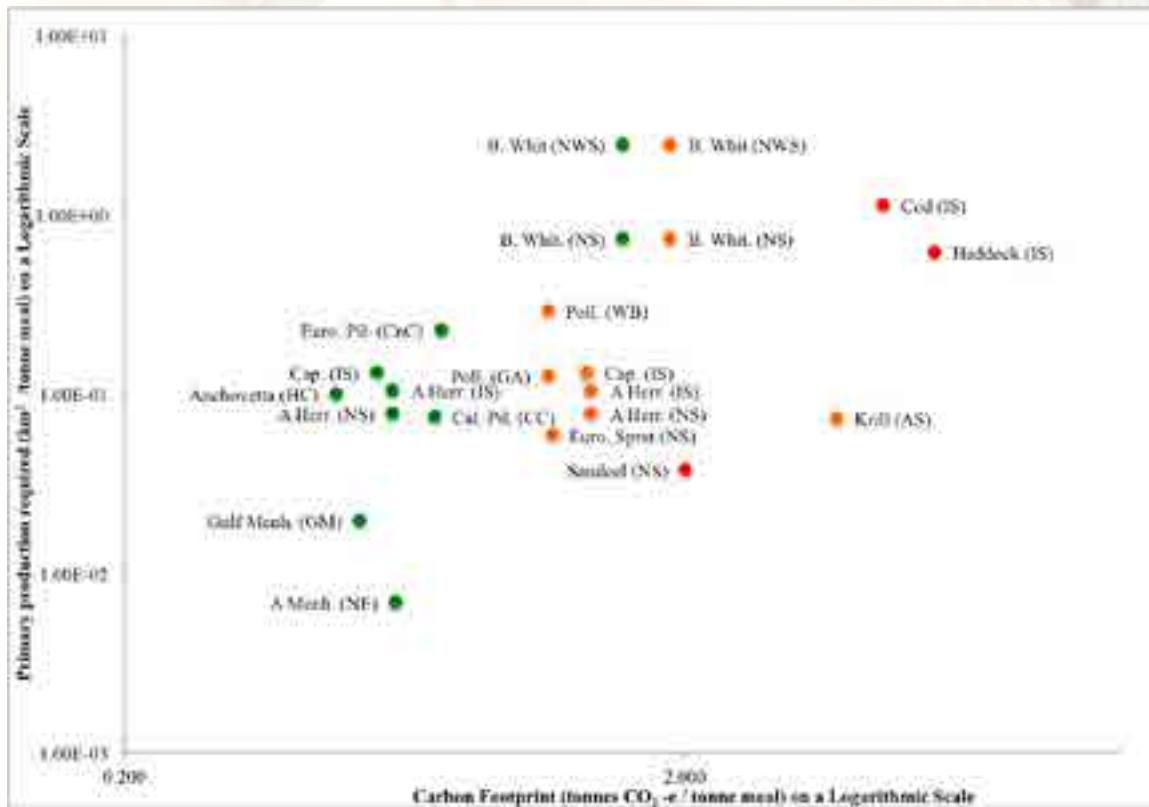


Figure 1A)

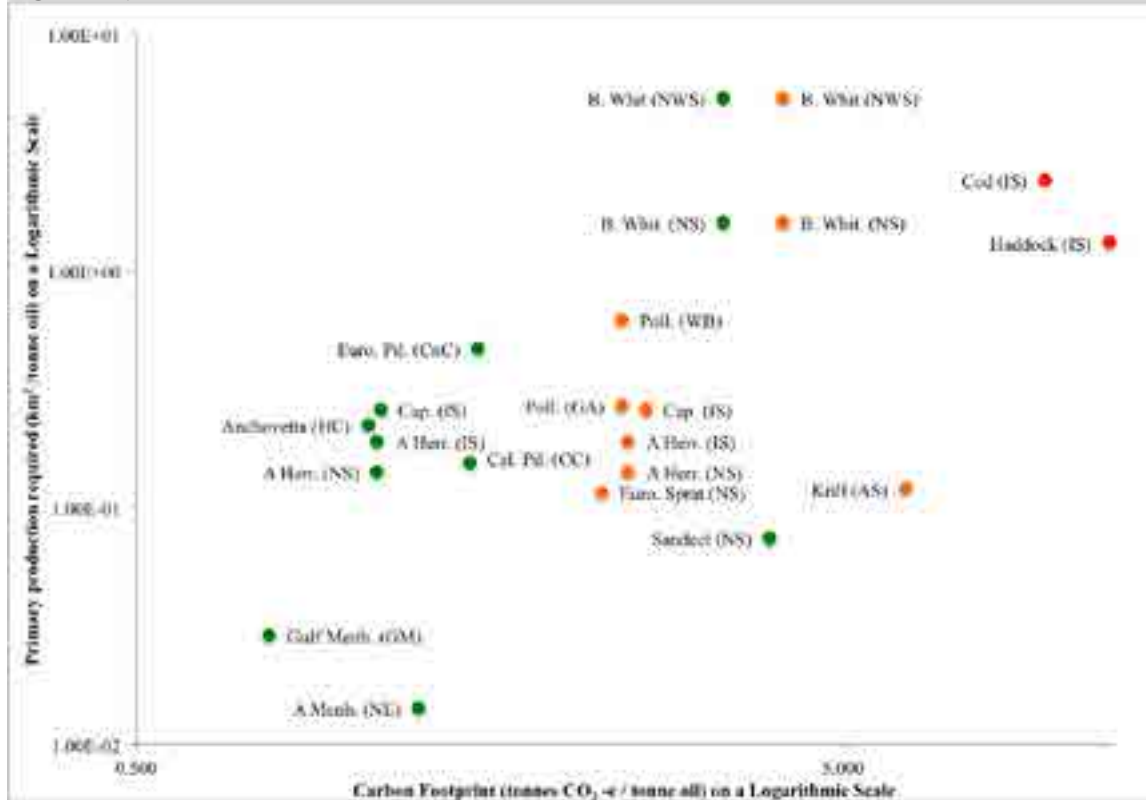


Figure 1B)

Figure 1: Carbon footprint and PPR of fishmeal (A) and fish oil (B) products on a logarithmic scale for both axes. Individual data points are listed by species with source ecosystem in brackets. Fishing method is denoted by the colours green (seine), orange (pelagic trawl), and red (mixed gear and bottom trawl). Shortened species names are: Anchovetta (*Engraulis ringens*); A Herring (*Clupea*

harengus); A Menh. (*Brevoortia tyrannus*); B. Whit. (*Micromesistius poutassou*); Cal. Pil. (*Sardinops sagax*); Cap. (*Mallotus villosus*); Cod (*Gadus morhua*); Euro. Pil. (*Sardina pilchardus*); Euro. Sprat (*Sprattus sprattus*); Gulf Menh. (*Brevoortia patronus*); Haddock (*Melanogrammus aeglefinus*); Krill (*Euphausia superba*); Poll. (*Theragra chalcogramma*); and Sandeel (*Ammodytes marinus*). Shortened ecosystem names are: Antarctic Shelf (AS); California Current (CC); Canary Current (CnC); Gulf of Alaska (GA); Gulf of Mexico (GM); Humboldt Current (HC); Icelandic Shelf (IS); Northeastern United States Continental Shelf (NE); North Sea (NS); Norwegian Sea (NWS). Two data points both from the Barents Sea (Haddock and Cod) are excluded from both figures because they lie far outside the range of the other results.

4. Discussion

Results of this research support many previous findings from LCAs of fisheries and aquaculture systems and studies of primary production required to sustain fisheries. The importance of the fishing stage to overall GHG emissions has previously been cited in many LCAs and related studies (Ziegler et al 2003; Fréon et al 2014). However, as found in a study of organic and conventional salmon feeds (Pelletier and Tyedmers 2007), the processing stage of FMFO can be a relatively large contributor to the overall life cycle of FMFO products, particularly when fishery-related FUI is low. Results of the current study affirm these findings as demonstrated through the large variance in the proportion of the carbon footprint attributed to processing. While direct fuel consumption in a fishery may be a reasonable proxy of the carbon footprint of most DHC fisheries (Parker and Tyedmers 2014), it is a less robust surrogate for reduction fisheries because of their relatively low fuel intensities and non-trivial processing-related emissions.

Prior research has hinted at the existence of substantial differences in the impacts of specific FMFOs (Pelletier and Tyedmers 2007, Pelletier et al. 2009, McGrath et al. 2015) or addressed impacts of a limited set of meals and oils (Parker and Tyedmers 2012); however, the extent to which impacts of meals and oils can vary has not been fully appreciated. Consequently, these nutritionally and economically valuable products should not be treated as environmentally equivalent or interchangeable. Indeed, given the highly divergent environmental ‘costs’ associated with many of them, feed formulators and other consumers of meals or oils seeking to produce more sustainable products should attend closely to their unique characteristics. In this context, this article echoes previous work that highlights the importance of harvesting low trophic level species from ecosystems with high transfer efficiencies for minimized biotic impact (Parker and Tyedmers 2012), and from fisheries that are fuel efficient, either because of the gear used, stock status, or species characteristics (Ziegler and Hornborg 2014; Parker and Tyedmers 2014).

5. Conclusions

Evaluating major sources of fishmeal and oil globally, using both the carbon footprint and PPR measures demonstrates substantial differences in the impacts of these products. The information presented in this article can be used to inform feed formulation decisions based on biotic and abiotic criteria for the most widely utilized FMFO products. While feed producers cannot change the source ecosystems of these fish species, they can select the products that originate from lower impact ecosystems and less carbon intensive supply chains. Hopefully, this will promote the use of less impactful ingredients in the formulation of feeds for aquaculture and other livestock sectors that wish to meaningfully address environmental sustainability concerns.

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ORAL PAPERS

DIET AND NUTRITION

II. Diet and Nutrition

169. Environmental Impacts of Diets: Development of an LCA Database to Link to Individual's Food Choices in the United States

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ABSTRACT

The recent recommendation to include sustainability considerations in the Dietary Guidelines for Americans has raised interest in the environmental impact of diets in a U.S. context. A few aggregate U.S. diet studies exist, but no work addresses the environmental impacts of individual diet choices, which is needed to accurately estimate the impacts of behavioral changes and thereby improve policy. Here we describe the development of a food environmental impact database linkable to individual food choices in the U.S., which could be used to estimate the variation in individual diet-induced cumulative energy demand and greenhouse gas emissions. Individual dietary recall data from the National Health and Nutrition Examination Survey for 2005-2010 serves as the basis for individual food choices; the more than 7000 foods consumed are linkable to 354 commodity items. To assign environmental impacts to these food commodities, we conducted a comprehensive review of LCA studies published since 2005. Impact factors along with metadata on scope and production characteristics were included in the database. The food LCA review identified 193 unique publications and 805 "entries", or unique combinations of food type and life-cycle production scenario. Twenty-eight percent of LCA entries went through the retail stage; 97% of entries considered global warming potential while only 37% included cumulative energy demand. Even with this extensive literature review, 37% of the food commodity items required proxies to link to global warming potential impacts. Data gaps were particularly noteworthy in herbs and spices, lightly processed food forms of fruits and vegetables (juices, dried), and some major cooking oils. Despite the frequent use of proxies, they accounted for only 2% of diet-level global warming potential, as the foods assigned proxy tended to be low impact and less frequently consumed foods. This cross-disciplinary research demonstrates an approach that can inform policy discussions regarding the sustainability of dietary choices, but it also points to research gaps that must be filled by the LCA community in order to improve on the available information.

Keywords: greenhouse gas emissions; cumulative energy demand; data gaps; NHANES;

1. Introduction

The 2015 U.S. Dietary Guidelines Advisory Committee recommended in their Advisory Report the inclusion of sustainability considerations in the 2015 revision of the Dietary Guidelines for Americans. The committee's recommendations were summarized in the following statement: "The major findings regarding sustainable diets were that a diet higher in plant-based foods, such as vegetables, fruits, whole grains, legumes, nuts, and seeds, and lower in calories and animal-based foods is more health promoting and is associated with less environmental impact than is the current U.S. diet." (U.S. Department of Health and Human Services, 2015) While these recommendations ultimately were not included in the 2015 Dietary Guidelines, they have kindled an interest within the U.S. in better understanding the environmental impacts of diets.

Considerable effort has been made in recent years to evaluate the environmental impact of dietary choices (Hallstrom et al., 2015; Heller et al., 2013). The bulk of this effort has evaluated aggregated (i.e., average) or stereotyped diets in European countries, with a focus on global warming impacts. Only a handful of studies have evaluated the environmental impact of diets in the U.S. (Heller and Keoleian, 2014; Marlow et al., 2009; Tom et al., 2016; Weber and Matthews, 2008). Further, if there is a desire to link dietary behaviors with both environmental and health outcomes, it is important to use the individual as the unit of analysis, rather than population average diets; population aggregates do not get cancer, the individuals within those aggregates do. In addition, individual-level data allow for more nuanced modeling of population-wide dietary change policies by linking individual-level demographics (e.g., age, gender, race-ethnicity, education, nutrition knowledge, environmental attitudes, etc.) to the behaviors of these groups and their environmental impacts. A few examples of environmental impact for individual, self-selected diets do exist in the literature (Sjörs et al., 2016; Vieux et al., 2012; Vieux et al., 2013), but none for U.S. individuals.

A major challenge in this field of research is the development of a database of environmental impacts of foods that is extensive and robust enough to represent the diversity of foods in self-selected diets. This paper describes the development of such a database that can be linked to the U.S. National Health and Nutrition Examination Survey (NHANES). We also characterize the literature review performed in its development and highlight data gaps in the existing food LCA literature.

2. Methods

Individual dietary recall data (16+ years of age, N=28,276) from the NHANES for 2005-2010 serves as the basis for individual food choices. NHANES sampling is selected to represent the U.S. population, and the dietary recall data contains reference to over 7000 food items (USDA, 2014). Many of these food items are aggregated foods (e.g., pepperoni pizza) and require recipes to assign to commodity foods that are typically represented in LCA studies. In order promote analysis at the diet level of pesticide and other residues in food commodities, the U.S. Environmental Protection Agency has developed the Food Commodities Intake Database (FCID) which links specific food items in NHANES through standardized recipes to foods in agricultural commodity form. We utilize this database to connect as-consumed food to 354 commodity farms.

Literature review

We conducted a systematic search in Web of Science and Google Scholar databases in February, 2016. Search terms included combinations of “LCA” and “life cycle” with “food”. Further refined searches targeted individual underrepresented foods. Articles and reports written in English and published in the past ten years (after 2005) that applied LCA methods to one or more food products and provided primary (i.e., not cited from elsewhere) mid-point impact assessment results were reviewed and inventoried. Peer reviewed journal articles as well as thoroughly documented reports from governmental and non-governmental organizations were considered. The literature review was limited to reports available in the public domain. Agricultural crops not expressly grown as human food (e.g., biofuels, timber, fibers) were excluded.

Data collected from the literature review were logged for each entry as indicated in Table 1. “Entries” represent unique food – production scenario combinations and multiple entries often existed from the same article.

Table 1: Specific data inventoried in literature review

Data type	Description
Food entry	
Food type	Beverages, cereals & grains, dairy, eggs, fish & seafood, fruit, legumes & nuts, meat, meat substitutes, oils & fats, sweeteners, vegetables, other
Specific food	As described in the original journal article
Food form	Fresh, canned, frozen, dried, cured or pickled, and other
Product or production specifics	Text field offering additional descriptive information about the specific entry
Compares production strategies/methods?	Yes/no field identifying whether the entry was compared against other production/distribution scenarios within the given LCA study
Country of origin	Indicates the country of production of the FOOD in question
Citation	
Year	Indicates year of document publication
Authors	Full listing of author names
Author affiliations	Listing of author affiliations as indicated in document
Source type	Peer reviewed journal, conference proceedings, government report, industry based report, NGO report, database, other
Bibliographic citation	Complete citation for document retrieval
Life cycle stages included (all yes/no fields)	
Agricultural production	Does the LCA boundary include agricultural production? In the case of wild caught seafood, this refers to the fishing stage.

LU/LUC	Land use and/or land use change included in GHGE inventory?
Transport: farm to processing	Does the study account for transport from farm gate to processing facility?
Processing	Is processing beyond farm gate commodity included?
Packaging	Does the LCA study account for packaging materials?
Transport: processing to retailer or distribution hub	Is a transport stage from the processor to retail or distribution hub included?
Retail	Are energy use and emissions associated with retailing included?
Transport: retail to home	Transport by the consumer from the point of purchase to the home?
Household storage	Is refrigeration or other storage in the home accounted for?
Prep and cooking	Preparation or cooking of the food?
Dishwashing	Dishwashing associated with consuming the food in question?
Capital goods	Production of major capital goods included at some life cycle stage?
Food waste	Was food waste included in the analysis?
Intermediary stages	Were results at intermediary stages reported?
Environmental impact values	
If available:	Cumulative energy demand; Greenhouse gas emissions; Water use; Land use; Freshwater eutrophication potential; Marine eutrophication potential; Acidification potential; Ozone depletion potential; Abiotic depletion potential; Human toxicity; Freshwater eco-toxicity; Photochemical oxidation potential; Marine eco-toxicity; Terrestrial eco-toxicity
<input type="checkbox"/> value at farm gate	
<input type="checkbox"/> value beyond farm gate	
<input type="checkbox"/> LCIA method and reporting unit	
Reported functional unit	Functional unit as reported in study
Adjustments	Adjustments made to align functional unit to "kg edible food"

For database consistency, mid-point indicator values were adjusted to a functional unit of "kg of food," with meat and fish/seafood adjusted to "kg of edible boneless weight". For most entries, this adjustment was minor (e.g., converting from tone to kg, or from L to kg using a reported density); meat and fish entries reported as live weight or carcass weight were adjusted to a boneless edible basis using conversion factors in Table 2. Given the dearth of data points for other environmental impact categories plus the strong regional dependence of eutrophication, water use and land use, further actions were limited to cumulative energy demand (CED) and greenhouse gas emissions (GHGE).

Table 2: Conversion factors to boneless weight for meats and seafood

	Live: carcass weight	Carcass: boneless weight	Live: boneless weight	Citation
Beef	1: 0.602	1:0.667	1:0.402	(USDA, 1992)
Sheep	1:0.508	1:0.658	1:0.334	(USDA, 1992)
Pork	1:0.724	1:0.729	1:0.528	(USDA, 1992)
Chicken	1:0.70	1:0.77	1:0.539	(Sanderson et al., 2010)
Farmed trout			1:0.65	(Buguen et al., 2010)
Farmed salmon			1:0.72	(Acharya, 2011)
Farmed sea bass	1:0.68	1:0.65	1:0.44	(Chef's Resources, 2016)
Farmed turbot			1:0.48	(Brown Trading Co., 2016)
Horse Mackerel			1:0.52	(FAO, 1989)
Tuna			1:0.58	(FAO, 1989)
Shrimp			1:0.57	(FAO, 1989)
Lobster			1:0.31	(Barrento et al., 2009)
Octopus			1:0.79	(FAO, 1989)

Linking to nutritional database

To link environmental impacts to the FCID, we followed a four-step process (see Table 3). First, we used data from original research on specific foods inventoried in the literature review, as described above. The mean, standard deviation, minimum and maximum values for CED and GHGE at farm gate and beyond farm gate were calculated for each specific food, and then matched to the FCID. Second, if we did not have an original research report on an FCID food, we turned to reports with previously-compiled food LCA data to supply environmental impacts (Audsley et al., 2010; Blank Consultants, 2016; Gonzalez et al., 2011; Nemerek et al., 2011; Tom et al., 2016). These resources contained data not captured in the literature review, perhaps due to non-English language reports or proprietary sources. Third, if there were still FCID foods without impact data, values from similar foods were used as proxies. Specifically, we took an average of CED and GHGE values from existing entries within a specific food grouping (e.g. berries, brassicas, brassica greens, brassica roots, citrus, fresh herbs, grains, other greens, nuts, other roots, dried spices, other tree fruit, tropical fruit) to proxy for a specific food item without data in that same grouping. Failing this approach, other proxies of foods with similar form were then assigned. Fourth, the FCID dataset includes minimally processed forms of fruits and vegetables (e.g. strawberry juice, dried apples). Where direct LCA matches were not available for these forms, we applied a mass conversion factor, gathered from nutritional databases (Bowman et al., 2013; USDA, 2015), to the base fruit or vegetable.

Because of the inconsistency in full life cycle boundary conditions across the literature review entries, cradle-to-farm gate emission factors were chosen for the vast majority of foods. The exceptions are foods within the FCID listing that require processing: flours, refined sugars, chocolate and cocoa powder, vegetable oils, roasted and instant coffee, instant and dried teas, corn milling products, wine, fruit juices, peanut butter, potato chips, soy milk, processed tomatoes. For these, life cycle impacts beyond farm gate were chosen.

Table 3: Process for assigning environmental impact data to FCID commodities

Stage	Approach to assigning environmental impact data to each specific FCID food commodity	Example
1	Mean of values from original research reports and peer-reviewed articles (literature review)	An average of 15 studies on beef for CED, 30 studies for GHGE
2	Aggregated value from a report with previously compiled impact data	Kale, from (Blank Consultants, 2016)
3A	Proxy assignment from stage 1 or 2 foods in the same group	Average of broccoli, cauliflower and cabbage for Brussels sprouts
3B	Proxy assignment from stage 1 or 2 foods of similar form	Bananas for plantain, pumpkin for winter squash, and cucumber for chayote
4	Mass conversion factor applied to base fruit/vegetable	Strawberry values converted for strawberry juice

Inedible portions

The FCID database typically uses a weight basis that excludes inedible refuse (skins, peels, pits, seeds) whereas the weight basis common in LCA studies is an as-delivered or as-purchased form (e.g., whole apples, bananas with peels). To reconcile this inconsistency, we applied conversion factors drawn from various nutritional databases (Bowman et al., 2013; USDA, 1992; USDA, 2015; USDA ERS, 2013). For example, the GHGE and CED factors for bananas on a as-purchased basis are multiplied by 1.56 (conversion factor from Appendix B of (Bowman et al., 2013)) to provide impact factors on a “pulp excluding peel” basis.

3. Results

Literature review characterization

The initial literature review resulted in 805 entries from 193 unique sources, though not all of these entries have a match in the FCID database. Figure 1 shows the distribution of entries by publication year, whereas Table 4 shows the distribution by food type, with meat, fruit, vegetables and dairy accounting for more than half of the entries. The majority of the publications inventoried were peer reviewed journal articles (69%); NGO reports, industry-based reports, government reports, and

conference proceedings represented 16%, 6%, 3%, and 2%, respectively. System boundaries varied

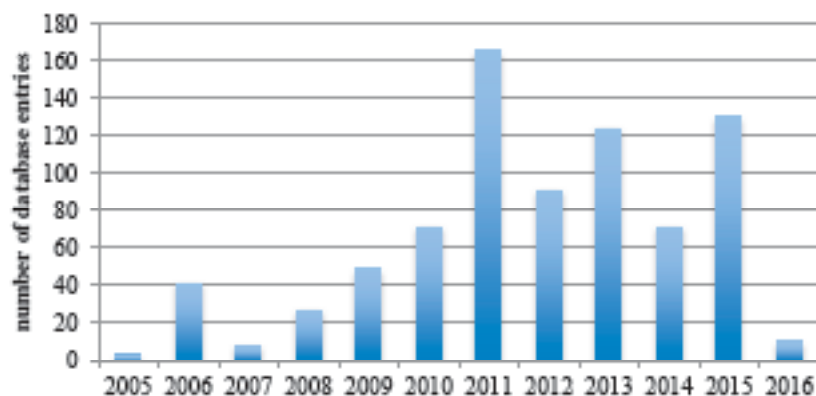


Figure 1: Distribution of food LCA literature review entries by year of publication, 2005 through February, 2016.

widely across the LCA studies inventoried: while all entries considered some form of agricultural production, 60% accounted for processing beyond farm gate, 28% followed products through to retail/regional distribution hubs, and 7% included some form of use (consumption) phase. Table 5 demonstrates that while global warming potential is evaluated for nearly all entries cataloged, the frequency of other impact categories drops off rapidly. Figure 2 shows that more than 59% of the recorded entries are based on food production in Europe (including Scandinavia and the British Isles) and that there is very limited representation from South America, Asia and Africa. Only 17% of entries were from North America.

Table 4: Characterization of literature review and linkage to the FCID, by food group.

Food groups	% of lit. review entries	# of FCID foods	% of FCID foods in group requiring proxy ²		% contribution of all foods in group to diet-level impact ¹		% of group level impact from proxies ²	
			CED	GHGE	CED	GHGE	CED	GHGE
Meats	15.9%	34	41%	18%	37.5%	61.8%	3.2%	<0.1%
Vegetables	15.8%	100	58%	54%	5.2%	3.5%	10.5%	6.8%
Fruits	14.3%	86	38%	19%	7.5%	5.5%	6.5%	1.1%
Dairy	13.2%	3	0%	0%	7.2%	9.9%	0%	0%
Fish and Seafood	9.3%	6	17%	0%	12.6%	2.8%	2.4%	<0.1%
Cereals and Grains	6.1%	29	41%	34%	6.2%	4.1%	58.5%	42.4%
Legumes and Nuts	5.5%	42	45%	29%	2.4%	1.6%	17.9%	2.3%
Eggs	3.7%	3	0%	0%	3.2%	3.3%	0%	0%
Oils and Fats	2.9%	9	56%	44%	0.3%	0.2%	1.9%	1.8%
Other	13.4%	42	26%	31%	17.8%	7.2%	0.3%	1.2%
Total diet	-	354	48%	37%	100%	100%	6.7%	2.2%

¹ Environmental impacts were summed over each food group for each individual's intake (based on NHANES 24-hour diet recall, N=28,776), and then averaged across all individuals.

² Only Stage 3 proxies (see Table 3) accounted for here.

Food database linkage characterization

The FCID database contains 354 food commodity forms for which assignments with environmental impact factors were made. Note that some of these were different forms of the same food (juices, dried), and animal based foods are divided into multiple components (meat, fat, liver, skin, meat byproducts). In Stage 1 of the linkage process (see Table 3), CED matches were made for 18.1% of the food commodities, and GHGE matches for 28.5%. An additional 20.3% and 20.6% of FCID entries (CED and GHGE, respectively) were matched in Stage 2 assignments. For 47.6% of CED

values and 37.4% of GHGE, values were assigned proxies from other foods in the database (Stage 3). The remaining 13.8% of CED and 13.6% of GHGE, primarily juices and dried fruits, were assigned values by applying mass conversion factors to the base fruit/vegetable (Stage 4). In other words, these values accounted for the concentration of raw fruit/vegetable that occurs through juicing or drying, but not the impacts of the processing stage itself.

Table 5: Frequency of environmental impact categories among entries in the food LCA literature review.

Impact assessment category	% of entries
Global Warming Potential	97%
Cumulative Energy Demand	37%
Eutrophication Potential	34%
Acidification Potential	32%
Water Use	18%
Land Use	29%
Ozone Depletion Potential	8%
Human Toxicity Potential	8%
Aquatic Toxicity Potential	5%
Terrestrial Toxicity Potential	4%

Table 4 also shows the number of FCID foods in various food groups, as well as the percentage of foods in these groups requiring Stage 3 proxy values. While the number of proxy foods is high, Table 4 also demonstrates that the contribution to total diet-level environmental impact from these proxy foods is quite low, since the foods requiring proxy tend to be low impact and less frequently consumed foods. Table 4 demonstrates, for example, that on average, the meats group contributed 62% of dietary GHGE, but less than 0.1% of this group's impact came from proxies. There were a number of proxies used in the cereals and grains group, and they accounted for 42% of the GHGE from this group. However, in terms of overall dietary impact the grains, on average, contributed only 4% of GHGE, so proxied grains account for only 1.7% of total dietary GHGE.

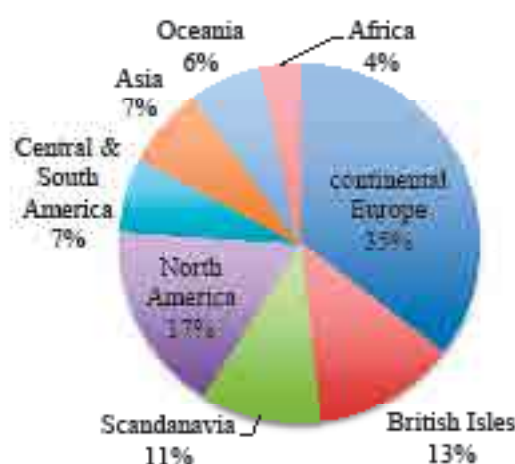


Figure 2: Distribution of Food LCA literature review entries across geographic regions.

4. Discussion

The literature review reported here confirms what other recent reviews (e.g., (Clune et al., 2016)) have found: LCA studies that might be used to link to dietary choices have increased significantly in recent years, but numerous data gaps exist for many food types. However, the current database is an important step in capturing the breadth of food LCA studies in a form that can be linked to existing individual dietary data. Table 6 suggests a number of foods for which major gaps in the publically available LCA literature exist. In addition, many foods important to evaluation of low impact diets –

nuts, legumes, meat substitutes – are poorly represented in the literature and deserve additional attention. Further, geographical representation is poor for most foods. In the approach presented here, we have addressed variability due to production practice or geography simply by averaging available values. Geographical specificity becomes increasingly important with other impact categories such as water use, eutrophication, or land use. Currently available data in these categories are extremely limited, although we hope to expand our database to water use impacts in a future iteration.

Table 6: Examples of foods with limited to no representation in the LCA literature

Grouping	Food examples
oils	almond, flax, sesame, safflower, peanut, cottonseed
grains	amaranth, buckwheat, millet, popcorn
vegetables	sweet corn, green mima, water chestnut, sweet potato, tomatillo, bamboo shoots
berries	blackberry, boysenberry, cranberry
fruit juices	nearly all, except lemon and orange juice
nuts & legumes	Brazil, cashew, chestnut, walnut, macadamia, pecan, pine, mung bean, lima bean, peanut butter, kidney beans, pinto beans
fresh & dried herbs	basil, cilantro, parsley, marjoram, dillweed, savory, peppermint, chives
spices	black & white pepper, cinnamon, coriander, dill, dried ginger, turmeric
sweeteners	corn syrup, maple syrup, honey
other	corn meal, corn bran, vinegar, seaweed, hops

5. Conclusions

The interaction between dietary choices, nutritional health and food system environmental impacts is a growing research interest. This work is part of a larger project aimed at evaluating the relationship between nutritional health and environmental impact, and exploring policy options for mitigating impact. Here, we describe a process for developing an LCA-based food environmental impact database, drawn from existing literature, which can be linked to individual dietary recall data for the U.S. In establishing linkages between existing LCA data and nutritional datasets, numerous proxy foods must be assigned, suggesting a need for the LCA community to diversify studies to less common foods. On the other hand, initial estimates suggest that the contribution of these proxy foods to total diet environmental impacts is low: 2% for GHGE (7% for CED). Additional evaluation and inter-disciplinary discussion are needed to establish priority research areas both in food LCA and nutritional science that will promote robust modeling on the influence of dietary choices.

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(Note: The full listing of LCA studies from the literature review is available from the authors upon request.)

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77. Food consumption in the EU: LCA and hotspot analysis to drive the selection of eco-innovation strategies

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ABSTRACT

Food consumption is amongst the main drivers of environmental impacts. Policies aiming at sustainable production and consumption need to identify hotspots in order to decide where and how to act to steer eco-innovation and to reduce impacts. As the impacts occur all along the food supply chain, life cycle assessment (LCA) method may support the identification of the hotspots of impacts and the comparison of alternative options for impacts' reduction. The present study aims at presenting a life cycle based method for hotspots analysis focusing on the life cycle impact assessment steps (characterisation, normalisation and weighting). The LCA results are then complemented with an analysis of hotspots of impact beyond those identified by LCA, to avoid excluding potential hotspots only because they are not fully captured by the current LCIA methods. A case study on hotspot analysis and interpretation of results is presented, building on the results of a previous study, which assessed the impact of EU food consumption based on the LCA of 17 representative food products. The results of the hotspot analyses are generally convergent in identifying the most impacting product groups (meat and dairy), whereas they are sometimes diverging in identifying the most relevant impact categories. In this case study, the identification of the most relevant impact categories is mainly influenced by the selection of the set of normalisation references compared to that of the weighting sets.

Keywords: interpretation phase, hotspot analysis, eco-innovation, food supply chain.

1. Introduction

Food consumption is amongst the main drivers of environmental impacts. Policies aiming at sustainable production and consumption need to identify hotspots in order to decide where and how to act to steer eco-innovation and to reduce impacts. As the impacts occur all along the food supply chain, life cycle assessment (LCA) method may support the identification of the hotspots of impacts and the comparison of alternative options for impacts' reduction.

According to ISO 14040 (2006), interpretation is the phase of LCA in which the findings from the inventory analysis and the impact assessment are considered together. It comprises the following elements: i) identification of the significant issues based on the results of the LCI and LCIA phases of LCA; ii) an evaluation that considers completeness, sensitivity and consistency checks; iii) conclusions, limitations, and recommendations. Besides, a recent report of UNEP (UNEP DPTIE, 2014) is giving guidance on hotspot analysis, recognizing that the identification of major sources of impact requires a systematic approach to the interpretation of the results.

The aim of the present study is twofold: firstly, it aims at presenting a life cycle based method for hotspots analysis focusing on the life cycle impact assessment steps (characterisation, normalisation and weighting). Secondly, the method presented is used to assess the environmental impacts of food consumption in EU and to identify related environmental hotspots through a detailed interpretation phase of LCA results. The present study includes: i) a hotspots analysis on characterized and normalised results, ii) the check of un-characterized elementary flows, iii) a sensitivity analysis of the results applying several LCIA methods, normalisation references and weighting factors. For all the analyses, product group contribution and impact category relevance are assessed. The results are intended to drive the selection of possible technological improvements and eco-innovation strategies aimed at reducing the overall impact of food consumption in Europe.

2. Methods

The LCA-based methodological steps for identifying hotspots in support to improvements and eco-innovation of food chains are:

- 1) Definition of a basket of products (BoP) for nutrition of EU citizens and LCA of the products selected in the basket;
- 2) Interpretation of BoP nutrition results and identification of hotspots:
 - hotspot analysis at the characterization stage, to understand the contribution of each product group to the overall BoP. Characterization is done with ILCD 1.04 (EC-JRC, 2011) and CML-IA, v 4.2 (Guinée et al. 2002);

- check of the elementary flows not characterized in the LCIA methods used, to understand which elements may be missing in the assessment of the impacts;
 - sensitivity analysis of the results using 3 sets of ILCD-compliant normalization references (Sala et al., 2015; Benini et al., 2015; Laurent et al., 2013) and 2 sets of CML-IA compliant normalization references (EU 25+3, year 2000 and World, year 2000 - Guinée et al., 2002)
 - sensitivity analysis of the results obtained complementing ILCD-compliant normalization sets with 7 ILCD-compliant weighting sets (Castellani et al., 2016, Stranddorf et al., 2005, Tuomisto et al., 2012; Bjørn and Hauschild, 2016, Pousinen and Goedkoop, 2015, Huppes et al., 2012) in addition to equal weighting of the impact categories.
- 3) Analysis of hotspots for the food sector identified in literature, including studies from different disciplines or domains, to complement LCA results;
 - 4) Identification and selection of potential improvement options (technical eco-innovation and more sustainable consumer behaviors);
 - 5) Assessment of the associated benefits through LCA studies on selected innovation scenarios.

The present paper starts from the results of step 1, illustrated in Nutamicola et al. (2016), and it focuses on the interpretation phase and hotspot analysis in support to decision making for the selection of eco-innovations (step 2). This is further complemented with the literature review on hotspots for the food chain, to include also aspects that might be neglected or out of scope in the current practice of the process-based LCA. A preliminary list of technical eco-innovations as well as consumer's behaviour towards more sustainable diets is presented as contribution to the identification of possible solutions. The assessment of the associated benefits for each of the eco-innovation selected is considered beyond the scope of the present paper and will be object of a further work package.

3. Results

The product groups included in the basket of food products that has been built to represent food consumption in EU-27 are: pig meat, beef, poultry, milk, cheese, butter, bread, sugar, sunflower oil, olive oil, potatoes, oranges, apples, mineral water, roasted coffee, beer, pre-prepared dishes. For each product group in the basket, an inventory model based on a representative product has been developed. The impact of each representative product is then multiplied by the mass of products in that product group that is consumed in one year by an average EU citizen. The paper by Nutamicola et al. (2016) illustrates the details of the process-based LCA study on the BoP nutrition for EU 27 citizens and presents the results of the characterization phase. The present work builds on those results and elaborates the interpretation phase (including also normalization and weighting). The detailed composition of the basket (in terms of product groups, representative products and associated quantities) is reported in Table 1 for completeness.

Table 1: List of product groups and representative products included in the basket

Product Group	Representative product	Per-capita consumption (kg/perc.·yr-1)
Meat	Pig meat	41.0
	Beef	13.7
	Poultry	22.9
Dairy	Milk & Cream	80.1
	Cheese	15.0
	Butter	3.6
Cereal-based	Bread	39.3
Sugar	Sugar	29.8
Oils	Sunflower oil	5.4
	Olive oil	5.3
Vegetables	Potatoes	70.1
Fruit	Oranges	17.4

Beverages	Apples	16.1
	Mineral water	105.0 (L)
	Roasted Coffee	3.5
	Beer	69.8 (L)
Pre-prepared meals	Meat based dishes	2.9

3.1 Hotspot analysis

According to the results of the characterization with ILCD and CML-IA methods, the product groups that emerge as hotspots in most of the impact categories, even if with slightly different level of contribution, are meat and dairy products and beverages (in Figure 1 the detail of results obtained with ILCD is presented). It is worthy to remember that this result is the effect of two factors: the magnitude of the impact related to the specific product group, and the quantity of its relative consumption at European level.

Agriculture is the life cycle stage contributing the most in almost all the impact categories. The majority of the contribution to impact is due to those processes related to animal feeding: "grass, at dairy farm", "grass, at beef farm", "Maize silage, at dairy farm" (source: AgriFootprint database - Blok Consultants, 2014). These processes are the major contributors to HT-cancer and HT-non cancer, TE_{air} and ME_{air}. As for the elementary flows, human toxicity impacts (both cancer and non-cancer) are dominated by the emissions of metals to water and to soil, especially chromium VI, chromium, zinc, copper and lead. These flows derive again from the agricultural process related to animal feeding.

Elementary flows of metals (especially copper and zinc, both to water and to soil) coming from the same animal feed related activities contribute also to freshwater ecotoxicity impacts, jointly with the use of pesticides (e.g. chlorpyrifos).

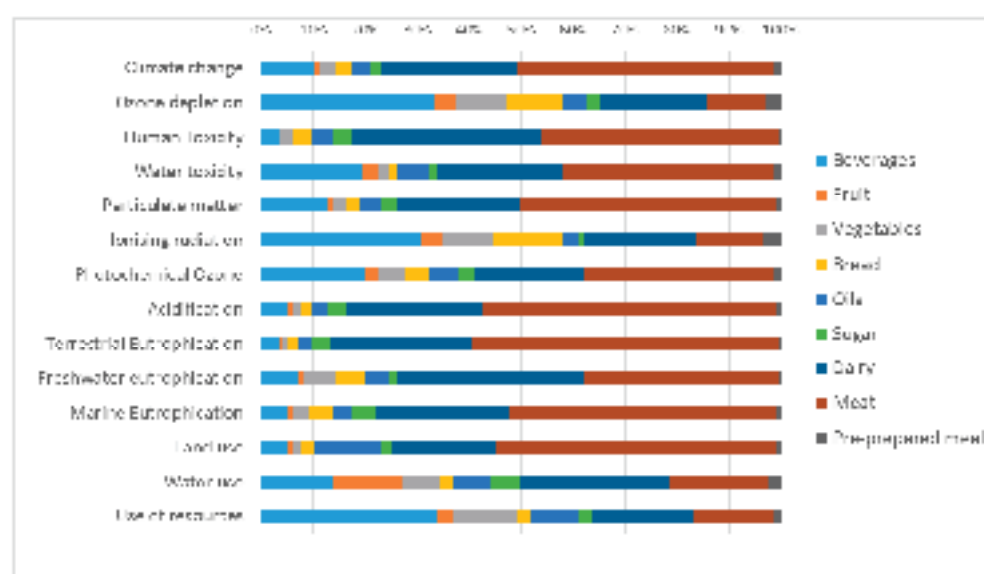


Figure 1: Product groups contribution to the impact at the characterization stage (method ILCD 1.04)

3.2 Check of flows not characterized

Apart from the total number of flows covered (543 for ILCD and 1350 for CML-IA, out of the total 1730 flows in the inventory of the basket), the main difference between the two LCIA methods is the ability to assign a characterization factor (CF) to the pesticides included in the inventory. ILCD has no CFs for 45 pesticides' emission to soil (plus 4 pesticides' emissions to air), whereas CML-IA has 173 (plus 32 pesticides' emissions to air). This may be the reason why human toxicity impacts appear as relevant in the results of ILCD characterization (and related normalization and weighting, as discussed below) and they appear as less relevant in the results of CML-IA characterization and related normalization and weighting.

3.3 Sensitivity analysis on normalization sets

Results at the normalization stage almost confirm the hotspots – in terms of product categories – identified at the characterization stage: meat and dairy products are again hotspots for all the methods and normalization sets applied, whereas other products that emerged before are considered less contributing at the normalization stage. This means that those products generate impacts in impact categories that are considered less relevant compared to the others, according to the normalization scheme adopted. For instance, beer appeared to be a hotspot for both ILCD and CML-IA at the characterization stage, whereas only CML-IA method identifies it as a hotspot at the normalization stage. This happens because the CML-IA normalization references (EU 25+3 and World) for marine aquatic toxicity – one of the impact categories for which beer is a hotspot – are relatively low compared to the ones in other LCIA methods.

Table 2: Summary of the results of the sensitivity analysis at the normalization stage comparing the relevance of impact categories according to ILCD and CML-IA methods

Impact category ILCD	EC-JRC EU27	EC-JRC Global	PROSUTIE Global	Impact category CML-IA	CML EU 25+3	CML World
Climate change	1.9%	1.1%	1.3%	Global warming (GWP100a)	3.6%	2.9%
Ozone depletion	0.0%	0.0%	0.0%	Ozone layer depletion	0.1%	0.0%
Human toxicity, cancer effects	6.2%	8.3%	2.5%	Human toxicity	2.7%	3.4%
Human toxicity, non-cancer effects	44.6%	69.2%	12.7%			
Particulate matter	2.6%	0.9%	2.1%	-		
Ionizing radiation	0.5%	1.1%	0.3%	-		
HH						
Photochemical ozone formation	1.0%	0.3%	0.3%	Photochemical ozone formation	2.1%	0.6%
Acidification	7.3%	2.86%	4.1%	Acidification	12.5%	5.7%
Terrestrial eutrophication	8.3%	4.0%	7.5%			
Freshwater eutrophication	3.4%	0.4%	4.8%	Eutrophication	6.1%	4.6%
Marine eutrophication	8.2%	2.0%	8.7%			
Freshwater ecotoxicity	5.8%	6.1%	45.0%	Freshwater aquatic ecotoxicity	15.6%	8.9%
-	-	-	-	Marine aquatic ecotoxicity	43.9%	64.7%
-	-	-	-	Terrestrial ecotoxicity	10.2%	7.0%
Land use	2.3%	0.0%	0.4%	-		
Water resource depletion	6.2%	3.3%	10.0%	-		
				Abiotic depletion	0.1%	0.3%
Mineral, fossil & ren resource depletion	1.8%	0.4%	0.4%	Abiotic depletion (fossil fuels)	3.1%	1.9%
	100%	100%	100%		100%	100%

Table 2 presents a summary of the results of the sensitivity analysis at the normalization stage comparing the relevance of impact categories according to ILCD and CML-IA methods. Results refer to the entire BoP. For each impact category of the two methods, values are expressed as percentage of the normalized value per citizen for that impact category with respect to the total normalized impact per citizen.

3.4 Sensitivity analysis on weighting sets

The contribution of specific products to the overall impacts associated to the BoP nutrition is very similar when using several ILCD-compliant weighting sets in combination with normalization references EU27. In fact, for all the weighting sets tested pork meat and beef meat are the highest contributor to the impacts (varying between 15% and 22%). According to all sets, cheese is the 3rd largest contributor (14% on average), whereas milk and poultry meat are most frequently identified as 4th and 5th largest contributors (roughly 8% each). On average, these product groups account for roughly 70% of the total impacts. Apples, olive oil and pre-prepared meals are those contributing the less, followed by mineral water and oranges. On average, their cumulative contribution is around 5%.

Conversely, the identification of the most relevant impact categories contributing to the overall impact of the entire BoP nutrition presents huge discrepancies when applying different weighting sets (Table 3). This is due to the intrinsic differences among the weighting sets, which build on different perspectives and attribute different weights to the impact categories.

The impact category "Human toxicity, non-cancer effects" results to be the highest contributor to the overall impacts in 6 of the 9 sets applied, ranging from 32% to 53% of the total impact. The three other methods do not have any weighting factor for this impact category, as the concept of planetary boundary on which they build upon does not encompass impact categories focused on human health only.

The discrepancies in the results suggest that it is controversial to identify the impact categories contributing the most to the overall impact, as different perspectives lead to different results.

Moreover, it is relatively straightforward to identify the product groups impacting the most when keeping constant normalization references although varying weighting sets.

Additional tests done applying the same set of weighting factors to results normalized with other sets of normalization factors seem to suggest normalization as the most sensitive choice if compared to the selection of a specific weighting set among those considered in this work. However, this result may change in case weighting methods building on different approaches are added to the ones included in this work. Nevertheless, the ranking of products is not significantly affected by changing normalization reference (e.g. from ILCD EU27 to ILCD Global – Laurent et al. 2013). This might be due to few impact categories driving the results regardless of the normalization and weighting methods selected, or to the fact that all impact categories are strongly correlated.

The review on hotspots identified in sectorial study available in the literature helped to identify the following additional environmental impacts associated to food production:

- Natural resource depletion (mainly in the agricultural stage) and the alteration of global biogeochemical cycles of N and P - used as fertilizers in agriculture - (e.g. Ainsley and Wilkinson, 2014; Leip et al 2014 and Smill, 2002). Those impacts are only partially captured by LCA: e.g., they are not assessed in the resource depletion impact category and only partially accounted for eutrophication (which in the case of BoP nutrition is mainly driven by the emissions of nutrients in sewage from human excretion).
- Land use and land use change impacts (Meier et al, 2014) on soil quality and biodiversity as well as reduction of natural ecosystems for food and feed cultivation. Both impact on soil quality and biodiversity are still under refinement in LCA (e.g. Vidal-Legaz et al 2016 for soil quality, Curran et al 2016 for biodiversity).
- Biodiversity loss due to the use of pesticides, for which currently LCA is providing potential impacts mainly for freshwater ecosystems. In fact, concerns such those related to decline of pollinators due to agriculture intensification are not captured so far in LCA (Crema et al 2016).

- Food waste along the whole food production chain as relevant source of impacts (Kruyer, 1995, Lundqvist et al., 2008, WRAP 2013), which need further methodological development for being properly modelled in LCA (Corrado et al 2016).

4. Discussion

The case study on food discussed in the present paper highlights some methodological issues to be considered in the interpretation phase and hotspot analysis in support to decision making (Castellani et al, 2016b).

The application of a precautionary principle may lead to add the different hotspots stemming from the applied methods, compiling a list of warnings to be further investigated. For instance, the inconsistency of ILCD and CML-IA results about the relevance of the impact category Human toxicity appears to be driven mainly by the fact that CML-IA method does not assign CPs to more than 170 emissions of pesticides, out of which about 120 are instead covered by ILCD. In such cases, the analyst should not derive from the sensitivity analysis that human toxicity is not considered a hotspot by the CML-IA method, but only recognize that it can be underestimated in comparison to ILCD.

The discrepancies in the results suggest that it is very controversial to identify on which impact category to act, whereas it could be relatively straightforward to identify the product categories impacting the most. This, however, may create problems when setting policy target aiming at hitting the most important sources of impact.

Besides, as within ILCD impact assessment method (similarly to other LCA methods) not all the environmental impacts are covered (for instance, biotic resources, indoor pollution, noise and erosion are not modelled), hotspots should be complemented with those coming from other sources and assessment methods. Therefore, a “beyond LCA” perspective is essential for a comprehensive evaluation of drivers of impacts, and further refinement of the method may be needed to ensure systematic cross-fertilization between LCA and other specific domains.

4. Conclusions

The results of the hotspot analyses and sensitivity analyses run on the case study highlight that food products deriving from animal husbandry and related feeding (e.g. meat and dairy products) are the ones with the highest contribution to the overall impact of the BoP. The most relevant impacts as assessed by ILCD and CML methods are related to human toxicity, water resource depletion, freshwater and marine ecotoxicity.

Other impacts not fully captured in LCA are: food loss happening throughout the whole chain; the alteration of biogeochemical cycles of N and P, e.g. used as fertilizers in agriculture; and impacts due to land use on biodiversity. The key areas for eco-innovation and improvement of the food sector resulting from the analysis, are: i) strategies for promoting diet shift from current EU average habits to diets with less meat content; ii) actions to reduce the amount of feed needed per animal (e.g. improving efficiency of feed by adding synthetic amino acids) ; iii) reduction of food losses throughout the whole supply chain and reuse of food waste (e.g. as a source of nutrients, to close the loop); iv) agricultural techniques to optimize nutrient cycles (e.g. avoidance of oversupply).

For each area, a set of improvements/innovation needs to be assessed through LCA to quantify potential benefits and highlight possible burden shifting among life cycle stages and/or impact categories.

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81. Proposing the Nutrient Density Unit as the Functional Unit in LCAs of Foods

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Abstract

This paper aims to establish a comprehensive measure of nutritional quality that can be used as a functional unit in LCAs (Life Cycle Assessments) of food products. The 'function' of food is to deliver required nutrients to the human body, not only filling (volume) or fuel (kcal). In order to find an appropriate unit, we analysed and evaluated existing nutrient density scores, quantifying the amounts of essential nutrients per gram or kcal. The conclusion is that these models have a common structure: they include macronutrients to encourage (protein, dietary fibre, and sometimes essential fatty acids) and macronutrients to limit (salt, added sugar and saturated fatty acids), generally accomplished with one or more micronutrients (vitamins, minerals). An index with macronutrients per kcal is sufficient to predict the total nutrient density. This resulted in the formulation of the Nutrient Density Unit (NDU), reflecting total protein, essential fatty acids, and dietary fibre, per energy density. These elements of the NDU correlate significantly with all other essential nutrients. The limiting macronutrients were left out because they can result in a negative unit, which cannot be addressed in LCAs. We propose the Nutrient Density Unit –at least for solid foods–, since it reflects the food's 'function' of supplying the essential macronutrients within human metabolic energy needs.

Introduction

The objective of this paper is to establish a comprehensive measure of nutritional quality that can be used as a functional unit in LCAs (Life Cycle Assessments) of food products. Functional unit is defined as 'quantified performance of a product system for use as a reference unit in a life cycle assessment study.' This unit is necessary to ensure comparability of LCA results.[1] To date, the most common functional unit for calculating the sustainability of food products has been 100g.[2] According to Heller et al.[3], however, it is desirable to quantitatively link the environmental impact of dietary patterns to their nutritional function. Nutritional quality indexes are a potential approach, but need further refining. In fact, the 'function' of food is to deliver required nutrients and energy to the human body, not mass, volume, or portions. Other studies have concluded that neither 100g nor 100kcal are the best

functional units to identify which foods to include in a sustainable diet [4,5] The function of food is more than filling (g or ml) or fueling (kcal), it also provides essential nutrients.

Method

We analysed and evaluated existing nutrient density scores in order to extract a simplified and dimensionless nutrient density unit. Dietary Guidelines define nutrient-dense foods as those 'that provide substantial amounts of vitamins and minerals (micronutrients) and relatively few calories.' [6] Nutrient density indexes summarize the densities of individual macro- and micronutrients compared to their recommended values (dietary reference intakes), sometimes related to metabolic energy density (kcal/g). The majority of the currently available indexes were described by Drewnowski and Fulgoni [7,8], and tested by Kennedy et al. [9] and Drewnowski et al. [10]

Greenhouse Gas Emission (GHGE) data are calculated according the method Agri-footprint developed to determine LCA for agricultural products. [11] The life cycle includes everything from the acquisition of raw materials and natural resources to final disposal, including food waste, and includes an estimate of the energy used to cook and prepare the food. The data are based on food products harvested or processed in the Netherlands in 2011, as well as imported products; in some cases, the data were updated in 2014. Most of the data is available in the Agri-Footprint database (www.agri-footprint.com). [12]

Main results

When making an environmental impact study, it is reasonable to assume that the primary function of food consumption is to supply nutrition; the ideal functional unit for comparing LCA of diets should therefore be nutritionally based. [3] A number of previous papers have used metabolic energy content, protein content, or both as their basis for comparison. In order to move forward, we need a comprehensive measure of nutritional quality coupled with LCA. A starting point might be the nutrient density approaches that have been used in nutrition science, such as the weighted nutrient density scores (WNDS) developed by Arsenault et al. [13]. Heller et al. [3] mentioned various approaches WNDS, (Alternative) Healthy Eating Index (HEI), Nutrition Rich Food Index (NRF), and Overall Nutritional Quality Index (ONQI). Drewnowski and Fulgoni [7,8,10] evaluated the Nutritional Quality Index (NQI), ratio of recommended to restricted (RRR), calories-for-nutrient (CFN), naturally nutrient rich (NNR), and NRF indexes. Kennedy et al. [9] compared three of those nutrient density indexes and found that rating individual food items does not depend on the method of classification.

When we compare these approaches, we find that all nutrient density models have a clear common structure (Table 1). [14] According to nutrient scientists, the most relevant nutrients to encourage are the essential macronutrients. Essential nutrients are those which are vital for health, but cannot be produced in the human body. Essential nutrients includes macronutrients; proteins (12 amino acids), essential fatty acids (EFA: linoleic acid and omega-3 fatty acids), and complex carbohydrates embedded in dietary fibres, micronutrients (all vitamins and some minerals), and water.

Table 1: Common structure of nutrient density models. [14]

Macronutrients to encourage	Macronutrients to limit
protein dietary fibre (essential fatty acids)	saturated fat (added) sugar sodium
Micronutrients to encourage	
vitamin C, iron, calcium (and other vitamins and minerals)	

Though many nutrient density indexes are available [3,7-10], the Nutrient Rich Foods index (NRF) was chosen. This is because it is frequently used, it fits in the common structure, and it has been validated and extensively compared with other methods. [15] Other authors have proposed comparable nutrient density scores. [5,13,16] Because NRF does not reflect energy density (kcal/g), we developed a dimensionless NRF index named Sustainable Nutrient Rich Foods (SNRF-index, Eq.1), with energy density in the denominator. SNRF is preferable for LCA purposes, because it is an index that predicts to a certain extent the sustainability of food products. [14]

Eq. 1: Sustainable Nutrient Rich Foods index (SNRF)

$$= \frac{\left(\frac{g \text{ EFA}}{12.4 \text{ g}} - \frac{g \text{ SFA}}{20 \text{ g}}\right) + \left(\frac{g \text{ plant protein}}{50 \text{ g}} - \frac{g \text{ sodium}}{2.4 \text{ g}}\right) + \left(\frac{g \text{ fibre}}{25 \text{ g}} - \frac{g \text{ added sugars}}{50 \text{ g}}\right)}{3 \times \left(\frac{\text{kcal energy}}{2000 \text{ kcal}}\right)}$$

The numbers in the denominators of Eq1. are the recommended values or dietary reference intakes(DRI). The DRIs of protein, dietary fibre, sodium, added sugar, and saturated fat are based on the US Food and Drug Administration's recommendations [10, 17], in line with the US Institute of Medicine [18] and the Nordic Nutrition Recommendations. [19] The 12.4 g reference intake of EFA is based on the Institute of

Medicine's DRI of EFA [18], as the sum of the recommendations for the specific fatty acids linoleic acid (5 energy%), and omega-3 fatty acids (0.6 energy%).[20]

In order to convert the index into a functional unit, the negative, non-functional 'nutrients to limit' were excluded. These are saturated fatty acids, sodium and added sugars. This results in the Nutrient Density Unit (NDU; Eq.2). Higher NDU represents a higher nutritional functionality, based on delivery of protein, essential fatty acids, and dietary fibre per kcal. The NDU is based on the nutrient content of products per 100g product. Eq.2 differs from Eq.1 in another aspect; instead of plant protein, it includes total protein. Although products with plant proteins are lower in environmental impact than animal proteins [14], both sources contribute to the total dietary reference intake of protein. For products without calories or essential nutrients, it is necessary to choose a virtual value: Their NDU is set at 0.01.

$$\text{Eq. 2: Nutrient Density Unit} = \frac{\left(\frac{\text{g essential fatty acids}}{12.4 \text{ g}}\right) + \left(\frac{\text{g protein}}{50 \text{ g}}\right) + \left(\frac{\text{g fibre}}{25 \text{ g}}\right)}{3 \times \left(\frac{\text{kcal energy}}{2000 \text{ kcal}}\right)}$$

The application of the proposed NDU is illustrated with the example of protein-supplying products. Table 2 compares the climate impact of six protein-supplying food products, based on different functional units. In this example, the NDU ranks the product from low to high: milk (0.99), eggs (1.48), nuts (1.52), pork (2.05), salmon (2.31), and pulses (2.87). Greenhouse gas emissions per NDU are lowest for pulses and highest for pork. Although pulses have higher GHGEs per 10 g protein than eggs, they have lower GHGEs per NDU. Table 2 illustrates that the impacts of milk (NDU=1), nuts and salmon depend strongly on the functional unit chosen.

Table 2: Greenhouse gas emissions (g CO₂-eq) of six protein rich food products measured with five different functional units: 100g, portion, 100kcal, g protein, and the Nutrient Density Unit.

Product	portion size	NDU	Greenhouse gas emissions (g CO ₂ -eq)				
			per 100g	per portion	per 100kcal	per 10g protein	per NDU
Pulses, brown beans, canned	75	2.87	358	188	225	352	87
Milk, semi-skimmed	358	0.99	108	270	285	318	110
Nuts, mixed, salted	25	1.52	228	57	36	102	150
Egg, chicken, boiled	50	1.48	282	141	207	228	190
Salmon, aquaculture, prepared	138	2.31	485	631	226	192	210
Pork, lean, 5-14% fat	108	2.05	709	709	449	336	345

This example is based on protein rich products, but the NDU can be applied to all types of food products. Fruits and vegetables are not essential for delivering protein, fats or carbohydrates. Their function is, especially, as a source of micronutrients, such as vitamins and minerals. Micronutrients are not included in the NDU, but expected to be embedded in the matrix of the essential macronutrients (e.g. vitamins are concentrated in the fibre-rich peel of the apple). Table 3 gives an overview of the NDUs of some product groups, with examples. Based on this table, the included food groups can be ranked by NDU from low to high: fats, grains, cheeses, breads, fruits, soy products, and vegetables. We can conclude that vegetables and fruits have a high NDU, resulting in relatively low GHGEs per NDU. By considering nutrient values related to energy density and not energy density as such, the unit focusses on the supply of essential nutrients, including vitamins and minerals, to the body.

Table 3: NDU and GHGE (g CO₂-eq/NDU) for several product groups and products.

Group	Product	NDU	g CO ₂ -eq/100g	g CO ₂ -eq/NDU
Grains/potatoes	Potatoes	0.85	63	74
	Pasta	0.74	274	371
	Rice, white	0.42	220	518
	Rice, wholegrain	0.68	220	321
Bread	Bread, wheat	1.28	90	70
	Bread, wholegrain	1.60	90	56
Vegetables	Endive	4.16	187	45
	Beetroot	2.74	43	16
	Cauliflower	3.58	129	36
	Cucumber	1.85	203	110
	Pepper, green, sweet	4.83	203	42
	Cabbage, green	4.54	97	21
	Lettuce	4.72	68	14
	Green beans	3.42	120	35
	Spinach	4.52	446	99
	Tomato	2.74	203	74
	Onion	2.16	37	17
	Carrots	2.48	43	17
Fruits	Apple	0.87	35	40
	Strawberry	1.33	342	257
	Berry, red	4.83	35	7
	Pear	0.95	35	37
Oils & fats	Butter	0.12	1,297	10,624
	Oil, sunflower	0.75	271	363
	Oil, olive	0.47	271	574
Soy products	Tofu	3.30	450	137

	Tempeh	3.59	450	126
Cheese	Cheese 48+	0.97	952	978
	Cheese 30+	1.51	883	585

Discussion

Smedman et al.[21] already indicated a relation between GHGEs and nutrient density. They calculated a nutrient density index based on the NNR score,[7] called the Nutrient Density to Climate Impact index, but they only applied the index to drinks. They suggested that future studies should consider both GHGEs and the nutrient density of foods and beverages. Their index was a good attempt to relate GHGEs to nutrient density, but, Scarborough & Rayner[22] questioned its objectivity. Additionally, they did not define functional units based on nutrient density. They suggested that using a functional unit involving only GHGEs per kilogram of a food item might lead to the conclusion that vegetable alternatives are always better than those of animal origin, but as we saw in Table 2 this is not always the case. Milk appears the exception.

The examples used in Table 2 are protein-rich products of animal and plant origin. The choice of origin could influence the quality of the protein. Balance studies have shown clear and predictable differences between protein sources in terms of digestibility, biological value and consequent net protein utilization, with values ranging from near-perfect utilization for animal proteins, to much lower values for some plant-based proteins.[23] Eq.2 uses a DRI for protein of 50g, but this value is debatable. The median protein requirement is 0.66 g/kg per day, the value set by the Institute of Medicine as dietary reference intake, which is 50 grams for an average adult of 75 kg.[24] WHO experts have concluded that protein source (animal, vegetable or mixed protein) does not have a significant influence on this requirement.[22] This implies that for human adults, net protein utilization values for diets of most sources are similar, but much lower than would be predicted.[25] According to the WHO, 0.83 g/kg per day protein (62g per adult) would be expected to meet the requirements of most (97.5%) of the healthy adult population. Although some plant-based diets might be harder to digest, thus increasing the protein requirement, the values identified for adult essential amino acid intakes indicated that all likely diets would be adequate.[23]

Other authors have suggested considering the full nutrient contribution of different animal and plant foods, instead of an index based on a selection of nutrients.[15,21] Nevertheless, Drewnowski et al. showed that limiting the number of nutrients in an index is

not problematic: models based on 9 positive nutrients provided health-related rankings similar to those based on 23. [8] Arsenault et al. [13] demonstrated that it is possible to predict the ability of a given food to meet - or fail to meet - a large number of nutrient recommendations on just a handful of key nutrients, including protein, fibre, saturated fat, sodium, and added sugar. Unfortunately, Arsenault et al.'s predictive formula is not useful as a functional unit, because it is very complex, does not include energy density, and could result in negative values. A functional unit with a negative value is not applicable. We further limited the number of nutrients to three. In order to demonstrate that the three macronutrients of the NDU are representative for micronutrients, namely vitamins and minerals, we calculated the correlations between the NDU of 394 products and the selected macronutrients. Table 4 shows that most micronutrients have high significant Pearson correlations with the elements of the NDU equation; total protein, dietary fibre or EFA. This is not the case for copper (weak with protein), and vitamin C, although vitamin C correlates strongly with the complete NDU. Although Table 4 suggests that NDU covers the density of most micronutrients, further research of this proposition is needed. It would be desirable to run sensitivity analyses of the results to the NDU mode of calculation, in line with the findings of Arsenault et al. [13]

Table 4: Pearson correlations between micronutrients and NDU, protein, dietary fibre and EFA content of 394 food products.

Nutrient	NDU	Plant protein	Total protein	EFA	Dietary fibre
NDU	1	.037	.136**	-.136**	.238**
Energy density (kcal)	-.354**	.339**	.187**	.797**	.097
Calcium (mg)	-.032	-.024	.376**	-.044	-.052
Folic Acid equivalent (µg)	.212**	.120*	.056	-.001	.135**
Phosphor (mg)	.022	.275**	.781**	.115*	.103*
Iron (mg)	.092	.233**	.382**	.055	.197**
Iodine (µg)	.025	-.054	.299**	.049	-.085
Potassium (mg)	.222**	.436**	.426**	.002	.588**
Copper (mg)	.016	.172**	.108*	.090	.050
Magnesium (mg)	.032	.783**	.345**	.373**	.478**
Niacin (mg)	.147**	.152**	.578**	.142**	-.036
Retinol act equivalent (µg)	.015	-.070	.091	.006	-.069
Selenium (µg)	.116*	.035	.386**	.201**	-.102
vitamin B1 (mg)	.104*	.124*	.268**	-.014	.187**
vitamin B12 (µg)	.064	-.110*	.216**	-.026	-.119*
vitamin B2 (mg)	.025	-.065	.201**	.000	-.084
vitamin B6 (mg)	.126*	-.038	.289**	.002	-.075

vitamin C (mg)	.217**	-.068	-.221**	-.139**	.096
vitamin D (μ g)	-.004	-.140**	.170**	.278**	-.155**
vitamin E (mg)	-.107*	.104	-.093	.651**	.001
Zinc (mg)	.010	.058	.336**	.004	-.016

** Correlation is significant at the 0.01 level.

* Correlation is significant at the 0.05 level.

Most of the mentioned nutrient density indexes are designed to give negative values. These negative values indicate that not all food and beverages are beneficial for health. Due to this, they cannot be applied as functional units in LCA studies. This raises the issue of anti-function or negative function that cannot be addressed in LCA. However, this issue needs to be investigated further. We solve this problem by disregarding salt, sugar and saturated fat from the initial nutrient density index.

The NDU has one mathematical disadvantage: It cannot be applied to products without calories, for example, zero-calorie soft drinks, tea, or water. It is arbitrary to set a constant value of 0.01 for all foods without calories. Other drinks with almost no essential nutrients also have a very low NDU of 0.01: juices, soft drinks, wine and beer. This raises the question whether the NDU is suitable for drinks. It is possible that this unit does not address the function of foods in delivering water to the body. This should be considered in future research, when discussing functional units.

Conclusions

The functional unit in LCAs of foods can be based on nutrient density, related to energy density instead of mass. We successfully applied nutrient density indexes (NRF and SNRF) as the basis of such a unit. We propose the Nutrient Density Unit, since it reflects the food's 'function' of supplying the essential macronutrients within human energy needs.

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194. Life Cycle Assessment of the U.S. Food Supply Chain Relative to Dietary Choices

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ABSTRACT

A tiered hybrid input-output based life cycle assessment was conducted to contrast the potential environmental impacts of food systems associated with current U.S. consumption patterns compared to recommended USDA food consumption patterns. Comprehensive environmental impacts of individual food groups were modeled with SimaPro[®] 8 as the computational platform and assessed with the characteristics of food consumption and losses of an average U.S. household based on USDA ERS loss-adjusted food availability data. Adoption of the USDA dietary guidelines leads to an increase in GHG emissions and other environmental impact categories. This is largely attributable to supply chain activities and losses at retail and consumer levels. Electricity use and refrigerant loss at retail along with electricity consumption of home refrigeration and cooking appliances contribute the most to the results. The recommended dietary reductions in red meat, poultry, grains, eggs, fats/oils, and sweeteners consumption and associated losses decrease GHG emissions, but this is offset by increases in vegetables, fruit/juices, milk/dairy, and fish/seafood consumption and emissions. This outcome highlights the importance of incorporating environmental considerations through the entire life cycle of food systems, especially at retail and consumer phases, into dietary guidelines. It also suggests that efforts to reduce perishable food loss through improved preservation, reduced transit times, and possibly local, seasonal production are increasingly important if there is a shift toward recommended food consumption patterns.

Keywords: environmental impact, hybrid LCA, greenhouse gas emission, dietary guideline

1. Introduction

Food security and sustainability has been described in many ways by many groups, including researchers, policy makers, producers, manufacturing companies, retailers, and consumers. The general principle of sustainability has been defined as using resources at rates that do not exceed the capacity of Earth to replace these resources (Godfray et al., 2010). Agricultural sustainability focuses more narrowly on using a set of practices to produce abundant food without depleting the resources or polluting the environment (Earles, 2005). The broad topic of sustainability is complex and involves a multidisciplinary approach to address the many scientific factors involved. Regardless of the definition, it is clear that a prevailing theme is maximizing food production efficiency and availability and minimizing waste and loss of valuable commodities. A better understanding of the food supply chain and associated impacts as a function of consumption patterns is crucial for efforts to engage in environmentally responsible actions and behaviors.

A number of life cycle assessment (LCA) studies associated with food consumption and loss have been reported (Duchin, 2005; Egilmez et al., 2014; Heller and Keoleian, 2015; Heller et

al., 2013; Jones and Kammen, 2011; Venkat, 2011; Virtanen et al., 2011). However, there are some deficiencies in detailed environmental impact assessment across the entire life cycle and differences in the methodological approaches. This study expands the boundary conditions and presents comprehensive environmental impact profiles of the cradle-to-grave food supply chain with an in-depth inventory analyses of retail and consumer phases. The objective of this study is to quantify the potential environmental impacts of food systems associated with current U.S. consumption patterns compared to the recommended USDA food patterns (FP) (USDA and USDHHS, 2010).

2. Methods

We employed a tiered hybrid economic input-output (I/O) based approach to estimate potential environmental impacts at food commodity group level using a SimaPro[®] 8 as the computational platform. We modeled each food group at sectoral scale and linked process-based models for the post production supply chain.

The functional unit for the LCA is the estimated cumulative amount of food consumption including losses in kilograms per average U.S. household per year over the entire supply chain across all commodity groups. The scope of the assessment is from the cradle to the grave. The system boundary comprises production, processing and packaging of food, transport through distribution networks, and storage at retail, consumption and disposal of food and waste. The resources used at the consumer phase include: transportation for shopping trips, home refrigeration, food preparation, and dishwashing. For environmental impact assessment we used the TRACI 2.1 impact framework which has U.S. specific characterization factors for the Comprehensive Environmental Data Archive (Bare et al., 2006; Suh, 2005). Figure 1 illustrates the food system model and associated environmental impacts for each supply chain stage and displays the percentage of food loss on a mass percentage basis.

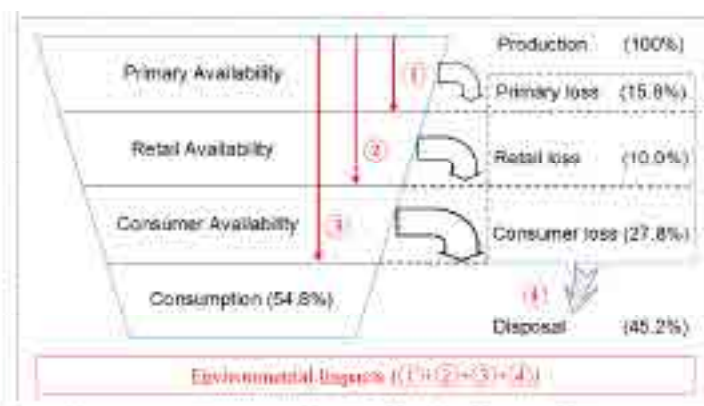


Figure 1: Schematic of food supply chain systems in U.S. and environmental impacts associated with food consumption and losses from production to disposal.

3. Life Cycle Inventory

We adopted the U.S. Department of Agriculture (USDA) Economic Research Service (ERS) loss-adjusted food availability (LAFA) database as the basis for defining both consumption and food loss (including edible, inedible, avoidable and unavoidable losses) (Buzby et al., 2014; USDA, 2015). As Figure 2 presents, the total food consumption and losses aggregate to 1,040 kg and 860 kg per household per year, respectively over the whole life cycle of each

food group for current consumption patterns; thus cumulative losses represent 45% of annual food production available to an average household. The aggregated totals increase to 1,380 kg of projected food consumption and 1,230 kg losses per household per year for the scenario of households adopting the USDA 2000 calorie per day dietary guidelines assuming the same fractional loss rates for each food category.

Retail stores consume significant energy and resources that contribute to environmental impacts. The distinct impact streams during the retail stage are electricity for store operations (overhead) and refrigeration system, loss of refrigerants due to leakage, natural gas consumption, and water usage. An economic allocation for each food group was estimated using household expenditure data (U.S. BLS, 2012). Each refrigerated food group was allocated a share of refrigerated space and a share of total grocery space to account for the refrigeration and overhead burdens, respectively. Each non-refrigerated food group was allocated a share of total grocery space to account for the overhead burdens (includes air-conditioning burden). The home refrigeration attributable to each refrigerated food group was calculated based on same allocation method as supermarket refrigeration.

Food preparation appliances including range with oven, microwave, dishwasher, etc. consumes substantial amount of energy. For most of the cooking appliances, we adapted the allocation scheme based on the fraction of expenditure per household rather than disaggregating the energy consumption of cooking appliances to each food group because of the large variation in cooking methods. Natural gas and water usage for food preparation is not included in this analysis because it is relatively small.

We modeled disposal of wasted food and packaging materials using the ecoinvent database (ecoinvent® v2.2 modified with U.S. electricity). Wastewater treatment unit process was chosen for liquid waste disposal, and for the disposal of packaging materials, we adopted the "plastic mixtures with 15.3% water to sanitary landfill" unit process.

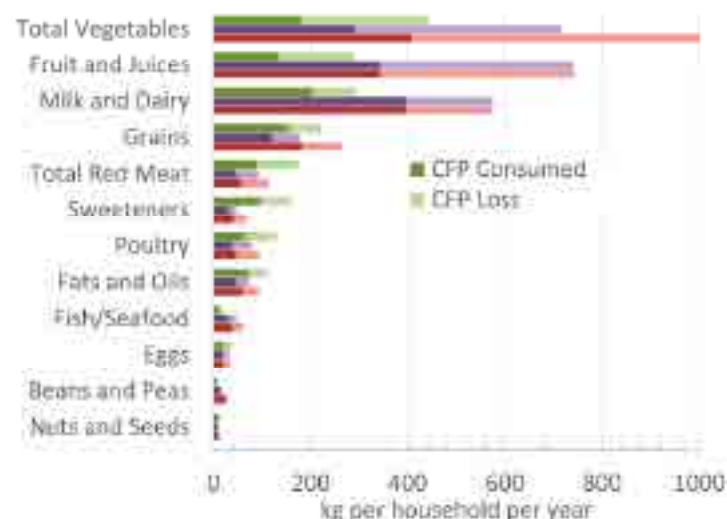


Figure 2: Comparison of food consumption and loss per U.S. household associated with current consumption patterns (CFP) versus USDA recommended consumption patterns (FP2000/2600).

4. Results

Figure 3 presents GHG emissions as a function of total food consumption and loss for each food group associated with current consumption patterns versus USDA recommendations. Current consumption patterns contribute 8.8 tons of carbon dioxide equivalents (CO_{2e})

emissions per household (2.5 residents) per year. This corresponds to 9.7 kg CO₂e capita⁻¹ day⁻¹. It increases to 11.4 tons CO₂e emissions per household per year associated with food system for the scenario in which an average household follows USDA recommendations which corresponds to 12.5 kg CO₂e capita⁻¹ day⁻¹. Vegetables, fruit/juices, milk/dairy, and fish/seafood groups are major contributors toward increased GHG emissions, mainly due to greater recommended intake and associated losses. For ozone depletion, vegetables are the key contributor followed by the fruit/juices and total red meat for current consumption. Fruit/juices are the largest contributor to ozone depletion under the USDA recommended FP, driven by refrigerant loss at retail. The recommended FP scenario leads to increases most for of the impact categories except for smog and eutrophication. The vegetables, fruit/juices, milk/dairy, and fish/seafood groups show a significant increase for all impact categories, whereas the total red meat, poultry, fats/oils, and sweeteners show a significant decrease for all of the impact categories when USDA FP recommendations were modeled, driven by quantitative changes in production and consumption of the food groups. Under the current consumption patterns, vegetables are the highest impact driver for ozone depletion, carcinogens, non-carcinogens, and ecotoxicity impact categories. Farming activities and waste disposal are the two primary processes causing these outcomes. Total red meat is the principal contributor to global warming, respiratory effects, acidification, and smog as well as fossil fuel depletion categories. Animal farming and animal production activities are the major processes driving these impacts. For eutrophication potential, eggs group is the principal contributor followed by poultry and milk/dairy groups. Egg production and poultry meat production are the two major drivers. For carcinogenic impact, heavy metals emissions associated with the landfill disposal of food waste drive the impact due to the sharing of burdens from all municipal waste disposed in landfills, as we adopted a generic landfill model from ecoinvent. The emissions from the electricity supply chain and combustion of fossil fuels are the next intensive contributors. For ecotoxicity impacts, heavy metals emissions associated with fertilizers and waste disposal process are the major contributors. Based on the results of this study, overall environmental impacts of the food system is predicted to be greater if U.S. citizens were to adhere to the USDA recommended dietary guidelines with the largest increases observed among the vegetables, fruit/juices, milk/dairy, and fish/seafood groups.

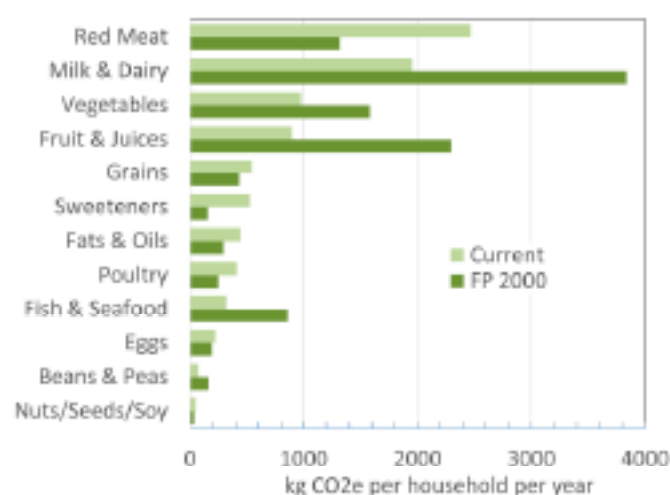


Figure 3: GHG emissions per household associated with current consumption versus USDA recommended consumption.

5. Discussion

While many previous studies have concluded changing dietary patterns to reduce consumption of animal products and increase consumption of plant foods decreases the potential environmental impacts, this study, based on U.S. diet choices, demonstrates conflicting results. A recent study by Tom et al. (2015) presented a similar result as this study despite the fact that the methodological approach was different. In this study, impacts at retail and consumption played a vital role leading to derive the different results. It clearly reveals the importance of modeling assumptions such as choice of allocation procedures and decisions to include or exclude some aspects of the supply chain that influence on the study results.

These household level results should not be extrapolated to national level because underlying assumptions of the I/O model and LCA would be violated. For instance, complete adoption of USDA dietary guidelines would correspond to a decline of approximately 40% in domestic beef production (Boucher et al., 2013), which would have significant impacts across multiple sectors. National scale modeling should be based on general equilibrium modeling of the economy to project the potentially large shifts in sector activity and thus changes in environmental impacts.

6. Conclusions

Under current consumption patterns, the total red meat group is the single largest contributor to GHG emissions, representing 27.8%. Based on the USDA recommended FP and assuming the same fractional loss rates for each food category as in the current consumption patterns, the fruit/juices (20%) and milk/dairy (33.6%) groups become the two major contributors to GHG emissions. Consequently, a shift in dietary patterns towards those recommended in the USDA dietary guidelines does not lead to GHG emissions reduction, but rather results in an increase of 28% with the consideration of food consumption and (unchanged) losses at each supply chain.

Food systems at the primary level (to the farm-gate) have higher environmental impacts under current consumption patterns – associated with relatively higher impacts of animal products, but the post-farm food supply chain, particularly at the retail and consumer levels alter those results. This outcome highlights the importance of incorporating environmental considerations through the entire life cycle of food production and consumption. We also call attention to the fact that accumulated reductions of food losses across the entire supply chain will have significant environmental benefits.

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46. Following nutrition recommendations is not enough to reduce the climate impact of food consumption

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ABSTRACT

It is commonly claimed that climate impact could be reduced by following nutrient recommendations. However, several kinds of diets can be derived from nutrition recommendations. Do they all cause lower climate impact than the current average diet? How would dietary change actually affect the nutritional quality of diet? Is it enough to follow national nutrition recommendations? The paper presents a comparison of the climate impact and dietary quality of the current Finnish diet with alternative diets harmonious with National Nutrition Recommendations (NNE). Also, corresponding land use changes in Finland will be presented. Diets studied were current (C), mixed (M), vegan (V) and plant-based diet favouring domesticity (D), each divided into sub-alternatives with varied amount of foods. Data for the current diet were from food consumption studies. In C2 the amount of sugar, wheat and alcohol were increased based on national statistics, because consumption studies underestimate their consumption. In M1, specific numerical advice in NNR were applied as such, e.g. 500 g red meat per week and 500 g vegetables and fruits per day, and e.g. the structure of consumption of vegetable and fruit was kept as in C1. Other diets apply advanced responsibility. The IPCC method was applied to climate impact. An assessment of land use change was based on average yield rates and food demand changes. Dietary systems included domestic and imported foods, preparation phases, freight and emissions from land use change in Finland. Environmental data were from our previous studies, Bioinvent and from the literature. Depletion of soil organic matter (SOM) in Finnish fields was added separately, as it was not included in previous studies. The nutrition quality of diets was assessed against NNR and based on the Finnish Food Composition Database. The diets following NNE did not necessarily reduce the climate impact of diet. V diets had the lowest impact on climate, followed by most of the D diets. None of the diets fulfil NNE completely. From the alternatives, V1 and V2 had most shortcomings. However, current diet was much worse. Findings indicate that nutrition and climate recommendations should be more integrated to meet a need for dietary change from a point of view of sustainability.

Keywords: Diet, life cycle assessment, nutritional quality, environmental impact, dietary change

1. Introduction

In 2014, the Finnish Ministry of Agriculture and Forestry wanted to look at the potential of dietary change to reduce the climate impact of the Finnish agricultural and food sector, while also taking nutrition into account. It was well known that many of animal-based food products produced in Finland or abroad have a higher impact on climate than plant-based products considered on a product and mass or volume basis at the farm gate. Globally, the livestock sector is responsible for 7.1 GtCO₂ per annum (Gerber et al. 2013), which is about 15 per cent of all human-driven greenhouse gas (GHG) emissions. The whole food sector is responsible for 20–30 per cent of the emissions. In Finland, the entire food chain accounts for 14 per cent of total climate impact, of which the share of meat production is 25 per cent and dairy 20 per cent (Virtanen et al. 2011). The role of animal production is thus obviously crucial in dietary change, and it is also recognized in the literature (Hedemus 2013, Reynolds et al. 2014, Tillman and Clark 2014). However, there are also other food product groups having a high climate impact assessed on a mass basis or as a part of a meal, such as greenhouse vegetables (Saarinen et al. 2011). Different food products have different nutrient composition and a different role in the entire diet, so that several types of food products are needed to offer a service called nutrition, and form a diet. In addition, the volume of a product in diet also affects the significance of the product from the point of view of climate impact. Thus, only consideration of a diet ensures observing a whole picture of eating.

National nutritional recommendations are tools for affecting diet at a general level. Awareness of environmental impacts related to food has already had an impact on national nutrition recommendations. For example, the most recent Finnish National Nutrition Recommendations (National Nutrition Council, 2014) contains an insight into food-related environmental issues and generalized recommendations. However, precise recommendations, for example recommendations to restrict the consumption of red meat, are predicated on health impacts, but are also strengthened by the high climate impact of meat.

It is commonly claimed that climate impact will be reduced by following nutrient recommendations (e.g. Scarborough et al. 2012). However, several kinds of diets can be derived from

nutrition recommendations (Macdiarmid 2013). Do they all cause a lower climate impact than the current average diet? Is it enough to follow NNR? There were also other concerns. How big a role does domestic land use change play in potential dietary change? Does change to plant-based diets increase the export of climate impact, and how, in turn, does favouring domestic products in the diet affect the results? And finally, how would dietary change affect the nutritional quality of diet? According to a review by Payne et al. (2016), there are so far not enough reliable studies to draw fair conclusions of the connections between nutritional quality and climate impacts, but Payne et al. (2016) also noticed, that low impact diets often contain only small amounts of salt and saturated fatty acids (safa), but do contain plenty of sugar and consequently low amounts of minerals, vitamins and trace elements.

This paper presents a comparison of the climate impact and dietary quality of the current Finnish diet with alternative diets harmonious with the Finnish NNR (National Nutrition Council 2014). Also corresponding land use changes in Finland will be presented.

2. Methods

Life cycle approach (ISO 2006a, 2006b) was applied to an assessment of climate impact. Several dietary alternatives were compared; current (C), mixed (M), vegan (V) and plant-based diet favouring domesticity (D), each divided into sub-alternatives with varied amount of foods (Table 1). Dietary systems included domestic and imported food items, preparation phases, freight and emissions from land use change in Finland (i.e. only related to domestic foods).

The average current diet (C1) was composed as weighted means of age group-specific diets and then multiplied by the number of inhabitants in Finland. Food consumption data for C1 were from the food consumption studies for different age groups (Hellidén et al. 2013, Hoppu et al. 2008, Kyttälä et al. 2008). Alternative diets (M, V, D) were composed based on NNR for adults and adjusted for other age groups based on average energy demand and the recommended division of energy intake from carbohydrates, protein and dietary fats for age groups (National Nutrition Council 2014) and age group-specific nutrition recommendations (National Nutrition Council 2010, Haanonen et al. 2004). In C2, the amount of sugar, wheat and alcohol were increased based on national statistics (Tykkä 2012), because consumption studies tend to underestimate the consumption of these foods. In M1, specific pieces of numerical advice in NNR were applied as such, e.g. 500 g red meat per week for adults and 500 g vegetables and fruits per day, and e.g. the pattern of consumption of vegetable and fruit was kept as in C1. This, however, is in a sense a deviation from NNR, because NNR recommends to increase the intake of roots, pulses and legumes. We came to our decision, since increasing the consumption of vegetables has been very slow so far, and the diversity of vegetables consumed in Finland is low. Other diets apply advanced responsibility based on our own expert knowledge. For example, all unnecessary foods from a nutritional point of view were removed from M2, and the amount of red meat was adjusted to close to the recommendation of the IARC (max 50 g/d on a population level). V1 in turn includes diverse plant-based protein sources, and the intake of greenhouse tomatoes and cucumber was restricted to the current consumption rate (C1), while V2 was lean on soya and the structure of the vegetable intake was as in C1 (meaning that the intake of all greenhouse vegetables increased compared to C1). In D1, protein sources were diverse and as in C1, but restricted not only to include beef, pork and poultry. D2 included beef and dairy products as in C1, but not pork or poultry. D3 included pork as in C1, but not beef, dairy products or poultry. D4 included poultry as in C1, but not beef, dairy products or pork. D5 included an increased amount of fish, and not beef, dairy products, pork or poultry. In every D diet, the intake of greenhouse tomatoes and cucumber was restricted to be as C1. Other product groups were adjusted to meet the energy division requirement between carbohydrates, protein and dietary fats. The total energy content of diets differed slightly between diets.

The division between domestic and imported food was based on national statistics on equity ratio (Tykkä 2012), and thus it does not reflect the real consumption of domestic and imported foods, but rather the potential consumption of domestic food. In addition, several assumptions and generalizations had to be made, as statistics were more aggregated than food categorization in diets.

GWP values (emission factors) for foods were taken from our previous studies (e.g. Saarinen et al. 2011, Saarinen et al. 2012, Urva et al. 2012, Silvenius et al. 2013, Silvenius et al. 2015), the

Emission database and the literature. In several cases, GWP values for food products were estimates based on several scientific articles, or GWP value for a similar kind of product were used. We did not dig into details of methodological issues of literature based data in this short run study, but in our own previous studies a methodology of IPCC (2006) has been followed in terms of characterization factors and emission sources in agriculture. Depletion of soil organic matter (SOM) in Finnish fields was added separately, as it was not included in the previous studies. The amount for depletion of SOM was based on Heikkinen et al. (2013), and the land use demand of different products was assessed based on average yields and consumption rates in Finland (Tilke 2013 and unpublished data from our previous studies). An assessment of land use change was based on average yield rates in Finland from years 1998–2013 (Tilke 2013) and changes in food consumption associated with diets.

Food preparation was taken into account based on average consumption of prepared food types (Hellén et al. 2013), estimated division of food consumption to home-made food, ready-to-eat food and catering (Hoppu 2008, Vainisalo 2008), energy consumption (Saxinen et al. 2011, Kauppinen et al. 2010) for food preparation and an emission factor for the Finnish electricity grid. Emissions from transportation of imports was assessed as a lump sum by using a 2,000 km as distance and an emission factor for cargo ships.

Results were initially presented as the climate impact of total food consumption in Finland ($1000 \text{ t a CO}_2 \text{ eq y}^{-1}$) to be used for internal purposes of agricultural and food policy, but they were also converted into the form of the climate impact of the average diet per person and day ($\text{kg CO}_2 \text{ eq day}^{-1} \text{ person}^{-1}$) so as to be more comparable with other studies.

Nutritional quality was assessed against NNR (National Nutrition Council 2014) and based on the Finnish Food Composition Database Fineli®. Intake of energy, carbohydrates, protein, dietary fatty acids (pufa (poly-unsaturated fatty acids), mufa (mono-unsaturated fatty acids), and sfa (saturated fatty acids)), fibre, nine dietary minerals and thirteen vitamins were considered separately for each age group.

Table 1: Amounts of foods in diets (g per day and person). C1= current, based on food consumption studies; C2=as C1 but corrected based on national statistics; M1=basic mixed, based on NNR; M2=mixed but no "junk foods", based on NNR; V1= diverse vegan based on NNR; V2=soy based vegan based on NNR; D1=plant-based, favouring domesticity, based on NNR; D2=as D1 but supplemented by beef and milk; D3= as D1 but supplemented by pork; D4= as D1 but supplemented by poultry; D5= as D1 but supplemented by fish.

g per day	C1	C2	M1	M2	V1	V2	D1	D2	D3	D4	D5
Vegetables	63	63	131	380	185	130	185	185	185	185	185
Greenhouse vegetables	76	76	156	97	103	156	103	103	103	103	103
Beans, nuts	15	15	50	51	89	99	59	43	43	43	43
Potato	80	80	80	80	81	80	97	80	80	80	80
Fruits	126	126	166	239	68	135	68	68	68	68	68
Berries	20	20	25	239	123	53	123	123	123	123	123
Cereals	130	183	166	203	249	268	249	249	249	249	249
Oils, fats	39	39	43	43	43	43	43	53	53	53	58
Fish	26	26	45	45	0	0	45	45	45	45	47
Egg	17	28	20	20	0	0	20	20	20	20	20
Meat, total	126	126	98	86	0	0	9	32	37	41	9
Beef	23	23	26	20	0	0	0	23	0	0	0
Pork	27	27	26	20	0	0	0	0	28	0	0
Poultry	29	29	32	37	0	0	0	0	0	32	0
Sausages	24	24	3	0	0	0	0	0	0	0	0
Cold cuts	16	16	3	0	0	0	0	0	0	0	0
Lamb, game, offal	7	7	9	9	0	0	9	9	9	9	9
Milk	519	519	544	525	0	0	0	417	0	0	0
Veg. milks	0	0	0	0	550	340	562	126	563	525	563
Sugar, etc.	31	83	60	0	4	4	32	32	32	32	32

Drinks	1346	1654	1302	745	1171	1171	1171	1171	1171	1171	1171
Salt	8	8	1	1	1	1	1	1	1	1	1
Other	28	28	26	26	26	26	26	26	26	26	26

3. Results

Climate impact results for diets are presented in Table 2 and Figure 1. M1 had the highest total impact on climate. It was the only alternative diet having a higher impact than current diets C1 and C2. V1 had the lowest total impact on climate, followed by V2 and D diets except D2.

Imported and domestic food ingredients and foods dominate climate impact (Figure 1). The share of impacts from domestic foods was lowest in V diets. The lowest share of imported foods was naturally in M1 and D diets, because domesticity was emphasized in the construction of the diets. Also in V1 domesticity was slightly emphasized as the consumption of soya was restricted and vegetables and so-called vegetarian milks were added compared to V2. Domesticity of products consumed did not guarantee low impacts on climate or cause extra load either. However, for example a comparison between V1 and V2 implies a slight advantage of domestic-based diets. In all cases, transportation of import had a negligible share of the total climate impact of the diets.

The manufacturing phase caused the second biggest impact on climate in each diet (Figure 1). It was at the same level in all the diets, and thus the share of manufacturing was biggest in V diets as their total impact was lowest.

Land use change caused minor impacts in every alternative diet, although the need for agricultural land differed considerably between diets (Table 1). In V, D1, D3, D4 and D5 diets, impacts from land use change decreased slightly, while in other alternatives it increased slightly. More impacts were caused by depletion of SOM from existing Finnish fields. (Figure 1.)

None of the diets fulfilled NNR completely, as there were shortages in the intake of nutrients (Table 3). According to the results, diets D2 and M2 had the fewest shortages, while C1 and C2 were worst, followed by V1 and V2. It was not possible to assess the amount of sugars and vitamin B6 due to the structure of the model and an incompleteness of data.

Table 2: Climate impacts of Finnish diets expressed as 1000 t_{CO₂} eq y⁻¹ and kg CO₂ eq day⁻¹ person⁻¹, change of climate impact of alternative diets (%) compared to current diets C1 and C2, and change in land use compared to C1. C1= current, based on food consumption studies; C2=as C1 but corrected based on national statistics; M1=basic mixed, based on NNR; M2=mixed but no "junk foods", based on NNR; V1= diverse vegan based on NNR; V2=soy based vegan based on NNR; D1= plant-based, favouring domesticity, based on NNR; D2=as D1 but supplemented by beef and milk; D3= as D1 but supplemented by pork; D4= as D1 but supplemented by poultry; D5= as D1 but supplemented by fish.

	C1	C2	M1	M2	V1	V2	D1	D2	D3	D4	D5	unk
Climate impact of diet	2522	2525	3092	3489	2572	2771	2085	2082	2029	2141	2026	1000 t _{CO₂} eq y ⁻¹
Climate impact of diet	4.79	5.14	5.61	4.94	2.85	2.89	3.4	4.44	3.28	3.82	3.68	kg CO ₂ eq day ⁻¹ person ⁻¹
Change of climate impact compared to C1			19	19	-40	-39	-28	-7	-25	-28	-28	%
Change of climate impact compared to C2			5	-15	-41	-41	-24	-13	-29	-28	-29	%
Change in land use compared to C1		10	2	15	-49	-49	-26	17	-31	-32	-32	%

Table 3. Incompatibility of nutrient intake in diets compared to NNR. C1= current, based on food consumption studies; C2=as C1, but corrected based on national statistics; M1=basic mixed, based on NNR; M2=mixed but no "junk foods", based on NNR; V1= diverse vegan based on NNR; V2=soy based vegan based on NNR; D1= plant-based, favouring domesticity, based on NNR; D2=as D1 but supplemented by beef and milk; D3= as D1 but supplemented by pork; D4= as D1 but supplemented by poultry; D5= as D1 but supplemented by fish.

	C1	C2	M1	M2	V1	V2	D1	D2	D3	D4	D5
Iron	Na, rub	as in C1	Na, rub	Na, (rub)	Na	as in V1	Na	Na	Na	Na	Na
Deficiency	D, B, A, folate Fe, I, Mg, K, poly, zinc, fiber	as in C1	D, poly, zinc	D	B2, B12, A, D, Fe for babies, zinc	as in V1	B2, D, zinc	D for babies	B2, D, Ca for babies	B2, D, Ca for babies and children, zinc	B2, D, Ca for babies and children

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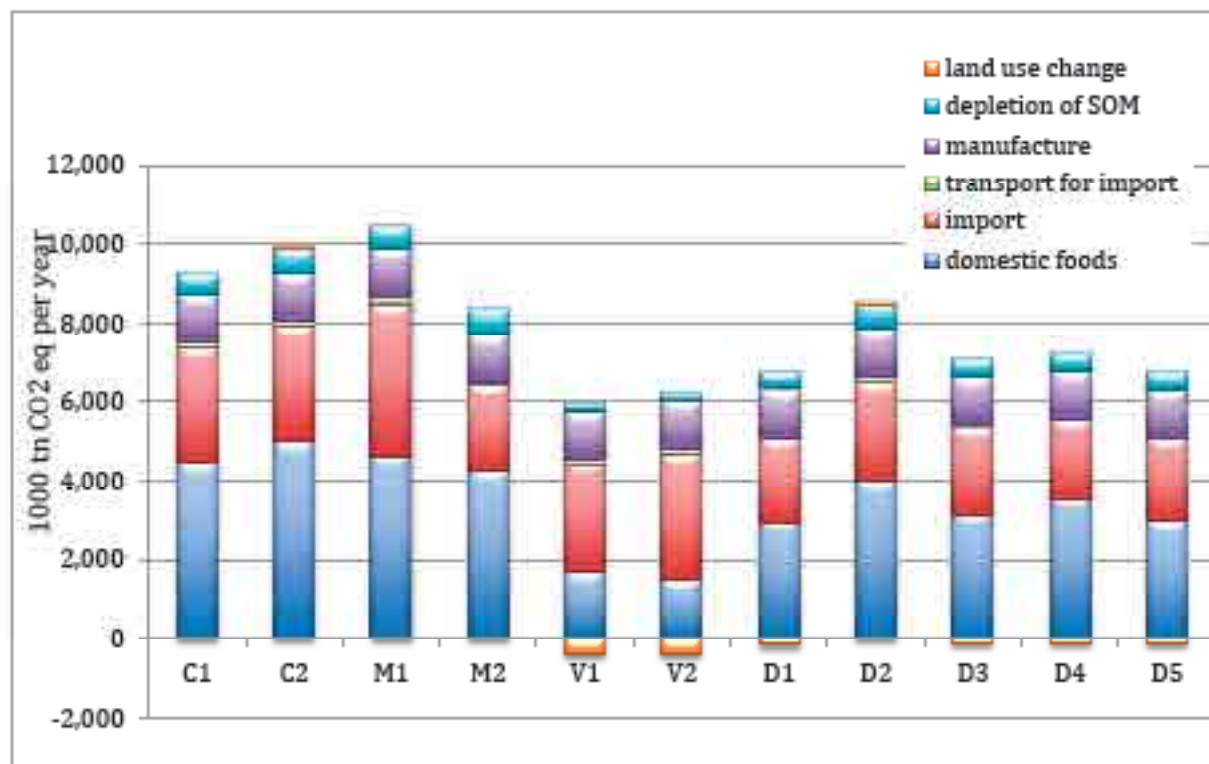


Figure 1: Climate impacts (1000 tn CO₂ eq per year) of Finnish diets. C1= current, based on food consumption studies; C2=as C1 but corrected based on national statistics; M1=basic mixed, based on NNR; M2=mixed but no “junk foods”, based on NNR; V1= diverse vegan based on NNR; V2=soy based vegan based on NNR; D1= plant-based, favouring domesticity, based on NNR; D2=as D1 but supplemented by beef and milk; D3= as D1 but supplemented by pork; D4= as D1 but supplemented by poultry; D5= as D1 but supplemented by fish.

4. Discussion

The diets following NNR do not necessarily result in decreased climate impact, as M1 had a higher impact on climate than C1 or C2. Heller et al. (2014) came to the same conclusion regarding the US situation. This contradicts results from a review by Hallström et al. (2015), in which omnivorous diets based on dietary guidelines (healthy diets) conferred a 0–35 per cent change compared to the reference. However, the other alternative diets in our study went below C1 and C2, and the change is in line with Hallström et al. (2015). In addition to a large total meat intake, the pattern of fruit and vegetable consumption accounts for the high climate impact of M1. In M2, the amount of greenhouse vegetables was kept on the present level, and the intake of roots, pulses and legumes was increased, as well as the intake of fruits and berries, since the recommendation suggests eating at least (not a maximum of) 500 g of vegetables, fruits and berries.

In general, the more diet is based on plant-based products the lower is its climate impact. The greatest increase of climate impact was caused by the introduction of beef and milk products (D2). The most favourable diet among the D diets was D1, which included a meagre amount but diverse selection of animal-based products. Very close was D5, which included more fish products than other diets. Also, Hallström et al. (2015) found out that vegan diets confer the biggest positive average change to climate impact.

A proper uncertainty analysis was not carried out. However, we identified an emission factor for beef as one of the major sources of uncertainty. Our factor was based on Usva et al. (2012), and it is rather low compared to other literature. The result for diets C1 and M1 would increase about 9 per cent if the emission factor for beef were doubled. It would highlight the difference between diets

including beef (C1, C2, M1, M2 and D2) and diets not including beef (V1, V2, D1, D3, D4 and D5), but not by very much. Another single product and corresponding emission factor which contributes significantly to the result is coffee. This is due to the fact that a great deal of coffee is consumed in Finland. If the emission factor for coffee (Bueser and Jungbluth 2009) were doubled, the climate impact of C1 would increase by 7 per cent, and decrease by 3.5 per cent if it were halved. Coffee is included in all the diets except M2.

For vegan diets, favouring a diversity of protein sources and outdoor vegetables seems to have slightly less climate impacts compared to relying on soya and the current structure of vegetable consumption. Export of impacts is also smaller in a more diverse vegan diet (V1). The difference between V1 and V2 could be greater, if impacts from land use change abroad were also to be included in the study. Domestic land use change, in turn, does not play a major role in one direction or another. Inclusion of emissions from agricultural soils to GWPs for products is more important.

Results for land use change are not directly comparable with results from the review by Hallström et al. (2015), because in that study only domestic land use change was included. Regarding M1 and M2, it seems that negative land use change (the need for additional land) is greater in this study, as land use change for a healthy diet in Hallström et al. (2015) is a 15–50 % reduction. On the other hand, more land would be released in V1 and V2 than regarding vegan diets in Hallström et al. (2015) (15–50 per cent), but in this case land use change related to import would supposedly affect results significantly.

The main concern in dietary change to a vegan diet relates to nutrition as there were severe shortages of essential nutrients in both vegan diets (V1 and V2). However, in many respects vegan diets would be favourable compared to the current diet. This is in accordance with Elorinne et al. (2016), who based their study on dietary records and measurements of biomarkers in plasma. They discovered that intakes of vitamins B12 and D were lower in a vegan diet than non-vegan, and found lower concentrations of serum D3, iodine and selenium for vegans, but more favourable fatty acid profiles (Elorinne et al. 2016). In general, animal-based foods provide, in addition to good quality protein, a lot of micronutrients (Neumann et al. 2002). We detected an effect of a moderate amount of animal-based foods on the nutritional quality of diets in our D diets, which included varied composition of small or moderate amount of animal-based foods. We found that the recommended intake of B12 could be achieved by introduction of any animal-based products to a vegan diet, but insufficient intake of B2 would be balanced only by introduction of (beef and) milk products. It also seems that balance in dietary fatty acids requests for fish-, cow- or pork-based products to be introduced into the diet (in addition to an increased amount of good quality vegetable oil, such as rapeseed oil). These findings are, however, very tentative and need further research based on more precise (disaggregated) data for food consumption and a sophisticated model. Also, Drewnowski et al. (2014) pointed out that additional research is needed on the point at which the higher climate impact of some nutrient-dense foods is offset by their higher nutritional value.

5. Conclusions

The results show that the climate impact of the Finnish diet can be reduced or increased by following NNR, depending on how the NNR are interpreted. It is not enough to adapt numerical recommendations related to different products groups, but further adaptation has to be done to achieve a decrease in climate impact. The results imply that nutrition and environmental recommendations should be better integrated in order to meet a need for a dietary change from the point of view of sustainability (see also Fischer and Garnett 2016). At first, consumption of red meat should be reduced in NNR. However, also the role of meat from other ruminants in the diet should be carefully considered when recommendations are made. Intake of some critical nutrients, such as B2, B12, vitamin D and A, and nufa, is at risk of being too low, while meat and milk consumption radically decreases. Secondly, special attention has to be paid to product choices within some product groups, such as vegetables, and giving more precise guidance. And thirdly, attention should also be paid to some products whose role in nutrition is not crucial, but which affect environmental impacts. The study was a tentative insight into the issue, and consideration of nutritional aspects in particular needs to be investigated in greater depth.

According to the recent reviews (Hallström 2014; Anestad and Fulgani 2015; Payne et al. 2016), knowledge on connection between climate impact of diet and dietary nutritional quality is still too vague. We did not just provide information on the issue to build growing evidence, but we took an approach in which we evaluated consistency of NNR as they include nutritional guidance in varied forms and nowadays also qualitative environmental guidance. This is a pragmatic approach as NNR is a tool to educate consumers and other practitioners to make difference to their choices. Our results hopefully help to improve quality of the guidance, and advances dietary shift to a direction of more sustainable average diet.

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153. Influence of Personal Consumption Behaviour on the Greenhouse Gas Emissions of different Dietary Patterns

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ABSTRACT

In the light of the considerable pressure exerted by food production on the environment, the assessment of the environmental burden of dietary choices has recently gained interest among the scientific community. Several studies based on life cycle thinking approach agreed that a transition from an omnivorous to a vegan or vegetarian diet would reduce the environmental impact associated with food production. However the majority of the these studies set the system boundaries up to the retail, excluding the consumption phase that can potentially give an important contribution to the environmental impact of diets. The aim of the present study was to assess how the personal consumption behaviour of an Italian citizen affects the emissions of greenhouse gases caused by three balanced dietary patterns (omnivorous, vegetarian and vegan), adopting a cradle to grave perspective.

The environmental performance of 36 balanced dietary scenarios was analysed through LCA, considering (i) different sources of heat for cooking, (ii) the attention of consumers to seasonality of fruit and (iii) the amount of food waste generated by consumers.

The production and preparation of food gave the highest contribution to the overall GHG emissions of all the scenarios analysed, however it was found that consumers' personal behaviour can play also an important role. Adopting a cradle to grave perspective is, therefore, of fundamental importance when assessing the greenhouse gas emissions of dietary patterns, both to provide comprehensive and valid information on sustainable consumption models to decision makers and to foster the consumers' environmental awareness.

Keywords: GHG emissions, omnivorous diet, vegetarian diet, vegan diet, food waste.

1. Introduction

The assessment of the environmental burden of food production and consumption through Life cycle assessment (LCA) has been rapidly raising interest in the last years (Nemecek et al., 2016). Particularly, the growing availability of data on the environmental performance of food items has allowed to adopt a wider perspective, assessing and comparing different dietary patterns (Nemecek et al., 2016). The majority of published peer-reviewed studies on dietary patterns mainly adopted a cradle to gate approach, excluding the consumption phase from their system boundaries (Halleström et al., 2015; Heller et al., 2013). Although modelling the consumption phase can be challenging due to high variability (Heller et al., 2013; Nemecek et al., 2016), it has been shown that it can considerably influence the impact on the environment (Gruber et al., 2016). Therefore, there is the need to include it in the system boundaries in order to reach a comprehensive deeper knowledge of the main contributions to the environmental impact and derive clear and practical hints for consumers and decision-makers towards the reduction of the environmental impact of food production and consumption.

The present study aims to assess the influence of personal behaviour of an average Italian citizen on the GHG emissions associated with three dietary patterns: omnivorous, vegetarian and vegan diet.

2. Methods

Three dietary patterns have been considered: omnivorous, vegetarian and vegan. Three average weekly menus (Table 1) have been hypothesized for an average Italian man, aged between 18 and 59 on the basis of recommended daily intake (LARN) released by the Italian Society of Human Nutrition (SINU) (SINU, 2014), reported in Table 2. In the cases in which SINU has not reported a reference range for nutrients intake, lower and higher burden of the range were assumed respectively equal to 80% and 120% of the average value. The nutrients content of food items was taken from BDA database (IBO, 2016) and from Environmental Products Declarations (EPD) (www.environmentaldec.com).

It was not possible to satisfy vitamin B12 requirements for the vegetarian and vegan diet, because it is mainly supplied by meat and the possible assumption of food integrators was not taken into consideration in the present study due to a lack of data.

Table 1: Weight and number of foods portions considered for the three dietary patterns

	Portion weight (g)	Weekly number of portions		
		Omnivorous	Vegetarian	Vegan
Bread/biscuits	50/20	23	23	23
Pasta and rice	80	7	7	8
Potatoes	200	2	3	3
Fruit	150	21	21	21
Fresh vegetables	80	7	7	7
Cooked vegetables	200	7	7	7
Butter/margarine	10	5	5	5
Oil	10	20	20	20
Milk and yogurt	125	14	14	14*
Fresh cheese	100	2	2	0
Ripened cheese	50	2	2	0
Meat	70	4	0	0
Eggs	60	2	2	0
Legumes	150	2	7	9
Cured meat	50	3	0	0
Fish	150	3	0	0
Tofu	180	0	4	6

* Soy milk

Table 2: LARN for a man aged between 18 and 59 and nutrients provided by the three analysed dietary patterns

	Energy (kcal/d)	Proteins (g/d)	Fat (%of kcal)	Carbohydrates (%of kcal)	Iron (mg/d)	Calcium (mg/d)	Vit B12 (µg/d)
Reference value	2590	63			10.0	1000	2.4
Min	2072	50.4	20%	45%	8.0	800	1.9
Max	3108	75.6	35%	60%	**	2500	**
Omnivorous diet	2180	75.1	30%	52%	10.5	943	4.7
Vegetarian diet	2124	66.1	29%	54%	18.4	998	1.1
Vegan diet	2153	70.3	28%	60%	23.2	1033	0.0

** The maximum tolerable dose was not reported for iron and Vitamin B12

Besides the choice of the dietary pattern, other elements of human behaviour were considered, namely (i) the source of heat for cooking (gas cooker, induction burner), (ii) the attention to fruit seasonality and (iii) the generation of food waste during the use phase. 36 scenarios were developed and their GHG emissions were estimated.

GHG emissions of food items were taken from EPDs (www.epfindec.com) and from published peer-reviewed articles. When a food category comprised different products, average GHG emissions were estimated. The cooking phase was modelled assuming a 55% efficiency for the gas cooker and 92% for the induction burner. The power of the burner (2/3 kW) and the cooking time was defined according to the food item. The attention to seasonality was considered for fruit but not for vegetables due to lack of data. Particularly, it was assumed that careful consumers do not buy fruit greenhouse-grown or exotic fruit, whereas an average of seasonal, greenhouse-grown and exotic fruit was considered for the other consumers. Data on average, minimum and maximum waste generation at consumers' were taken from the work by Vanham et al. (2015), considering only avoidable waste.

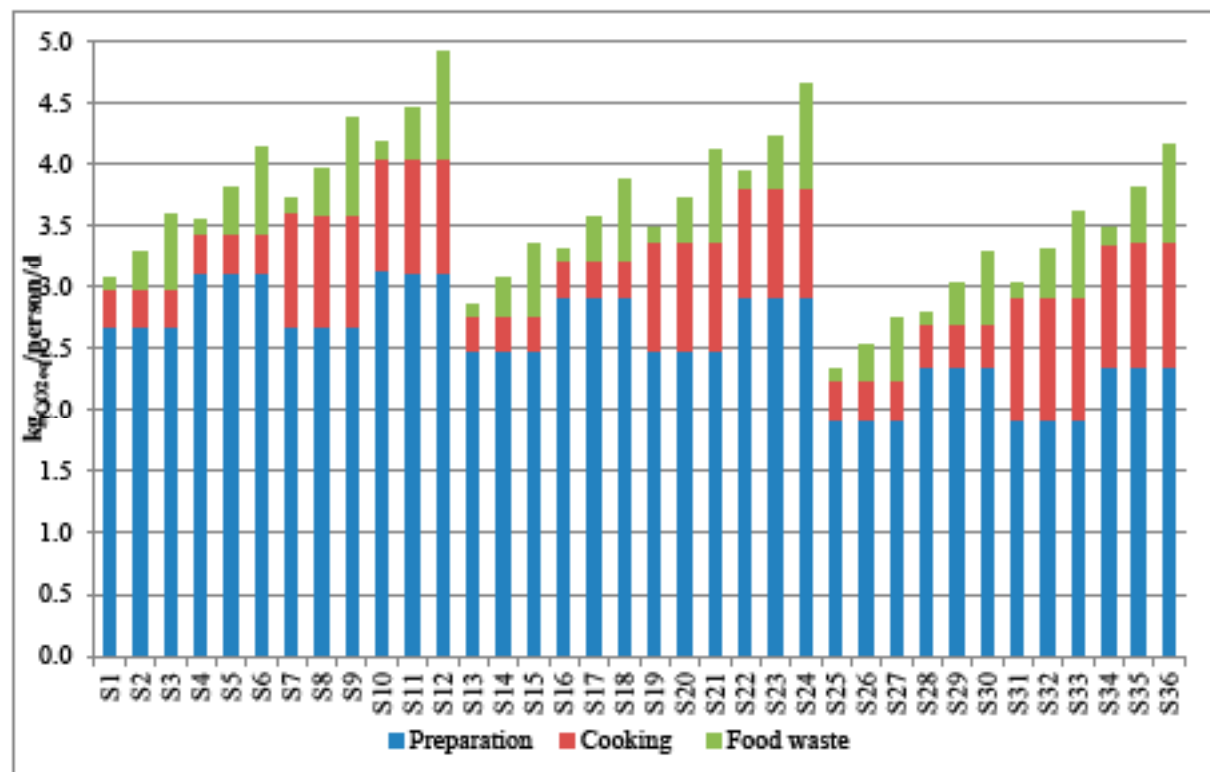
3. Results

The GHG emissions of 36 scenarios have been assessed (Table 3). The GHG emissions of each scenario are reported in Figure 3, highlighting the contributions of (i) the preparation phase, which includes all the stages of the supply chain for the production and preparation of daily food intake excluding the energy for cooking, (ii) the heat for cooking and (iii) the food waste, which comprise all the stages of the supply chain for the production of food that is discarded by consumers.

Table 3: Scenarios analysed

Scenario	Type of diet	Source of heat	Seasonality	Waste at consumers
S1	Omnivorous	Gas cooker	Only seasonal	Low
S2	Omnivorous	Gas cooker	Only seasonal	Average
S3	Omnivorous	Gas cooker	Only seasonal	High
S4	Omnivorous	Gas cooker	Seasonal and non-seasonal	Low
S5	Omnivorous	Gas cooker	Seasonal and non-seasonal	Average
S6	Omnivorous	Gas cooker	Seasonal and non-seasonal	High
S7	Omnivorous	Induction b.	Only seasonal	Low
S8	Omnivorous	Induction b.	Only seasonal	Average
S9	Omnivorous	Induction b.	Only seasonal	High
S10	Omnivorous	Induction b.	Seasonal and non-seasonal	Low
S11	Omnivorous	Induction b.	Seasonal and non-seasonal	Average
S12	Omnivorous	Induction b.	Seasonal and non-seasonal	High
S13	Vegetarian	Gas cooker	Only seasonal	Low
S14	Vegetarian	Gas cooker	Only seasonal	Average
S15	Vegetarian	Gas cooker	Only seasonal	High
S16	Vegetarian	Gas cooker	Seasonal and non-seasonal	Low
S17	Vegetarian	Gas cooker	Seasonal and non-seasonal	Average
S18	Vegetarian	Gas cooker	Seasonal and non-seasonal	High
S19	Vegetarian	Induction b.	Only seasonal	Low
S20	Vegetarian	Induction b.	Only seasonal	Average
S21	Vegetarian	Induction b.	Only seasonal	High
S22	Vegetarian	Induction b.	Seasonal and non-seasonal	Low
S23	Vegetarian	Induction b.	Seasonal and non-seasonal	Average
S24	Vegetarian	Induction b.	Seasonal and non-seasonal	High
S25	Vegan	Gas cooker	Only seasonal	Low
S26	Vegan	Gas cooker	Only seasonal	Average
S27	Vegan	Gas cooker	Only seasonal	High
S28	Vegan	Gas cooker	Seasonal and non-seasonal	Low
S29	Vegan	Gas cooker	Seasonal and non-seasonal	Average
S30	Vegan	Gas cooker	Seasonal and non-seasonal	High
S31	Vegan	Induction b.	Only seasonal	Low
S32	Vegan	Induction b.	Only seasonal	Average
S33	Vegan	Induction b.	Only seasonal	High
S34	Vegan	Induction b.	Seasonal and non-seasonal	Low
S35	Vegan	Induction b.	Seasonal and non-seasonal	Average
S36	Vegan	Induction b.	Seasonal and non-seasonal	High

Figure 3: GHG emissions of the 36 scenarios analysed. Preparation = all the stages of the supply chain for the production and preparation of the daily food intake excluded the heat for cooking; Cooking = heat for cooking; Food waste = all the stages of the supply chain for the production of food that is discarded by consumers



4. Discussion

The carbon footprint of the scenarios analysed ranged between 2.3 kg_{CO₂e}/person/day, in case of vegan diet, use of gas cooker, low waste and choice of seasonal fruit and 4.9 kg_{CO₂e}/person/day for the scenario with omnivorous diet, use of induction burner, high waste and choice of both seasonal and non-seasonal fruit.

The preparation phase represented the highest contribution to all the scenarios, ranging between 53% (S33) and 88% (S4) of the total GHG emissions. The highest emissions for food production and preparation were associated with the omnivorous diet, whereas the lowest could be obtained with the vegan one.

Consumers' choices on fruit seasonality, source of heat for cooking and amount of food waste had an influence on the overall impact of the diet.

Choosing only seasonal fruit instead of greenhouse-grown one allowed to reduce the GHG emissions of the diet by about 0.5 kg_{CO₂e}/person/day.

Cooking with induction burner instead of cooking boiler resulted in increasing GHG emissions by about 0.6 kg_{CO₂e}/person/day. However, it has to be considered that Italian energy mix was assumed for the present study and different results could be obtained if the electric energy were generated domestically from a renewable source (e.g. solar).

The impact of wasted food was influenced by the type of food, the source of heat for cooking and the attention to fruit seasonality, and it ranged between 3% (S13) and 19% (S33) of the overall GHG emissions. Indeed wasting food means "wasting" all the GHG emissions that had happened to produce it. The highest impact of food waste was associated with an omnivorous diet, whereas the lowest were found to be in relation to the vegan diet. However, differences among the three diets were lower than the one found for the preparation of food because generally higher amount of food of vegetable origin is wasted.

Overall, at the same conditions, vegan diets resulted to cause lower GHG emissions than vegetarian and omnivorous ones, however choosing a vegan diet is not always the best solution to reduce the GHG emissions of the diet and attention should be paid also for consumer's personal behaviour. Indeed, if on one hand a transition from scenario S1 to scenario S25 would allow to reduce by about 25% the GHG emissions of the diet, on the other hand, scenario S36 generated a 25% increase of the GHG emissions compared to scenario S1.

5. Conclusions

The present study highlighted that, despite the prevalent role of food production and preparation, consumers' behaviour can give an important contribution to the GHG emissions of the diet. Indeed, according to the present analysis, a vegan diet could generate higher GHG emissions than an omnivorous one if low attention is given to fruit seasonality, amount of wasted food and the source of heat for cooking. Furthermore the present analysis did not assess the impact of other variables that are influenced by consumers' personal behaviour, such as the distance to the retailer and the means of transport used by consumers to go shopping. Adopting a cradle to grave perspective is, therefore, of fundamental importance when assessing the GHG emissions of diets, both to provide to decision makers comprehensive information on sustainable consumption models and to foster the environmental awareness of consumers.

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269. Environmental impacts of scenarios for food provision in Switzerland

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ABSTRACT

This study evaluates and compares the environmental impact of food provision for the average consumption in Switzerland with 6 different scenarios, per person and year. Furthermore, the environmental impact of the recommendation for a sustainable and healthy food intake is assessed. The full environmental impacts are assessed with the ecological scarcity method 2013. Additionally, the global warming potential is evaluated (IPCC GWP 100a, incl. RFI). The lower the share of animal-based food in the diet is, the lower the environmental impacts are. The provision of "meat and fish" and "animal products" is most relevant to explain the differences in the overall environmental impact between the food scenarios.

Keywords: food provision, diet, vegan, ovo-lacto-vegetarian, ovo-lacto-vegetarian, flexitarian.

1. Introduction

This study has been commissioned by the WWF Switzerland to provide guidance to consumers in terms of the environmental impact of different food consumption scenarios (Jungbluth, Eggenberger et al. 2016).

There are different options to determine the amount of food production of a specific nutrition type. The environmental impacts of food consumption in Switzerland were already investigated from different starting points (Figure 4), such as: top-down splitting the overall environmental impacts to different consumption areas in an input-output analysis (Jungbluth, Nathani et al. 2011), food availability on the Swiss market (SBV market availability, Jungbluth, Iiten et al. 2012; Schweizerischer Bauernverband 2013), data for large distributors such as supermarkets (MIGROS, Jungbluth 2011) or canteens (SV Group, Jungbluth, Keller et al. 2015), data from the Swiss BFS household budget surveys on food purchases (BFS 2012; Samer, Beretta et al. 2015; Jungbluth, Eggenberger et al. 2016), meals consumed (Jungbluth, Keller et al. 2015), nutritional recommendations (Brunner and Casetti 2014; Eggenberger and Jungbluth 2015).

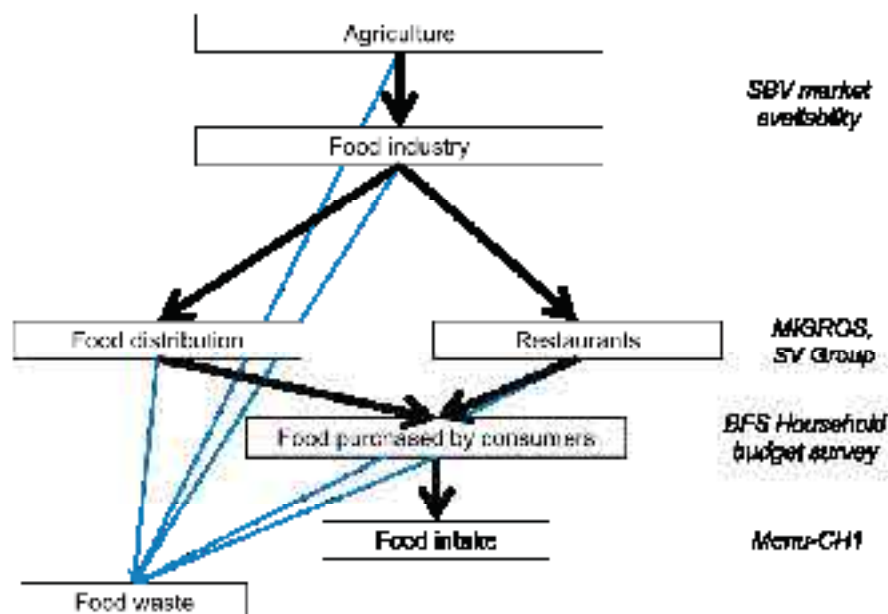


Figure 4: Food flows (black arrows) and different points of investigation to estimate the environmental impact. Food waste flows are shown with blue arrows.

The commissioner proposed 7 different diet styles, predefining their labelling and some basic characteristics. Further information on food provision per diet type and the environmental impacts was then elaborated by ESU-services. The information available on the amount of food produced for the Swiss market in 2012 (Schweizerischer Bauernverband 2013) was used in this study to assess the average provision. Various studies and dietary recommendations were used in order to determine the production of food per specific nutrition type (Taylor 2000; USDA and USDH 2010; Meier and Christen 2013; Schweizerischer Bauernverband 2013; Leitzmann 2014; SGE 2014; van Dooren, Marimssen et al. 2014). The final report has been used for an article in the quarterly WWF member magazine.

2. Goal and Scope

This study evaluates and compares the environmental impact of the food provision for the average diet and for 6 different food provision scenarios in Switzerland, per person and year in a life cycle assessment (LCA). The starting point is the average availability of food products on the Swiss market. From this starting point, the following six scenarios are estimated: vegan, ovo-lacto-vegetarian, ovo-lacto-pescetarian, flexitarian, protein-oriented, meat-oriented. Furthermore, the impacts of just providing the food as recommended by the SGE (2014) for a sustainable and healthy diet is used as an eighth scenario (FOODprints[®]). In this last scenario, food losses in the life cycle are not accounted for.

The analysis includes the full life cycle of the food products until they are purchased in the supermarket. Transport to home and preparation are not included in this study. As far as data is available, conventionally produced consumption goods are included and compared. Organically cultivated food is not included in the framework of this study. For vegetable cultivation, the share of greenhouse cultivation is estimated. Switzerland is set as origin of food products whenever possible. Transport is estimated with average distances. Food losses in the life cycle are included in the data available for the food supply. Food losses in agriculture are roughly assessed (Schweizerischer Bauernverband 2013).

Health aspects are only considered in the recommendations of SGE for a sustainable and healthy diet but they are not a general focus of this study. Therefore, different scenarios for food supply are not necessarily comparable from a nutritional point of view. There might be certain health problems associated with the different scenarios. Also, the present consumption patterns of e.g. alcohol or sugar might not be healthy. Furthermore, individual demand for nutrients depends on e.g. age, health, gender, type of work or pregnancy and thus individual food intake patterns might be quite variable.

The total environmental impacts are assessed with the Ecological Scarcity Method 2013 and are presented as eco-points (Frischknecht, Büsser Knüpfel et al. 2013). Furthermore, the greenhouse gas potential is also evaluated (IPCC GWP 100a, IPCC 2013). The higher effect of greenhouse gas emissions by airplanes is included in this assessment with an RFI¹ factor (Jungbluth 2013).

¹ Radiative Forcing Index

3. Life Cycle Inventory (LCI)

The starting point is the amount of food available for the Swiss market in 2012 (Schweizerischer Bauernverband 2013). It includes the production in Switzerland and the balance of imports and exports. This information is complemented with additional statistics, for example data on beverage consumption. The values are then related to a single person in order to assess the environmental impact per year and person. This scenario corresponds to the Swiss average diet as it was estimated by the commissioner with a weekly consumption of 1kg of meat, 21 portions of milk products and 3 to 4 eggs.

Based on this average market availability, 6 different diet scenarios are estimated as shown in Table 4. The deviation is based on different studies for dietary recommendations and pre-defined values by the commissioner. The scenarios only model a change in protein intake. Other factors, for example the provision of alcohol and mineral water, were not changed in comparison to the Swiss average scenario. The included amount of food provision is considerably higher than the finally eaten amount of food because food losses occurring at various points within the life cycle are included.

Additionally, the FOODprints® scenario is investigated. It refers to recommendations for a sustainable and healthy diet by the Swiss Society for Nutrition (Schweizerische Gesellschaft für Ernährung, SGE 2014). This "ideal" scenario is based on intake recommendations and does not consider food losses along the product life cycle. Therefore, it shows a lower amount of food produced and calorie intake. This scenario does not include any products transported by plane or cultivated in a heated greenhouse. Unhealthy food products such as alcoholic beverages and sweets are considerably reduced. Mineral water is fully replaced by tap water.

Table 4: Market availability (2012) and derived assumptions per diet and food product group. The FOODprints® scenario refers to the actually consumed amount of food.

Food product group	Unit	Average 2012	Vegan	Ovo- lacto- Vegan- itarian	Ovo- lacto- Pesci- tarian	Flexi- tarian	Protein- oriented	Meat- oriented	FOOD prints®
Vegetables	kg	107	200	133	133	128	33	33	131
Fruits	kg	81	76	76	76	68	30	30	75
Cereal Products	kg	171	171	171	171	171	171	171	111
Eggs and Honey	kg	13	0	16	16	12	33	20	9
Milk, Milk Products	kg	144	0	144	144	144	208	144	155
Meat	kg	50	0	0	0	16	78	104	13
Fish	kg	8	0	0	14	4	8	8	3
Meat alternatives and soy milk	kg	0	129	16	14	8	0	0	11
Fats and Oils	kg	30	30	30	30	38	30	30	11
Pulses	kg	1	8	4	4	3	1	0	3
Nuts	kg	4	13	13	11	8	4	2	9
Non-alcoholic beverages, without tap water	kg	215	215	215	215	215	215	215	34
Alcoholic beverages	kg	94	94	94	94	94	94	94	31
Total (without beverages)	kg	587	637	602	612	582	610	582	333
Calories (without beverages)	kcal/d	3227	2980	3288	3285	3282	3338	3282	2371
Protein	g/d	101	173	90	94	98	147	121	77

The average amount of food available on the Swiss market is set to 600 kg per year and person, beverages not included. This amount is decisively higher than the finally consumed food because various food losses occur within the product life cycle. The following assumptions were made for the different food provision scenarios:

Based on the average scenario and as compensation for the complete abstinence from meat and fish, the amount of meat alternatives, pulses and nuts was increased for the ovo-lacto-

vegetarian and the vegan scenario. The increase corresponds to the conversion factors such as they are applied in van Dooren et al. (2014) between the average Dutch consumption and the vegetarian and vegan diet. Therefore, the protein supply is considered to be sufficient in all scenarios.

For the amount of vegetables in the vegan scenario, the respective amount in the ovo-lacto-vegetarian scenario was multiplied by factor 1.5. The provision of fruits was not modified. These estimations are oriented towards the dietary recommendations for the vegan diet, as they are stated in van Dooren (2014, factor 2) and in USDA (2010, no increase). Furthermore, soy milk substitutes the amount of milk and milk products consumed in the average scenario (see van Dooren, Marinissen et al. 2014).

For the ovo-lacto-vegetarian scenario, the amount of vegetables and fruits consumed in the average diet was increased by 25 percent (mean value between vegan and average diet). The flexitarian scenario is calculated as the mean values of the average and the ovo-lacto-vegetarian scenario.

For the protein- and meat-oriented scenarios, the amount of animal products was increased to the values defined by the commissioner. In contrary, the amount of vegetables and fruits is only half of the consumed amount in the average scenario, because a compensation of the higher values in meat and milk product consumption is assumed.

The amount of grain products is set the same for all diet scenarios. The same is true for oils and fats, as well as for alcoholic and non-alcoholic beverages (except FOODprints®). Tap water is not included in Table 4, but taken into account in the life cycle inventory analysis.

Table 5 provides further information on the food products included in the life cycle inventory analysis per food product group. The modelling of the LCI is based on the assumptions for food product groups shown in Table 4. For the LCI, more detailed data is used. Each food product group is allocated to single food products as they are shown in the middle column (for example division of the overall amount of meat in beef, veal, pork and poultry). The inventory analysis is based on data of the ecoinvent database (ecoinvent Centre 2010; LC-inventories 2016) and data of the ESU-database (ESU 2016; Jungbluth, Keller et al. 2016). In the LCIA, the various food product groups are limited to a reduced number of food categories, what allows an easier interpretation of the results at a later stage (right column).

Table 5: Food product groups, the included food products and the respective food category in LCIA

Food product group	Included food products	Food category in LCIA
Vegetables	Diverse sorts of vegetables, white mushrooms, herbs	Vegetables and Fruits
Fruits	Diverse sorts of fruits	Vegetables and Fruits
Grain Products	Bread, flour, grain, rice, potatoes, sugar, biscuits	Grain Products
Eggs and Honey	Eggs, Honey	Animal Products
Milk, Milk products	Milk, cheese, yoghurt, cream, whey protein powder	Animal Products
Meat	Diverse sorts of meat such as beef, veal, pork, poultry	Meat and Fish
Fish	Fish, mollusk, crustaceans	Meat and Fish
Meat alternatives and soy milk	Tofu, seitan, soy milk	Vegetable Proteins
Fats and Oils	Diverse sorts of vegetable oils, margarine, butter, chocolate	Fats and Oils
Pulses	Pulses such as lentils	Vegetable Proteins
Nuts	Nuts such as almonds	Vegetable Proteins
Non-alcoholic beverages, without tap water	Coffee, tea, fruit juices, soda, mineral water	Beverages
---	Tap Water	Beverages
Alcoholic beverages	Beer, wine, Spirits	Beverages

4. Life Cycle Impact Assessment (LCIA)

Figure 5 shows that the environmental impact of the average food consumption scenario amounts to about 5 million eco-points per person and year. This figure is comparable to the results calculated with an input-output analysis for Switzerland (Jungbluth, Nathani et al. 2011).

The FOODprints®-scenario causes a low environmental impact but cannot be fully compared to the other diet styles because only the food intake is considered. Regarding the food provision scenarios, the food provision for the vegan diet scenario causes the lowest environmental impacts. The highest impacts are calculated for the scenarios “protein-oriented” and “meat-oriented”. The impacts caused by the product groups “meat and fish” and “animal products” are most relevant and explain the main differences in the overall environmental impact between the different food scenarios (see Figure 5).

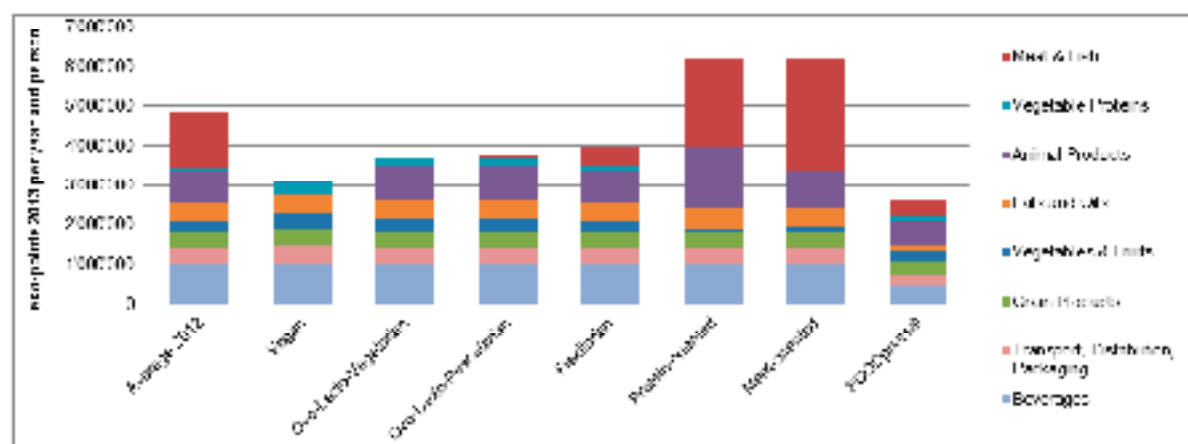


Figure 5: Environmental impacts of different dietary scenarios split per food product groups (eco-points per year and person).

The same statements made for the environmental impact (Figure 5) are also true for the results regarding the global warming potential as shown in Figure 6. The pattern is very similar and only small differences are identifiable for the global warming potential compared to the total environmental impacts. The share of animal products (mainly milk products) is higher than in the total environmental impact because the methane produced by cattle has a higher relative impact. In addition, the emission of greenhouse gases in transport has a bigger share in the global warming potential than it has in the total environmental impacts.

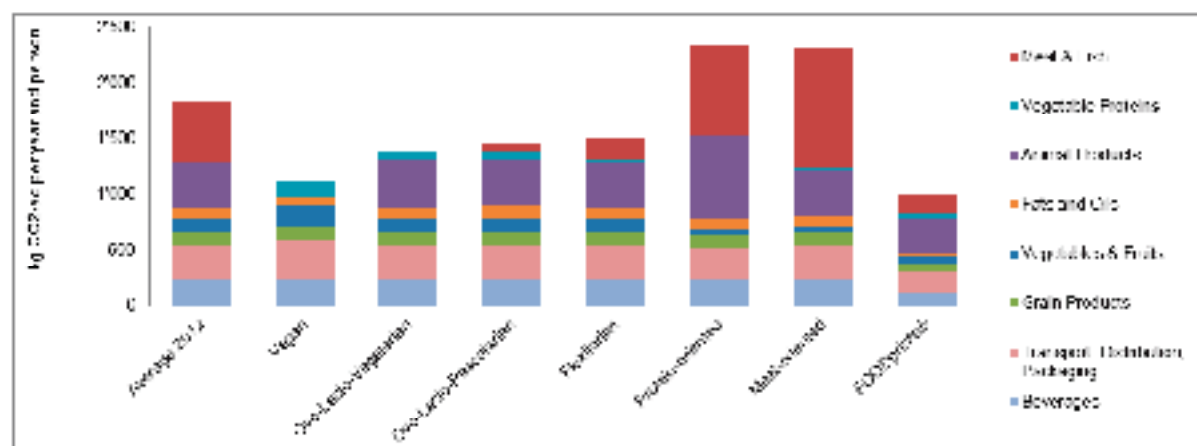


Figure 6: Global warming potential of the different dietary scenarios, split per food product group (kg CO₂-eq per year and person)

The analysis of shares of the different environmental impact categories is presented in Figure 7. The impact categories “heavy metals into soil” and “main air pollutants” and “particulate matter” show the highest variability regarding their influence on the overall impact. “Heavy metals into soil” play a more important role in the vegetable based food provision scenarios. The environmental impact of this impact category is predominantly caused by the fertilizer and pesticide usage in coffee and wine production. However, as the absolute amount of coffee and wine provision is defined the same for all scenarios (except FOODprints®), differences in the share of that impact category are mainly explained by the lower overall impact of vegetable based food provision scenarios. In contrary, the food provision scenarios rather based on animal products show a higher share in main air pollutants and particulate matter. This is caused by the ammonia and nitrogen oxides emissions in livestock breeding. The share of the other impact categories does not vary between the different scenarios in a relevant way.

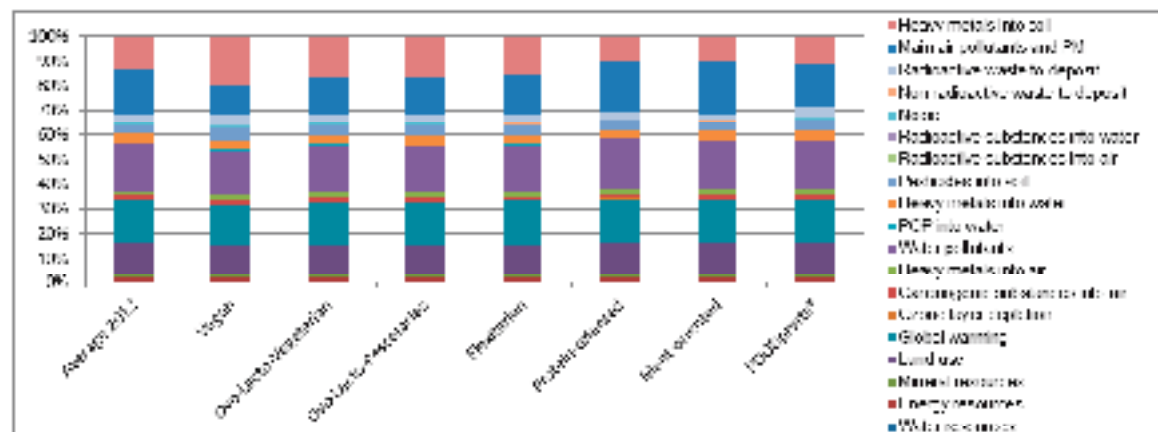


Figure 7: Environmental impact of the different scenarios, split per impact categories of the Ecological Scarcity method 2013

5. Interpretation

The results confirm the important role of meat and fish provision concerning the environmental impact of diets in Switzerland. Vegetable proteins in the meat-reduced diets cause a lower environmental impact. This is even true for the vegan diet scenario, which is characterized by an increased amount of consumed vegetable proteins in order to substitute meat, fish and other animal products such as milk and eggs. The impact of other animal products is to be highlighted as well when addressing reduction potentials of environmental impacts. After meat and fish, this food product group is the second most important source of the environmental impact of diets.

For all scenarios, the same amount of beverages was assumed (except FOODprints®). Therefore, this food product group does not influence the differences between the diet scenarios. Nevertheless, the beverage provision has to be considered when assessing environmental impact reduction potentials. On average, almost a quarter of the total impact is caused by beverages (particularly wine and coffee).

Figure 8 provides an overview on the results of the studies conducted so far in Switzerland and includes some of the results obtained in the study at hand (“market availability”, “scenario, availability, vegan”, “scenario, availability, meat oriented”, “scenario, intake recommendation, FOODprints®”). The results depend on the starting point of the analysis. The highest environmental impacts are those obtained with the top-down approach (input-output-analysis) (Jungbluth, Eggenberger et al. 2016) followed by the modelling of the food availability presented here. If impacts are calculated based on recommended diets, the amounts of food are much lower than the real market availability. This can be explained by food waste in different stages of production and by a possible overconsumption. Estimates based on nutritional recommendations also tend to underestimate the impacts because they seem to omit parts of frequently consumed food (e.g. alcohol or sweets). Hence, there are huge differences between the impact results when considering different starting points in terms of the amount of food included.

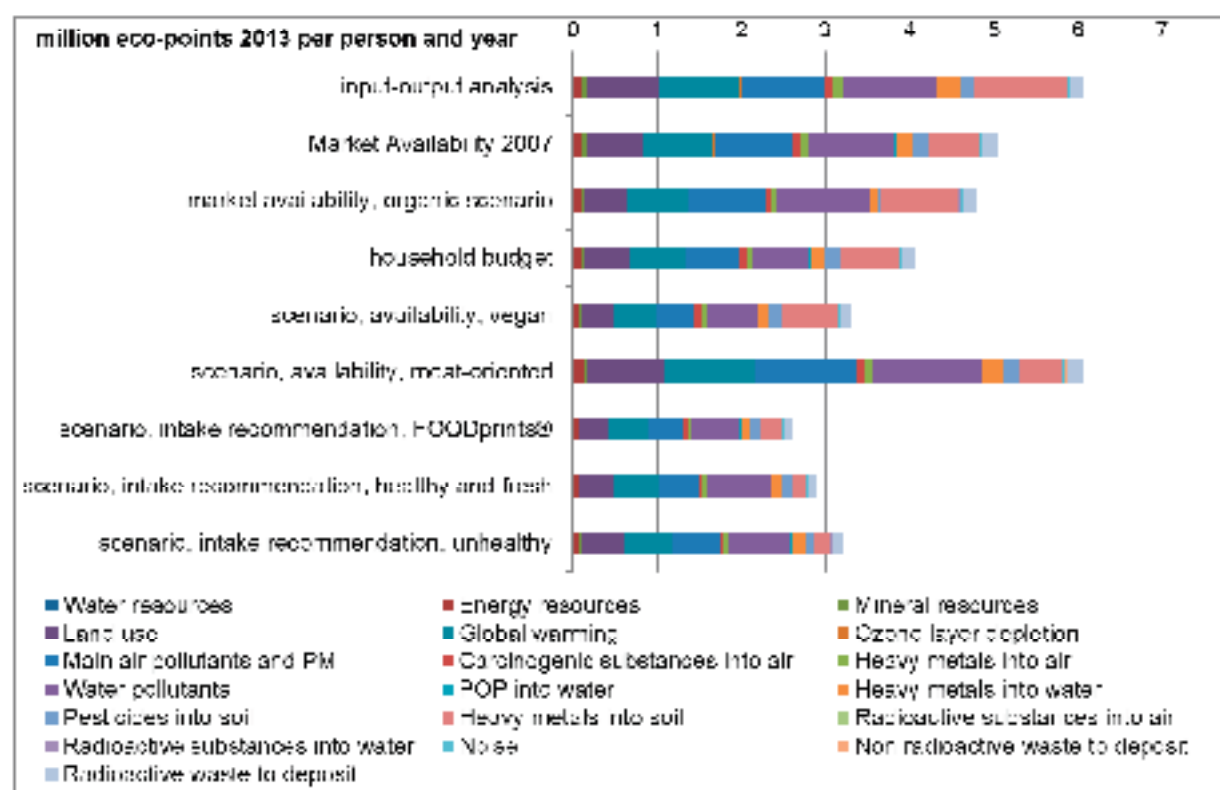


Figure 8: Environmental impacts of food consumption calculated for Switzerland (eco-points per person per year). Calculation is conducted with different statistics, accounting methods and scenarios.

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ORAL PAPERS

CASE STUDIES: PRODUCTS

12. Case studies: products

253. Global warming potential of chocolate ice cream considering parameter uncertainties

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ABSTRACT

This paper aims to evaluate the global warming potential (GWP) associated with the production and consumption of chocolate ice cream considering parameter uncertainties. A further objective is to identify hotspots and opportunities for improvements. Two types of chocolate ice cream are considered - regular and premium - with the main difference being the fat, sugar, cocoa powder and egg content, which are higher for the latter than the former. The study follows the ISO 14040/14044 guidelines for life cycle assessment (LCA). The functional unit is defined as '1 kg of chocolate ice cream packaged in a plastic tub'. The scope of the study is from 'cradle-to-grave', comprising production of raw materials, ice cream manufacturing, retail, consumption and end-of-life (EOL) waste management; transport across the whole life cycle is also considered. Data have been sourced from manufacturers, literature and LCA databases. The results show that the GWP of premium ice cream is 6% higher than that of the regular version (6.9 and 6.5 kg CO₂ eq./kg, respectively). The raw materials contribute the majority (74%) of the impact. The next contributing stage is manufacturing, adding around 10%, followed by the packaging and retail with around 7% each. For the ingredients, cocoa is the hotspot (2.7 kg CO₂ eq./kg ice cream), followed by milk (2-2.1 kg CO₂ eq./kg). For the former, this is due to cocoa cultivation and, for the latter, agricultural and livestock practices. Within the manufacturing stage, energy consumption is the major contributor to the GWP (86%). The results of the uncertainty analysis obtained by Monte Carlo simulation show that the impact ranges from 6.1-6.9 kg CO₂ eq./kg for the regular and 6.4-7.2 kg CO₂ eq./kg for the premium ice cream. The probability of the premium ice cream having higher GWP than the regular is estimated at 92% suggesting high confidence in the results.

Keywords: energy use in food chains, global warming potential, ice cream; life cycle assessment.

1. Introduction

Ice cream is a widely consumed frozen dessert. In the UK, it contributed 80% of the total sales of frozen desserts in 2013 (Key Note, 2014). Market analysts predict a growth of the ice cream market of about 13% from 2014-2018 (Mintel, 2014), reaching the sales value of £1.27 bn (Key Note, 2014). Chocolate and vanilla are the dominant ice cream flavours in the UK and, therefore, can be expected to make up the bulk of this anticipated growth in sales.

Ice cream can be categorised by its ingredient composition into the following: super premium, premium, regular, reduced fat, low fat and non-fat ice cream (Marshall et al., 2003). Additionally to this categorisation, the British Frozen Food Federation (BFFF) subdivides ice cream into take-home and wrapped impulse ice cream (BFFF, 2014). The former is for consumption at home whereas the latter is for immediate consumption. Consumers in the UK prefer the premium and regular versions of take-home ice cream, thus, sales of take-home ice cream contribute more than 78% of the total national sales of ice cream (Key Note, 2010).

Regardless of its economic and social importance, there is little information on the environmental sustainability of ice cream. Consequently, this paper intends to contribute to a greater understanding of how its production and consumption affect the environment. The focus is on the global warming potential (GWP) with the aim of identifying opportunities for mitigate climate change impacts along the whole supply chain. Life cycle assessment (LCA) has been used as a tool for these purposes.

2. Methods

The LCA study follows the ISO 14040/14044 methodology (ISO, 2006a, 2006b). The goal is to assess the GWP of chocolate ice cream manufactured and consumed in the UK. The functional unit is defined as '1 kg of chocolate ice cream packaged in a tub'. The system boundary is from 'cradle-to-grave', encompassing the following life cycle stages (Figure 1): raw materials production,

manufacturing, retail, consumption, end-of-life (EOL) waste management and transport throughout all stages.

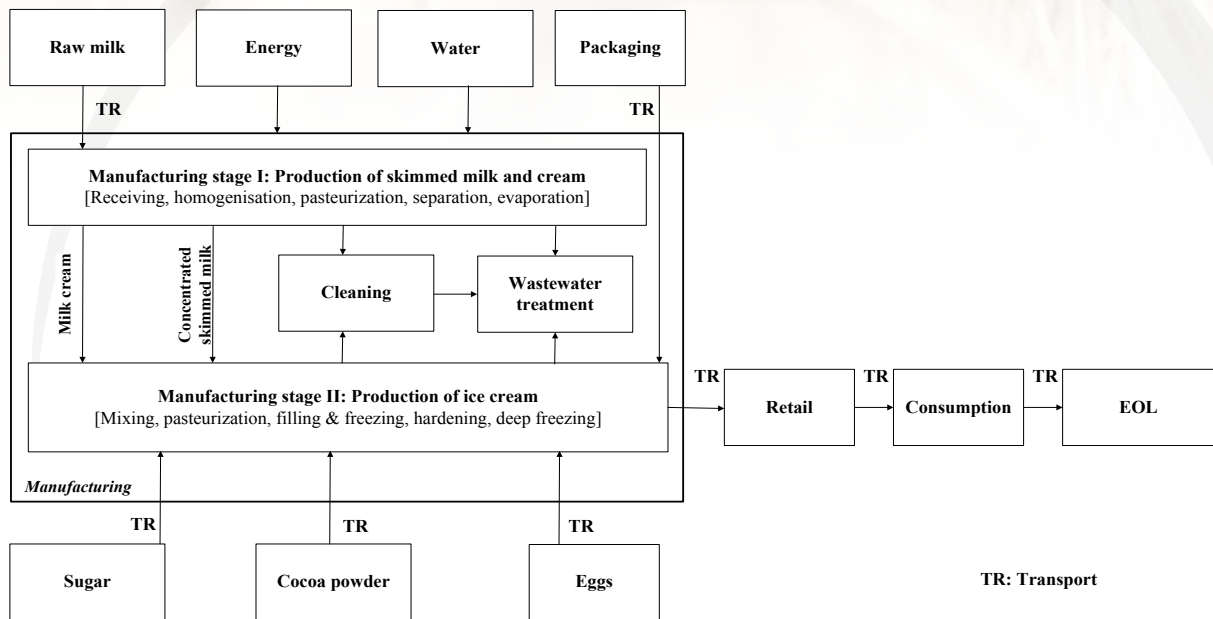


Figure 1: System boundaries and the life cycle of chocolate ice cream

Raw materials (ingredients): Raw milk is the principal component. The other ingredients are sugar, eggs and cocoa powder. Raw milk and eggs are produced within the UK whereas sugar is partly produced from sugar beet (60%) in the UK, with the rest being from imported sugar cane (40%). Cocoa is cultivated in West Africa imported and processed to cocoa powder within the UK.

Manufacturing: Manufacturing includes two different stages: pre-processing of raw milk (Manufacturing stage I) and the core manufacturing stage of ice cream (Manufacturing stage II; see Figure 1). Throughout pre-processing, the separation of cream and skimmed milk takes place as well as the standardization of their mix to achieve the intended fat and no-fat-milk solids (NMS) in the final product. The other processes within the pre-processing stage are raw milk reception, homogenization, pasteurization and separation of cream/skimmed milk. The evaporation of the skimmed milk also takes place before the standardization, adjusting the NMS concentration of milk. Along the second manufacturing stage, sugar and cocoa powder are added enhancing taste and flavour. Egg yolk is also added, but only to the premium version, stabilizing the mix. The second manufacturing stage includes additionally pasteurization, freezing and hardening. Finally, ice cream is packaged and stored in freezers.

Packaging: All packaging materials used throughout the life cycle were included within the packaging stage; this includes primary, secondary and tertiary packaging as well.

Retail: It is assumed that manufacturers transport the finished product directly to wholesalers where it is stored for seven days (Marshall et al., 2003) in a freezer until purchased. Energy consumption at the freezer as well as for side utilities was included. Additionally, this paper considers water consumption at supermarket and impacts due to the use of refrigerants.

Consumption: Storage at consumer's freezer for thirty days (Foster et al., 2006) is assumed and the associated impacts evaluated. Additionally, the consumption stage considers energy and water consumption due to cleaning activities of the utensils used.

End-of-life (EOL) waste management: EOL waste management of packaging is based on current UK waste management practices for plastics: 80% is landfilled and 20% incinerated.

Transport: All transportation steps are included in the study (see Figure 1).

Two types of chocolate ice cream are considered: regular and premium. Their composition, detailed in Table 1, is in accordance with the European average ice cream composition (European Commission, 2006).

Table 1: Composition of regular and premium chocolate ice cream (Marshall et al., 2003)

Ingredients	Chocolate regular	Chocolate premium
Fat	10%	16%
No-fat-milk solids (NMS)	12%	8%
Sugar	16%	19%
Cocoa powder	3%	3.5%
Egg yolk	0%	1.4%
Stabiliser-emulsifier	0.3%	0%
Water	58.7%	52.1%

The life cycle inventory data have been obtained from manufacturers, literature and the Ecoinvent V2.2 database (Ecoinvent, 2010). For the manufacturing stage, two main assumptions were made. Firstly, skimmed milk and cream are the only products derived from raw milk after the separation process. Secondly, the impact of the co-products has been allocated according to their economic value.

GaBi V 6.1 (Thinkstep, 2014) was used to model the system and estimate the GWP, following the ReCiPe 2008 methodology (Goedkoop et al., 2013). GWP due to land use change was also considered. Finally, Gabi V6.1 and @RISK7 (Palisade, 2015) software were used to perform Monte Carlo simulation to examine uncertainties in the model parameters. The results are presented and discussed in the next section.

3. Results and Discussion

As shown in Figure 2, the total GWP is estimated at 6.5 and 6.9 kg CO₂ eq. per 1 kg chocolate ice cream for the regular and premium versions, respectively. The raw materials contribute the majority (74%) of the impact, followed by manufacturing (10%), retail and packaging (6-7%); see Table 2. The share of transportation is small (3%) and that of consumption and EOL negligible (0.3-0.7%).

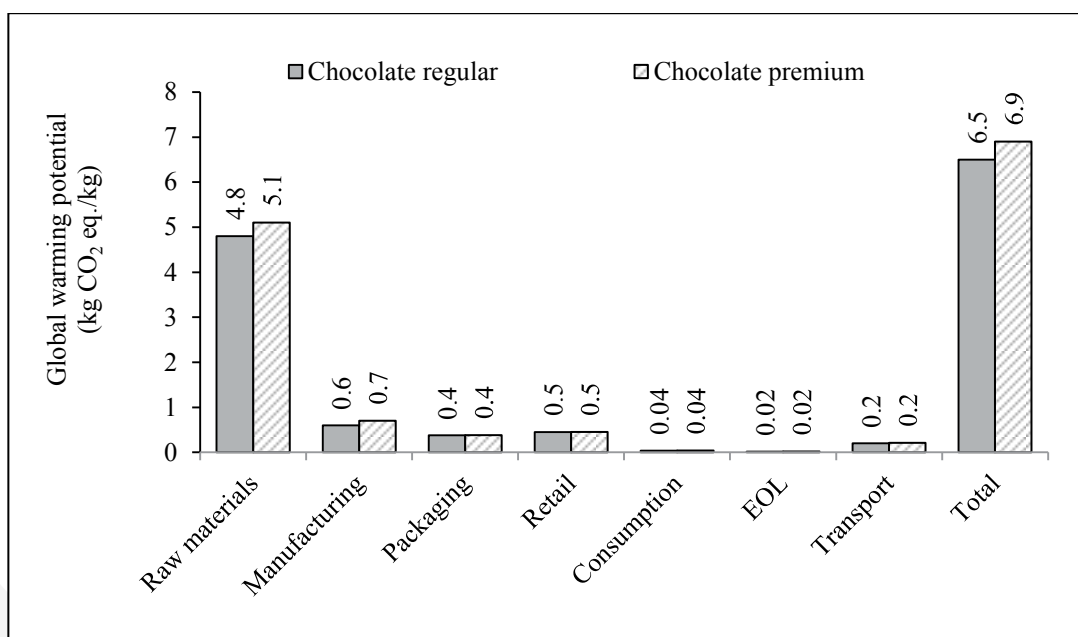


Figure 2: Global warming potential of regular and premium chocolate ice creams

Table 2: Contribution of different life cycle stages to the GWP of ice cream

Life cycle stages	Chocolate regular	Chocolate premium
Raw materials	74%	74%
Manufacturing	9%	10%
Packaging	6%	5.5%
Retail	7%	7.1%
Consumption	0.7%	0.6%
End-of-life waste management	0.3%	0.3%
Transport	3%	2.5%

Regarding the ingredients, cocoa is the main hotspot (2.7 kg CO₂ eq./kg), followed by milk (2-2.1 kg CO₂ eq./kg). For the former, this is due to cocoa cultivation and the associated land use change. For the latter, it results from manure storage, enteric fermentation and production of animal feed.

In the manufacturing stage, electricity consumption is the key contributor to GWP (86% of the total impact from manufacturing). Electricity consumption during refrigeration is also the primary source of GWP in the retail stage; its share is about 93% of the total GWP from retail. Leakage of refrigerants does not affect GWP since ammonia has been assumed as refrigerant.

The results of the uncertainty analysis, obtained by Monte Carlo simulation, are depicted in Figures 3 and 4. Uniform distributions were assumed for all the parameters included in the analysis. As indicated in

Fig, possible variations in the GWP are between 6.1 and 6.9 kg CO₂ eq./kg for the regular and 6.4-7.2 kg CO₂ eq./kg for the premium ice cream.

To compare the impacts of premium to the regular version, a further Monte Carlo simulation was performed (Figure 4). The probability of the premium ice cream having higher GWP impact than the regular was estimated at 92%.

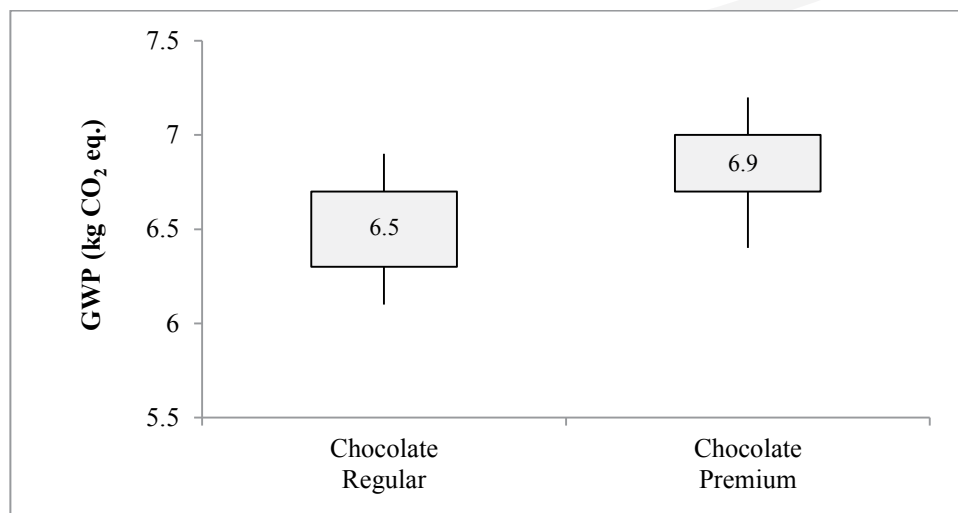


Figure 3: Fluctuation in GWP due to parameter uncertainties

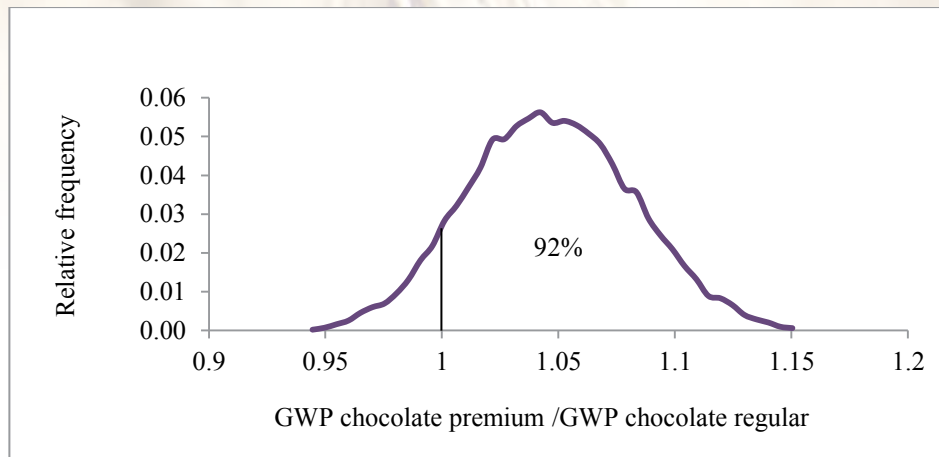


Figure 4: Probability of the GWP being higher for premium than regular ice cream

4. Conclusions

This study estimated GWP of chocolate ice cream, considering the regular and premium versions of the product. The impact is estimated at 6.5 kg CO₂ eq./kg for the former and 6.9 kg CO₂ eq./kg for the latter. Mitigation measures need to focus on the most significant life cycle stages: raw materials, manufacturing and retail. The raw materials account for 74% of the total GWP, which can largely be attributed to the farming activities. The production stage is responsible for a further 9-10% of the impact, most of which is due to electricity consumption. Finally, the retail stage adds further 7%, almost all of which is due to electricity consumption during refrigeration. Therefore, the improvement options include improved agricultural practices, energy saving measures in manufacturing and more efficient refrigeration.

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247. LCA case study of usual and prebaked industrial bread

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ABSTRACT

The consumer expectation to buy fresh bread even short before closing time lead to the use of prebaked bread that is crisped up in the shops on demand. Being aware, that the quick-freezing of prebaked bread as well as the maintaining of the cool chain need a lot of energy, JOWA AG wanted to know whether this prebaked bread conflicts their sustainability strategy from an environmental perspective. The goal of this study was to compare the environmental impact of usual industrial bread and prebaked industrial bread. Data for both the usual and prebaked bread were collected for every single process step from cradle to grave including bread losses (production, point of sale (POS), consumer stage). Different environmental impacts were analyzed. However, for the interpretation of the environmental impacts single score results of the ecological scarcity method 2013 and of the ILCD method were used. The agricultural production of the grains contributes most to the results (up to 85%). The environmental comparison of the whole chain leads to the conclusion that even though there is extra energy demand of the cool chain, the prebaked bread performs at least equally than usual bread due to more efficient production, lower bread losses at the POS and consumer stage (prebaked bread crisped up at the shop stays longer fresh at home). The use of prebaked bread meets the consumer expectation of fresh bread even short before closing time without conflicting the sustainability strategy of the retailer as the environmental impact of prebaked bread is not higher than of that of usual bread.

Keywords: bread, frozen, process chain, losses,

1. Introduction

JOWA AG, the industrial bakery of the Swiss retailer Migros plans an additional bread production factory design for prebaked bread in order to meet the consumer's expectation of buying fresh bread even short before closing time. It is expected that the energy demand is much higher due to the involved cooling chain of the prebaked bread. Stakeholder as well as consumer groups therefore suspect a possible conflict of this strategic alignment with the sustainability strategy of Migros. The JOWA management wanted to better understand the environmental hotspots and impacts of the prebaked bread production line compared to the usual production line of industrial bread.

2. Goal and Scope

Goal and Scope

The goals of the study at hand are:

- Detailed LCA analysis of the production of prebaked bread including also losses at the consumer stage.
- LCA comparison of prebaked bread with usual bread from industrial bakery on
 - Gate to gate: core processes only
 - Cradle to gate: processes up to point of sale
 - Cradle to grave: whole process chain including losses.

The functional unit was

- 500 g of bread (semi-white), at the point of sales (POS), for gate to gate and cradle to gate analysis
- 500 g of bread (semi-white), consumed, for the cradle to grave analysis

The compared breads are sold under the same name and have the same weight. But they have slightly different nutrient and energy contents, mainly because the prebaked bread has somewhat higher water content and some rye whey is used. Therefore, as a sensitivity analysis the nutrient density approach was used (Drewnowski, 2005) which considers different nutrients such as calories, proteins, fat, vitamins and trace elements. This approach allows to compare the breads based on equal nutritional value (see also Kägi et al. 2012).

Definition of bread variants:

Usual bread:

Bread that is baked by an industrial bakery at night or early in the morning and then distributed to the stores.

Prebaked bread:

Bread that is prebaked by an industrial bakery and shock-frozen immediately after baking. It is then stored in cold storage, distributed to the stores and crisped up in the stores on demand.

System boundary

All processes from cradle to grave were considered (figure 1) with a special focus on the in-house processes of the bakery and the losses at the point of sale (POS) and at home due to different storage life of the breads.

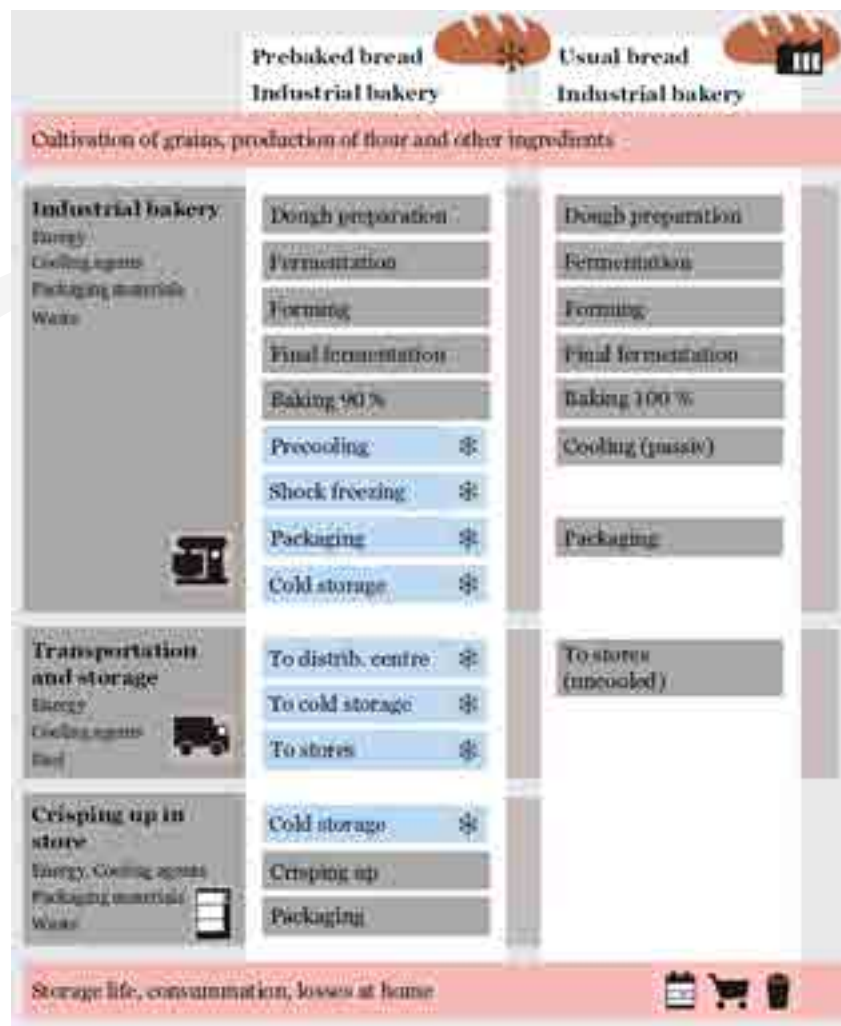


Figure 1: System boundary of prebaked and usual bread

Data / Inventory

Data for the production of the ingredients such as wheat grains, salt etc. was taken from the ecoinvent database v2.2 (ecoinvent 2010). Data for both the bread production were collected for every single process step for the year 2014. Data is based on real measurements at the production lines or yearly statistics of the Swiss bakery. Furthermore, distribution and storage data were collected as well

as data for crisping up at the store and bread losses during production, POS and consumer stage. Losses at the consumer stage were derived by a study of Englert and Dohner (2015).

Impact Assessment Method

Different environmental impacts were analyzed such as climate change, acidification, eutrophication, photochemical ozone formation, ecotoxicity, human toxicity, land use, water resource depletion and mineral, fossil & renewable resource depletion. However, for efficient decision support for the retailer's management (Kägi et al., 2016) single score results were used for the interpretation of the environmental impacts. We used the ecological scarcity method 2013 (Frischknecht & Büsler Knöpfel, 2013) and the ILCD method (JRC, 2011) with the weighting scheme proposed by Huppes and van Oers (2011).

Furthermore, an uncertainty analysis was performed in order to see whether there were significant differences.

3. Results

3.1 Analysis of core processes

In a first step, the management was very much interested in the analyses of the core processes only (gate to gate) neglecting the production of the ingredients and the losses at home.

This analysis (figure 2) shows that the prebaked bread has a factor 2 higher environmental footprint due to the electricity demand for the cooling chain and due to the secondary packaging for transportation. Whereas usual bread is transported in multipath containers, the frozen prebaked bread is transported in one-way cardboards (about 50 g of cardboard is used per 500 g bread). Heat for baking is much higher for the usual bread because the prebaked bread can be produced much more efficient on the production line without product changes during production time. The bread demand of several days or weeks can be produced in one batch whereas for the usual bread only the demand of one day can be produced at once.

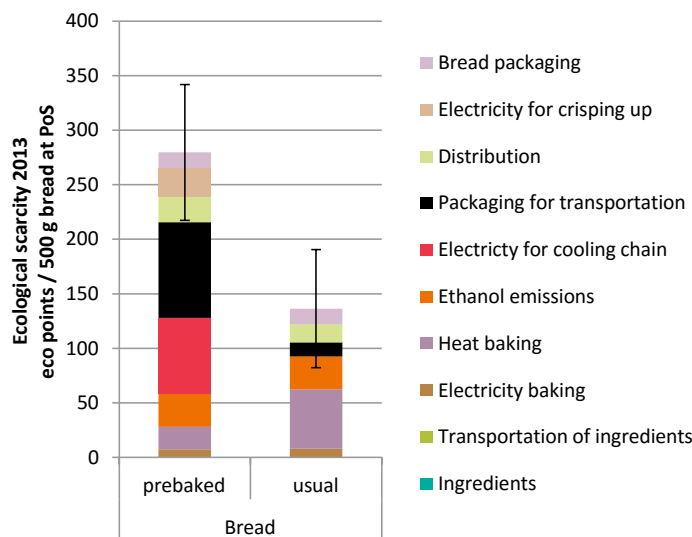


Figure 2: environmental footprint of bread at the point of sale but core processes only (gate to gate).

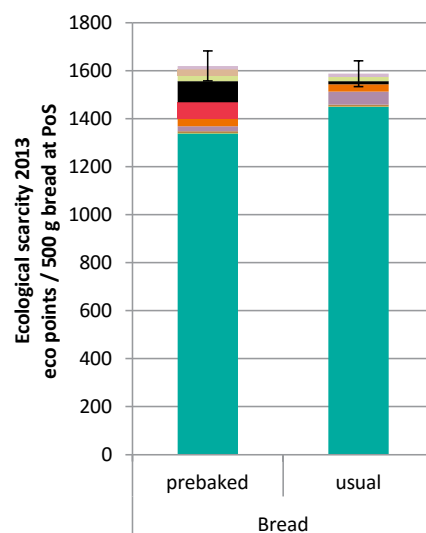


Figure 3: environmental footprint of bread at the point of sale (cradle to gate).

3.2 Comparison at the point of sale (POS)

The comparison at the POS (cradle to gate) shows the relevance of the ingredient production (figure 3). The agricultural production of the grains contributes most to the results (up to 85%). The two bread variants do not show any significant differences anymore. Even though the core processes of the prebaked bread have a twice as high footprint, it needs slightly less ingredients per 500 g bread at

the POS because it has slightly higher water content due to less water evaporation during baking (it is only baked to 90 % and then shock-frozen. During the crisping up the evaporation is less relevant).

3.3 Comparison per consumed bread, at home

The environmental comparison of the whole chain (cradle to grave) shows that even though there is extra energy demand for the cool chain, the prebaked bread performs slightly better than usual bread due to lower bread losses at the POS and consumer stage (figure 4 and 5). At the POS the bread losses are lower because the prebaked breads can be crisped up on demand whereas the amount of usual bread has to be preordered daily by the stores from the industrial bakery. In order to fulfill the consumer's expectation of fresh bread even short before closing time with usual bread only, rather too much than too little bread is ordered leading to quite high losses¹. At home, prebaked bread crisped up at the shop stays longer fresh and less bread is lost in average. This fact was also proven by a blind degustation study (Englert and Dohner 2015).

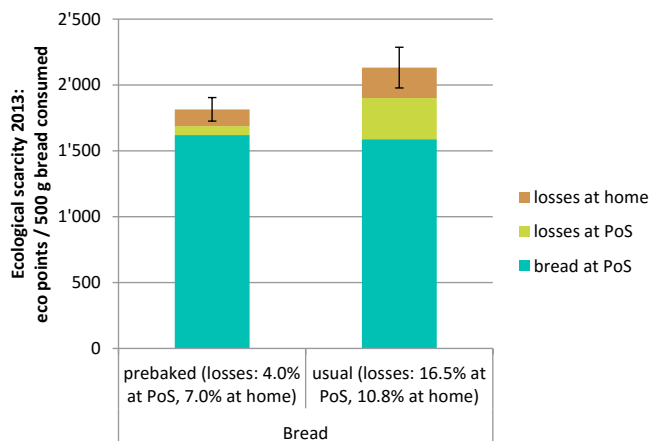


Figure 4: Environmental footprint of bread consumed at home using the ecological scarcity method 2013.

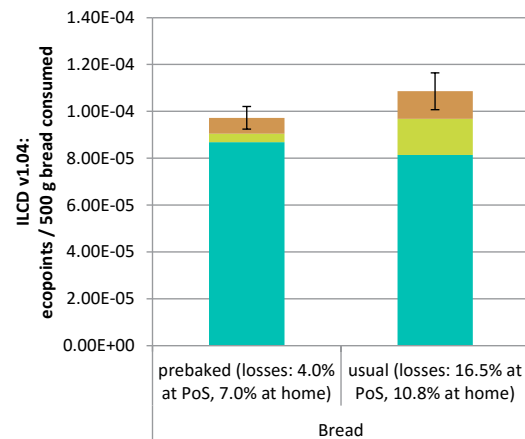


Figure 5: Environmental footprint of bread, consumed at home using the ILCD v1.04 method with the weighting scheme of Huppes and van Oers (2011).

3.4 Sensitivity analysis per equal nutrient density

As a sensitivity analysis the results of the two breads were adjusted to their nutrient density allowing a comparison per equal nutritional value. Figure 6 shows that the results do not change compared to the comparison per equal weight. The reason is that even though the two bread are sold identically with the same product name the prebaked bread contains some rye flour whereas the usual bread contains only wheat flour (due to the differences in the process chain). Rye flour has a higher nutrient density than wheat flour. So the prebaked bread shows an almost equal nutrient density per 500 g even though it has higher water content.

¹ The goal of this study was to compare usual with prebaked bread. However, nowadays most stores already use a combination of usual bread during the day and frozen prebaked bread in the afternoon in order to reduce losses. This combination leads to equally high losses as using prebaked bread only.

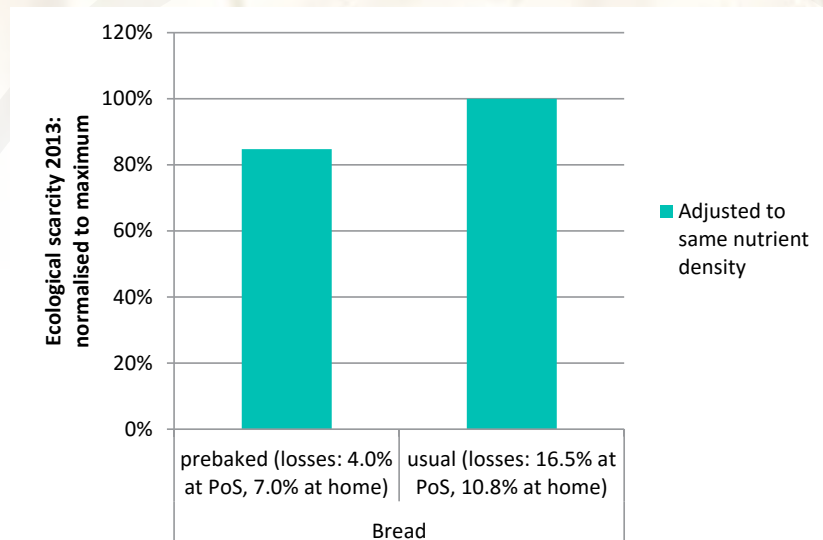


Figure 6: Environmental footprint of bread, consumed at home, adjusted to the same nutrient density value.

4. Discussion

The hypothesis that the additional energy use for the cooling chain may show a higher environmental burden is only true for the gate to gate analysis of the core processes. If the whole process chain is included, this disadvantage disappears as other processes such as the cultivation of wheat show a much higher contribution to the environmental footprint than the core processes and the prebake bread contains slightly less flour and more water.

It should be kept in mind, that in this case study the prebaked bread can be produced more efficient than the usual bread. Furthermore, the results show how important it is to include even the losses at the consumer stage, as this significantly influences the results. A very important aspect is the storage life of the bread at home.

The sensitivity analysis showed that even though prebaked bread has slightly higher water content and therefore lower flour content, the nutrient density does not differ. This is because the prebaked bread contains a small fraction of rye flour which has a higher nutrient value than wheat flour.

The sensitivity analysis with the single score of the ILCD method showed very similar results to the ecological scarcity results leading to the same outcomes and conclusions.

5. Conclusions

The use of prebaked bread meets the consumer's expectation of fresh bread even short before closing time without conflicting the sustainability strategy of the retailer as the environmental impact of prebaked bread is not higher than that of usual bread. However, if the same bread variety is produced with prebaked bread, its production would not be as efficient anymore due to shorter batches. Furthermore, the stores already combine usual bread during the day with prebaked bread during the afternoon till closing time. With this strategy the consumer's expectation is met without too much losses at the POS and still having a broad assortment of breads during the day. To use only prebaked bread would not further reduce the losses at the POS but it would lead to a loss of bread variety in the stores.

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5. The production of pomegranate (*Punica granatum*) in an arid coastal area in Peru: can the cultivation site attain carbon neutrality?

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ABSTRACT

Pomegranate (*Punica granatum*) cultivation worldwide has increased in the past few years due to the growing perception that this fruit has numerous medical and medicinal benefits. Despite a number of studies that analyze the properties of the fruit of this deciduous tree from a medical and dietary perspective, environmental assessments are yet to be conducted in depth. Hence, the main objective of this study is to present the life-cycle environmental impacts derived from the cultivation, processing and distribution abroad (e.g., The Netherlands, Cyprus and Canada) of fresh pomegranate grown at an innovative farm in a hyper-arid area in the region of Ica, 250 km south of Lima (Peru). The Life Cycle Assessment method was used for this study and international standards related to its implementation were considered. Data acquisition was performed at the cultivation site and supported by the ecoinvent® database, while environmental impacts were modelled using a variety of methods, including the IPCC 2007 model or ReCiPe. For the global warming potential impact category, biogenic carbon sequestration was included in the assessment, using the following two models: the first one modelled the aerial carbon sequestered by the pomegranate trees, and the second, used the IPCC Soil Carbon Tool for soil storage. The main results from this analysis demonstrate that on-site GHG emissions can be mitigated to a great extent in the first years of production thanks to biogenic carbon sequestration. However, through time this tendency is reverted and in years of maximum pomegranate productivity GHG emissions are estimated to outweigh those linked to sequestration, despite the relevant minimization of emissions when using innovative irrigation schemes as compared to the conventional flood irrigation in the region. For the remaining impact categories, water depletion was relevant due to the use of this resource in an area where water is scarce, as well as the eutrophication potential due to emissions of fertilizing agents. Interestingly, despite the threat in terms of water depletion and security, the expansion of Peru's agricultural frontier in hyper-arid areas appears to be a feasible strategy for carbon fixation, although current agricultural practices, such as the use of machinery or electricity, need to be optimized in order for the carbon balance to be positive. In addition, important differences in environmental impact were observed depending on the modes of transport considered through the supply chain until the final destination of the product, especially from gate to shelf.

Keywords: GHG emissions; horticultural products; Life Cycle Assessment; Peru

1. Introduction

Peru has experienced a notable agricultural boom in recent decades, with production concentrating mainly along the hyper-arid coast. Hence, this expansion of the agricultural frontier has not translated into a direct threat to forested lands, like in other tropical nations (Meyfroidt et al. 2010; Oré and Damonte 2014). In fact, most land gained for crops was previously either desert land or fallow land. In this context, land use changes (LUCs) are considered a relevant source of GHG emissions, especially when associated with agricultural products (Searchinger et al. 2008; Giampietro and Mayumi 2009; Hertel et al. 2010).

LUC-related environmental impacts are usually quantified for agricultural cultivation sites that have been recently developed (past two decades), and, therefore, constitute a direct consequence of recent LUCs. In fact, one of the main consequences usually is the fact that the carbon cycle of these areas may change substantially. Newly developed cultivation sites tend to translate into a net sequestration of carbon in the biosphere, or to the emission of previously stored carbon in the form of carbon dioxide, depending on the net change in total vegetation due to LUCs (Vázquez-Rowe et al. 2014). This is a critical issue in tropical countries given that LUCs usually entail the deforestation of areas with high carbon stock sequestered in the forest canopy and in tropical soils (Bala et al. 2007).

Based on this discussion, the main objective of this study was to determine the actual environmental impacts of cultivating different types of horticultural crops in the hyper-arid coastal region of Peru. Although the project included the analysis of green asparagus (Vázquez-Rowe et al. 2016a) and blueberry, the present study focuses on a newly constituted pomegranate (*Punica granatum*) site due to a higher potential to sequester carbon. However, despite the importance of

LUCs and carbon sequestration in terms of greenhouse gas (GHG) emissions, results for other anthropogenic-induced environmental pressures are also presented.

2. Methods

The main goal of the study was to estimate the final GHG emissions linked to the cultivation, processing and distribution to final destination of pomegranate produced in the region of Ica, 250 km South of the capital (Lima), including biogenic carbon sequestered at the agricultural site under a high-frequency intermittent drip irrigation system (HFDI system), as well as its comparison to a conventional drip irrigation system.

The ISO standard 14040 was used to conduct the Life Cycle Assessment (ISO 2006). The function of the system was the delivery of a certain amount of fresh pomegranates to the final point of destination in international markets. For this, a 3.9 kg box of pomegranates ready for regional distribution in the country of destination was selected as the functional unit, since this was the only box format used by the company.

The production of pomegranate was analyzed in four different years of cultivation, from 2011 to 2014, which correspond to the first 4 years of occupation of a 51-ha area in the district of Paracas, approximately 260 km South of Lima (see Table 1). The system boundary was limited to the cultivation site, the transport to the processing plant, the packaging at the processing plant and delivery of the final product, first to the port of Callao, and then internationally by marine or air freight.

Table 1: Operational characteristics of the cultivation site in the period 2011-2014. Source: Vázquez-Rowe et al. (2016b)

	Unit	2011	2012	2013	2014
Characteristics of the site					
Surface area	ha	51	51	51	51
Number of trees per hectare	p	420	420	420	420
Commercial type of pomegranate	WONDERFUL				
Pomegranate yield	t/ha	1.61	4.34	13.56	21.84
Total pomegranate production	t	82.0	221.1	691.6	1113.6
Marketable pomegranate production	t	73.8	199.0	622.4	1002.2
Pomegranate destined to local sales	t	0.0	2.2	48.4	100.2
Water use	m ³ /ha	2456	3967	5958	6259
Electricity consumption at cultivation site	MWh	137.69	133.16	201.65	256.02

ha hectare, p piece, t metric ton

Primary data were retrieved from the pomegranate producers. The data provided covered most processes in the foreground system, such as the use of fertilizers, the use of plant protection agents or the use of diesel in the use of machinery. Moreover, the company provided information regarding the source of most raw materials, yields in the different years under evaluation, types of packaging materials and final destinations of sold production. Background data were obtained from the ecoinvent® 3.01 database (Weidema et al. 2013) and modified to adjust to national conditions in Peru.

The assessment methods used to compute the results were the IPCC 2007 GWP method and ReCiPe midpoint (IPCC 2007; Goedkoop et al. 2009). For the determination of GHG emissions using the IPCC method, biogenic emissions and capture processes, including land use changes (LUCs), soil

carbon storage, carbon storage in products (including biomass) and non-CO₂ GHG emissions and removals from livestock, manure and soils (ISO 2013), were included.

Biogenic carbon sequestered in the soil was modelled through the use of the IPCC Soil Carbon Tool (IPCC 2015), which determines the annual carbon stock change in a dynamic manner. For this particular case, it was assumed that the land used for pomegranate production shifted from desert land to long-term cultivated. Sandy soil conditions, as well as a tropical dry climate were included in the model (Zhang and Shao 2014). Aerial carbon sequestered by the pomegranate trees was considered based on several parameters that were retrieved from multiple bibliographical sources (Clark et al. 1986; Myers and Goreau 1991; Birdsey 1992; DeWald et al. 2005). A model, described in Vázquez-Rowe et al. (2016b) was created to account for aerial and root sequestration of the trees. All biogenic carbon was allocated evenly to the boxes of pomegranate produced through the 13 year timeline that was assumed for pomegranate trees prior to being replaced. As recommended by the British Standards Institute (BSI 2012) the total production of pomegranate in this timeframe was calculated thanks to the yield estimations established by the on-site specialist.

Water for irrigation was obtained from aquifers in the area and pumped to the surface using electricity. The electricity mix for Peru was used for calculations, based on the description provided by Vázquez-Rowe et al. (2015). On field fertilization emissions were computed based on the emission factors described in Vázquez-Rowe et al. (2016a).

3. Results and Discussion

Overall GHG emissions, excluding biogenic CO₂ per FU, added to 4.35 kg CO₂eq in year 2014, as shown in Fig. 1, if a weighted average in the distribution stage to final destination is assumed. If biogenic carbon is considered within the final computation, this number decreases considerably to 3.62 kg CO₂eq. Hence, this implies that approximately 17 % of the GHG emissions engendered in the supply chain are mitigated through on-site direct carbon sequestration. In other words, approximately 602 g CO₂eq of the total of 2374 t of CO₂ sequestered at the site due to aerial and root sequestration in the 13 years of production were allocated to each pomegranate box, as well as an additional 132 g CO₂eq due to soil carbon storage.

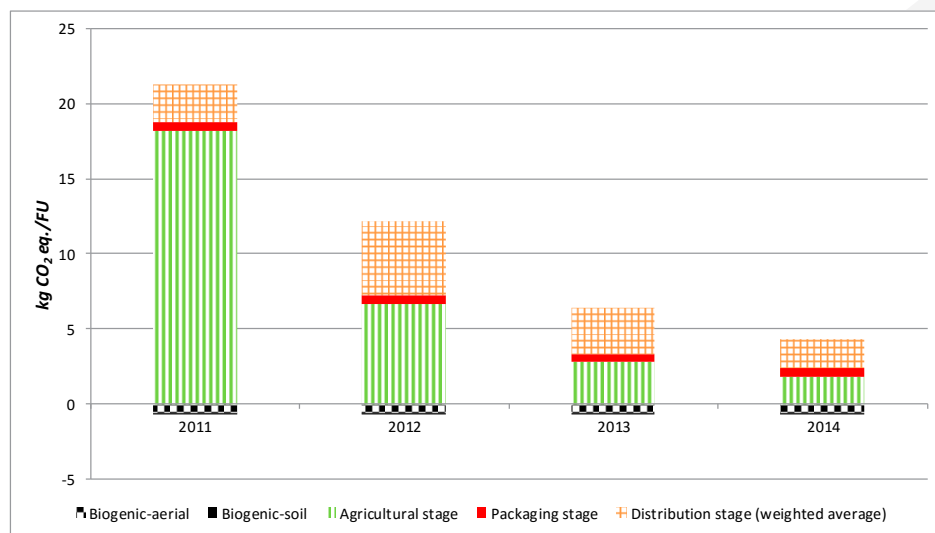


Figure 1: Annual GHG emissions in the agricultural stage of pomegranate production (data per FU= 3.9 kg box of fresh pomegranate)

Following this sequestration perspective, which is considered conservative as compared to other approaches (see Vázquez-Rowe et al., 2016b), the first years of production with low production yields show a much higher final carbon footprint: 20.61 kg CO₂eq per FU in 2011, 11.45 kg CO₂eq in 2012 and 5.67 kg CO₂eq in 2013.

When comparing the environmental impacts of the HFDI system under study and a conventional pomegranate irrigation system (see Fig. 2), GHG emissions are reduced by 15.8% thanks to the lower amount of fertilizers used and the reduced amount electricity used to pump water for irrigation. These same activities manage to reduce water depletion from 6.38 m³/FU to 4.73 m³/FU (26%) and particulate matter formation by 20.1%. Hence, the HFDI system implemented on site appears to show interesting reduction potentials that if applied to other agricultural sites in the area could help reduce the dependence on scarce water resources.

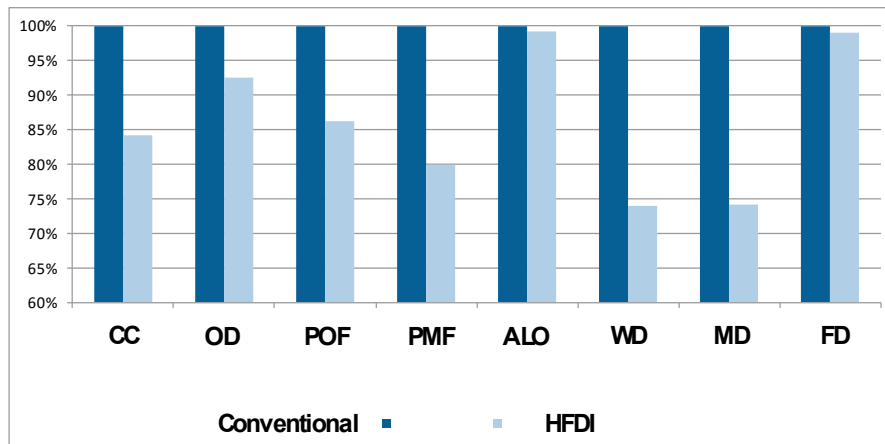


Figure 2: Environmental impacts of pomegranate production at the high-frequency intermittent drip irrigation (HFDI) system site as compared to conventional production of pomegranate

5. Conclusions

Pomegranate has become an increasingly important agricultural product for the Peruvian economy. The greater part of its production areas concentrate in extremely dry areas with hyper-arid conditions. This fact implies that its production is a constant challenge regarding water stress (World Bank 2008). Despite this threatening circumstance to local conditions, it should be noted that the Peruvian agricultural frontier in areas with little vegetation cover has continued to expand through the past decades, reducing stress over the carbon stocks of tropical forest areas in other regions (Asner et al. 2010, 2012). In fact, the results obtained in this study highlight the benefits of expanding the agricultural frontier with perennial trees such as pomegranate. On the one hand, this type of species need a lower amount of water per hectare as compared to other popular agricultural products in this area, such as cotton or green asparagus. On the other hand, pomegranate trees provide an opportunity to store carbon in areas where natural conditions impeded any previous sequestration of CO₂.

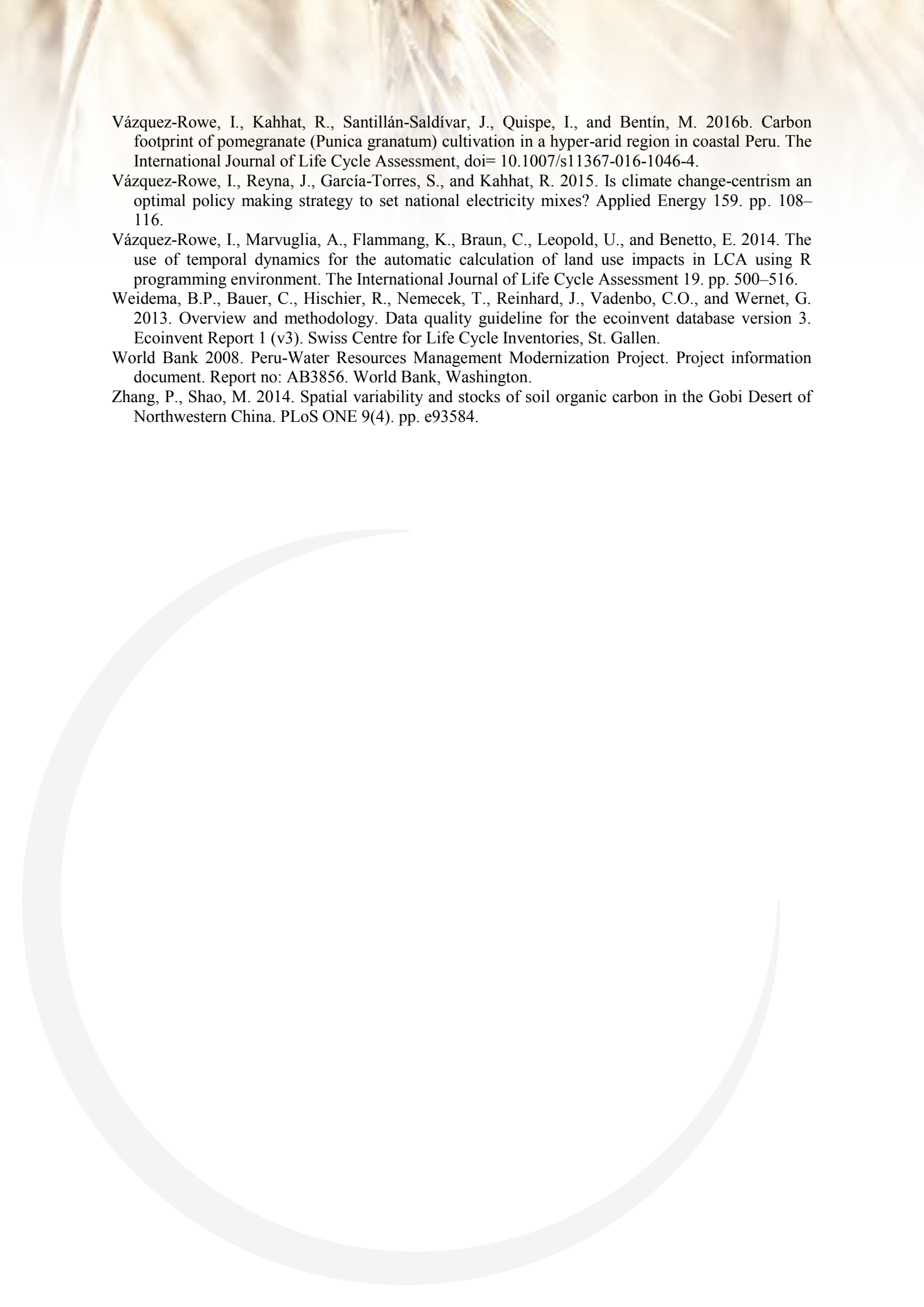
Despite these apparent benefits, it should be noted that numerous improvement actions must be done to improve the carbon balance of pomegranate production. Therefore, changes in the electricity matrix in Peru, as well as the use of fertilizers that are less energy-intensive in their production phase could be important steps forward in this direction.

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4. Using Life Cycle Assessment to determine the environmental profile of *pisco* in different Peruvian regions

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ABSTRACT

Peru destines an important fraction of its production of grapes to the production of its local brandy, named *pisco*. Although production of this distillate concentrates in the Ica valley, five different regions destine vast areas of grape production for *pisco*-making. In fact, *pisco* has become an emblematic drink in Peru among locals and tourists, and is slowly starting to become trendy abroad. This trend implies that vineyards are gradually expanding in hyper-arid areas of Peru, where water is scarce and transport distances can be considerable. In addition, unlike winemaking, the production of *pisco* entails an outstanding amount of grapes to produce one bottle of product. Hence, this study consists of an environmental assessment of the supply chain of *pisco*, including viticulture, vinification, bottling and distribution. For this, Life Cycle Assessment was the selected method with the aim of understanding those material and energy flows and operational activities that engender the most relevant environmental impacts. Therefore, this study identifies improvement actions in this industry on the basis of different winery sizes, geographical regions and production processes. Three wineries were inventoried, which were considered representative for different regions and winery sizes in Peru. Results demonstrate similar trends to those observed in other viticulture and vinification processes. In terms of GHG emissions, the bottling stage constituted an important hotspot, as well as the trellis in viticulture, due to its low lifespan since irrigation is done through flooding. Water depletion, crucial in a hyper-arid area, was magnified by the use of low efficiency irrigation systems, such as flooding, or by the use of inorganic fertilizers, which are intensive in its use. Other categories, such as eutrophication were also identified as important sources of impacts. One of the main findings was that regardless the national framework that regulates *pisco* production; substantial differences were observed through wineries.

Keywords: GHG emissions; horticultural products; Life Cycle Assessment; Peru.

1. Introduction

Pisco is a colorless or yellowish colored brandy that has been produced in Peru since the XV century, becoming an emblematic alcoholic beverage at a national level. In addition, exports have started to increase rapidly in the past decade and they are expected to continue in the following years (PRODUCE, 2014, 2016). *Pisco* is produced in five different regions along the southern coast of Peru: Lima, Ica, where most of the production concentrates, Arequipa, Tacna and Moquegua. In 2014 a total of 22,873 hectoliters of *pisco* were produced and exports translated into economic revenue of approximately 5.5 million USD (SUNAT, 2016).

Grape production concentrates along the relatively fertile valleys that intermittently appear within the coastal Peruvian desert. This area is known for its limited annual rainfall, which usually does not reach 10 mm (SENAMHI, 2015). Therefore, in this area of endemic water stress vineyards rely on water arriving mainly from small rivers and aquifers. Moreover, vineyards cohabit with a series of other crops, most of which rely on considerable amounts of water, such as cotton and green asparagus. Based on this brief description, it is evident that water constitutes an important limiting factor. Despite this situation, agricultural expansion has not ceased in recent years, with new crops expanding across the hyper-arid hills relying ultimately on further water pumping to subsist.

Based on this brief discussion on some of the major environmental problems that occur in areas where *pisco* is produced, we consider that the use of environmental management tools is imperative to understand the entire complex system and propose holistic and integrative ways to solve the environmental impacts. In this context, Life Cycle Assessment (LCA) has been applied to estimate the main environmental burdens linked to the production of *pisco* from a cradle-to-gate perspective with the main aim of analyzing the main environmental hotspots linked to its production and identifying the main improvement actions that could avoid unnecessary environmental impacts.

2. Methods (or Goal and Scope)

LCA was applied using the guidelines provided by the ISO 14040 standard (ISO, 2006). The function of the system was to produce a bottle of the local Peruvian brandy, *pisco*, and ready for distribution for wholesaling in three different wineries in Peru in year 2014. Hence, the functional unit (FU) that was used, in line with previous LCA studies analyzing alcoholic beverages, was one 500 mL bottle of *pisco* at the gate of the winery. System boundaries included viticulture, vinification, distillation and bottling and packaging of the final product (see Fig. 1). Vine nursing was not included within the boundaries due to the lack of data for this stage.

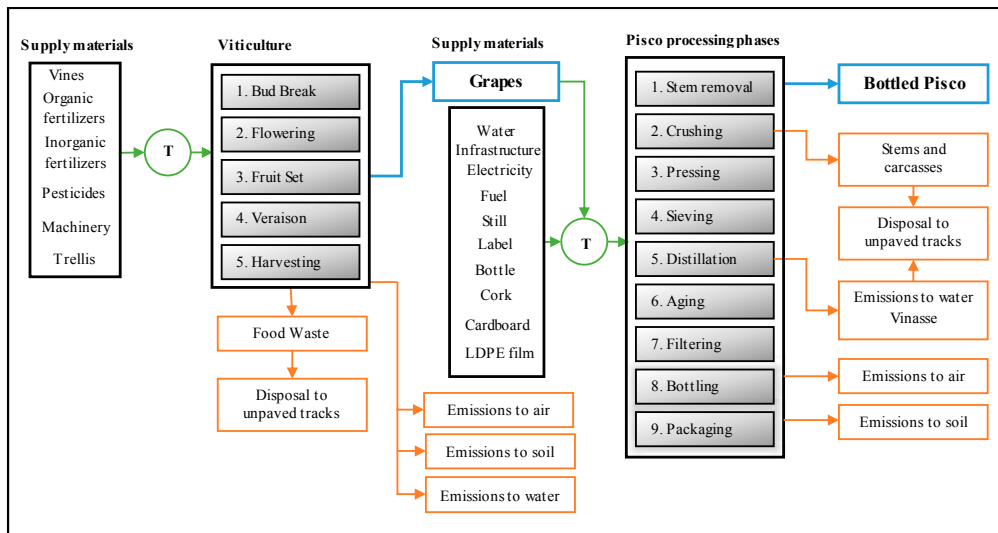


Figure 1: System boundaries of *pisco* production

Data were acquired through questionnaires in three different wineries. Although data quality was outstanding for most operational activities, it had to be completed with bibliographical sources in some cases. The general data for each winery can be found in Table 1. In the viticulture stage, the fertilization emissions on field were modelled based on the recommendations of different bibliographical sources. On the one hand, N_2O emissions were modelled based on the IPCC guidelines (IPCC; 2006). On the other, NH_3 , NO_x and NO_3 emissions were modelled following the indications provided in Nemecek (2013). Pesticides were included using the recommendations available in Nemecek and Kägi (2007) for their production phase. However, as mentioned below, their emissions were not included given that toxicity indicators were not considered in the assessment. For the production of *pisco* at the wineries, the emission factors of different air pollutants were modelled depending on the energy carrier used for distillation: wood, liquefied petroleum gas (LPG) or diesel. In addition, electricity consumption was modelled based on the electricity mix for 2013 provided by Vázquez-Rowe et al. (2015). The Life Cycle Inventories (LCIs) were built based on the data gathered. A summary of the LCI is provided in Table 2 referred to the FU.

Table 1: Operational characteristics of the pisco wineries sampled

	Unit	I	II	III
Region	-	Lima	Ica	Arequipa
Size of winery	-	Medium	Industrial	Medium
Grape production (on site)	kg	30 000	345 017	30 750
Grape production (purchased)	kg	60 000	310 515	115 000
Production volume	L	9 800	52 394	22 750
Type of fuel for distillation	-	Wood	Wood	LPG
Electricity consumption at winery	kWh	2 787	6 036	4 556

Table 2: Summary of annual inventory data per functional unit

	Unit	I	II	III
<i>Viticulture stage</i>				
<i>Inputs from technosphere</i>				
Fertilizers				
Organic fertilizer (as N)	g	7.07	0.54	75.52
Organic fertilizer (as P ₂ O ₅)	g	4.47	0.40	32.55
Inorganic fertilizer (as N)	g	195.92	3.63	93.76
Inorganic fertilizer (as P ₂ O ₅)	g	281.63	3.63	218.76
Other inputs				
Wood	dm ³	2.91	8.74	15.10
Galvanized steel	g	-	190.38	304.18
Diesel (machinery)	mg	-	26.84	-
<i>Pisco processing and bottling stages</i>				
<i>Inputs from nature</i>				
Fresh grapes	kg	3.74	3.29	2.53
<i>Inputs from technosphere</i>				
Wood (for still)	m ³	3.57E-4	1.08E-3	-
LPG (for still)	kg	-	-	2.23E-1
Electricity	kWh	1.42E-1	5.76E-2	1.00E-1
Grape transportation	km	2.65E-3	1.27E-2	6.45E-2
Water consumption	m ³	N/R	1.29E-3	N/R
Glass bottle (500 mL)	g	440	412	440
Cork	g	7.50	7.50	7.50
Cardboard	G	41.7	41.7	41.7
Output				
<i>Pisco</i>	mL	500	500	500
Residues				
Vinasse	L	1.50	1.60	1.50

The methods used to compute the results were the IPCC 2013 method for GHG emissions (IPCC, 2013), and ReCiPe midpoint for the remaining impact categories: particular matter formation, water and fossil depletion, marine and freshwater eutrophication and terrestrial acidification (Goedkoop et al., 2009). Toxicity impact categories were excluded from this paper, although their computation is under way in the framework of the project.

3. Results and Discussion

The main results computed for the three wineries are presented in Table 3 for global warming potential (GWP). As observed, in two out of three wineries, the main environmental impacts are associated with the viticulture stage of *pisco* production, with the production and emissions from organic and inorganic fertilizers being the main environmental hotspot. In fact, in winery III, in which the amount of nitrogen from organic and inorganic fertilizers is similar, the GHG emissions linked to organic fertilizers were substantially lower despite the long freighting distances that *guano* and other manure is transported through the Peruvian road network. The use of other raw materials, however, such as different types of plant protection agents, was not as relevant, except for the use of trellis. Most of the latter is made out of wood (mainly eucalyptus, although other more resistant species can be used for the parts of the vine planation system that withhold more weight), which has a lifetime of 4-5 years, as well as galvanized steel or concrete. Machinery was used only in one of the three wineries. More specifically, the industrial size winery used machinery, whereas the other two (medium-sized wineries) presented a very low level of mechanization in the vineyards.

Table 3: Global warming environmental impacts for pisco production in each individual winery (results reported per functional unit)

	I	II	III
Viticulture			
Inorganic fertilizers	1.26E+00	9.68E-02	1.34E+00
Organic fertilizers	1.60E-01	5.62E-02	4.69E-01
Pesticides	2.98E-02	8.29E-04	8.70E-02
Trellis	4.65E-02	5.39E-02	1.22E-01
Machinery	-	5.69E-03	-
<i>Sub-total</i>	<i>1.49E+00</i>	<i>2.13E-01</i>	<i>2.02E+00</i>
<i>% of system total</i>	<i>72.00</i>	<i>27.04</i>	<i>57.01</i>
Winery			
Electricity consumption	4.80E-02	2.05E-02	3.40E-02
Still	7.60E-03	1.55E-03	3.50E-03
Wood for still	4.70E-03	1.43E-02	-
LPG for still	-	-	8.36E-01
Transport (grape)	2.10E-03	1.34E-02	5.00E-02
Transport (other)	2.85E-03	-	3.25E-02
<i>Sub-total</i>	<i>6.53E-02</i>	<i>4.97E-02</i>	<i>9.56E-01</i>
<i>% of system total</i>	<i>3.15</i>	<i>6.30</i>	<i>26.99</i>
Bottling			
Bottle (production)	4.58E-01	4.56E-01	4.61E-01
Cork, label and cap	2.70E-02	2.67E-02	2.70E-02
Bottle (transport)	3.00E-02	4.34E-02	7.90E-02
<i>Sub-total</i>	<i>5.15E-01</i>	<i>5.26E-01</i>	<i>5.67E-01</i>
<i>% of system total</i>	<i>24.85</i>	<i>66.66</i>	<i>16.01</i>
SYSTEM TOTAL	2.07E+01	7.89E-01	3.54E+00

The second stage of *pisco* production, fermentation and distillation, constituted the stage with lower environmental impacts, although the use of fossil fuels (LPG) rather than wood in the still increases the environmental impacts considerably. Electricity consumption, the production of the still, or the transport of grapes and other raw materials to the winery only represented small contributions to the final impact. In fact, only winery III uses LPG in its distillation process. The difference in relative contribution of the winery stage therefore increases to 27% of the total GHG emissions, whereas in the other two wineries the impacts of this phase are below 7%.

Finally, the bottling stage presents a very high impact related to the production of the white glass bottle. In fact, in all cases the production of the bottle represented approximately 450 g CO₂eq per unit, with small variations depending on the production origin (i.e., electricity mix in Chile or Peru) and transport distances. The remaining inputs in this stage were irrelevant when the entire supply chain from *cradle-to-gate* is analyzed. However, the relative contribution of this stage is highly dependent on the GHG emissions engendered in the remaining stages of *pisco* production. Hence, in winery II, where viticulture and vinification processes are less intensive, the bottling represents 67% of total impact, whereas in winery I it represents 25% and in winery III 16%.

Differences between *pisco* bottles produced in the three wineries are substantial. In the first place, grapes produced in winery II are cultivated in land that has recently become arable land, with high nitrogen and phosphorus contents in the soil. Therefore, the amount of fertilization required for the vineyards was very low as compared to the other two wineries. This explains why the production of a bottle of *pisco* in this winery adds up to GHG emissions of 789 g CO₂eq, whereas in the other two wineries the value increases to 2.07 kg CO₂eq (winery I) and 3.54 kg CO₂eq (winery III). The higher values in winery III are linked to the higher reliance on plant protection agents, the use of abundant quantities of organic fertilizers and, especially, the use of LPG in the distillation process.

For the remaining impact categories, different trends were identified regarding the predominant activities. For instance, in the case of eutrophication, 70% of the total environmental impact of the

vinification stage of the total environmental impact was linked to the vinasse residues, and in the viticulture stage most of these emissions were linked to the on-site fertilizer emissions (PO_4^{2-} and NO_3^-). Regarding water depletion, most of this was associated with direct irrigation, whereas the remaining water use concentrated in the distillation and in the production of chemical fertilizers. For particulate matter formation, NH_3 emissions due to the application of diverse soil amendments, mainly organic and chemical fertilizers. In addition, the use of wood in the trellis and for fuel in the distillation process also accounted for important natural land transformation 70% of the total natural land transformation for the vinification stage.

Results demonstrate that the environmental impacts related to the viticulture stage are in line with the impacts identified in vineyards analyzed in semi-arid areas in Europe, such as Mediterranean Spain or Italy (Rugani et al., 2013; Vázquez-Rowe et al., 2013) On the contrary, substantially lower impacts were computed as compared to impacts found in wet Atlantic areas, such as Galicia or Nova Scotia (Point et al., 2012; Vázquez-Rowe et al., 2012a, 2012b). However, environmental impact values of the final product are substantially higher than in the case of wine products, except in the case of winery II due to unusual fertilization conditions. This trend is related to the lower yield when obtaining *pisco* as compared to wine. In fact, the winery I reported a grape/*pisco* ratio of 9.2 kg/L, winery II 12.5 kg/L and winery III a value of 6.4 kg/L, whereas wine products usually have a ratio of approximately 1.5 kg/L (Vázquez-Rowe et al., 2012a).

Improvement actions, while still under analysis thanks to the inclusion of additional wineries in the sample studied, indicate that the use of inorganic fertilizers increases the GHG emissions of the process considerably, especially due to their production in nations such as China or Russia, with relatively dirty electricity mixes, since Peru does not produce these agricultural amends. However, the main organic fertilizer available in Peru, *guano*, currently has strong extraction restrictions and is not vastly available. The trellis showed relatively low environmental impacts thanks to the avoidance of abiotic materials such as concrete or cement, which are often used in vineyards.

In the vinification stage, the use of fossil fuels in the distillation process implies an important augmentation of the GHG emissions in the entire process, while other energy and material flows in this subsystem are relatively minor as compared to other operations throughout the supply chain.

5. Conclusions

As far as we were able to ascertain, this study constitutes the first full Life Cycle Assessment linked to the production of grape-related products in Peru and one of the first initiatives in the Latin America – Caribbean (LAC) region. Furthermore, despite the numerous LCA studies that have been developed in the wine industry, very few encompass the production of distilled products such as *pisco*. Results demonstrate that the environmental impacts related to the viticulture stage are in line with the impacts identified in vineyards analyzed in other semi-arid areas in the world, although certain operational activities differ to those observed in literature. However, given the low yield of *pisco* from grape production, overall environmental impacts are, in general terms, substantially higher than those observed in the wine industry.

Current research is focusing on analyzing the toxicity impacts linked to the production of grapes to produce *pisco*, given the wide range of different plant protection agents that are used, some of which are forbidden in the US or Europe. For this, PestLCI as adapted to Peruvian conditions, and USEtox are being used to compute the results (Dijkman et al., 2012; Hauschild et al., 2008).

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143. Cradle-to-grave lifecycle impacts of Italian Mozzarella Cheese

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ABSTRACT

A cradle-to-grave life cycle assessment was performed to estimate the environmental impacts associated with Italian mozzarella cheese consumption. Moreover, the difference of impacts between mozzarella produced directly from raw milk and mozzarella produced from purchased curd were highlighted. Data for raw milk production and post manufacturing plant stages were derived from literature and assumptions, while data about mozzarella manufacturing was obtained from one plant which produced both types of mozzarella. The life cycle inventory model was constructed including data from raw milk transport, mozzarella manufacturing, delivery of packaged mozzarella to distribution and retail centers, mozzarella purchase by final consumer, consumption and disposal. Data were attributed to each manufacturing stage based on product-specific information; where specific information was not provided, milk solids allocation was used to allocate the remaining inputs and emissions between mozzarella and the coproducts.

Raw milk production drives the impacts per kg of mozzarella, while electricity usage, packaging and transport contribute a large part of emissions after the farm gate. Mozzarella produced from purchased curd has larger emissions than mozzarella produced directly from raw milk mainly due to transport of curd from different dairy plants, and electricity usage from additional processes at the mozzarella plant to process the curd into mozzarella. The study includes allocation scenario analysis, normalization and uncertainty analysis.

Keywords: Life cycle assessment, environmental burdens, climate change, energy use, mozzarella manufacturing

1. Introduction

Mozzarella cheese is one of the most common dairy products consumed around the world and its production is increasing. It represents a strategic product in the future of the global dairy sector (Koeleman, 2015). Italy is a big mozzarella producer and consumer, reaching 5 kg of mozzarella consumed per capita per year in 2012 (Assolatte, 2012). Mozzarella manufacturing is a complex process, which includes several operations and requires numerous inputs and outputs, resulting in a burden on the environment. It is necessary to assess these environmental burdens in order to improve the sustainability of the mozzarella production chain. This study's goal is to increase the knowledge of environmental consequences of mozzarella production and consumption. The results could be used by national and international producers to improve natural resource use efficiency and at the same time to maintain dairy production as a source of healthy food and revenue.

This study presents a cradle-to-grave environmental impact analysis of mozzarella cheese production and consumption, although strong emphasis is given to manufacturing plant. Secondary, the study highlights the difference between two types of mozzarella: one is manufactured using directly raw milk and the other using purchased curd.

2. Methods

2.1. Goal and scope

The goal was to assess potential environmental impacts associated with production and consumption of mozzarella cheese, with deep emphasis on manufacturing plant, where less knowledge exists on environmental burdens and where many efforts can be applied to increase the sustainability of the production. The scope was a cradle-to-grave impact assessment of Italian mozzarella. The assessment was conducted 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 kg of mozzarella consumed (wet basis, including packaging). The system boundaries begin with the raw milk production at farm, raw milk transport by truck to the manufacturing plant, mozzarella manufacturing, delivery to distribution and retail centers using trucks, consumer purchase, home refrigeration and consumption, and final. The data collected were assigned to mozzarella and coproducts (fat whey, skimmed whey, and whey cream).

2.3. Cradle-to-grave inventory data

Data collected for this work were primarily derived from a survey. The manufacturing plant purchased raw milk from Italian and European dairy farms. The Italian raw milk impacts were represented by a study conducted on 34 Italian dairy farms (Dalla Riva et al., 2015), while the foreign raw milk was modeled using background data from ecoinvent database (Ecoinvent[®] v3.2 default system model by Weidema et al., 2013). During 2015, the third largest Italian mozzarella producer plant was surveyed in order to collect the data about mozzarella manufacturing, based on calendar year 2014. Data collection covered purchased resources (energy, materials, chemicals, water), production (mozzarella and coproducts), and waste streams (solid and liquid). Packaging was classified as primary packaging, which is in direct contact with the mozzarella, and secondary packaging, which is associated with mozzarella delivery and plant purchases. Transport of raw milk to the manufacturer, as well as delivery of final products to distribution center, to retail and to final consumer, was also accounted. All transportation was conducted using trucks, although a small amount of mozzarella was shipped by airplane and ship. SimaPro[®] 8.1 (PRé Consultants, The Netherlands 2014) was used as modeling software; the ecoinvent database (Ecoinvent[®] v3.2 default system model by Weidema et al., 2013) was used for secondary data (all upstream processes and emissions, some transport information and materials like fuels and chemicals). Detailed information about refrigeration, mozzarella shelf-life and consumption phase were deduced from published literature for all the post-manufacturing stages (Barilla, 2015; Broekema and Kramer, 2014; Kim et al., 2013; Quested, 2013; Flysjo, 2011). Packaging waste of consumed mozzarella were included in the analysis.

2.4. Life cycle impact assessment

ReCiPe midpoint (H) V1.11 (Goedkoop et al., 2009), and cumulative energy demand (Frischknecht et al., 2007) were the method to assess six impact categories, climate change (CC), ozone depletion (OD), terrestrial acidification (TA), freshwater eutrophication (FE), human toxicity (HT), photochemical oxidant formation (POF), and three inventory categories, cumulative energy demand (CED), water depletion (WD) and land occupation (LO).

2.5. Allocation

Fat whey, skimmed whey and whey cream were the three coproducts at manufacturing plant, all were source of revenue. Allocation is crucial in LCA studies. Several resources were directly assigned to mozzarella and to coproducts using product-specific information (such as primary packaging assigned only to mozzarella). When product-specific information was not available to assign the resource to the appropriate product (such as raw milk), the default allocation was based on milk solids content of the product. The content of fat, protein, lactose and ash were used to determine the allocation factors. No-allocation (all of resources and emissions were assigned to mozzarella), economic allocation, fat allocation and protein allocation were the scenario analyses to investigate the difference in the final environmental burdens of consumed mozzarella.

3. Results

3.1 Environmental impact assessment results

The results are reported as a contribution analysis by stages and by inputs. The mozzarella lifecycle was grouped into stages: raw milk production, raw milk transport, manufacturing, packaging (production, usage, transport and waste treatment), delivery to distribution center and delivery to retail, consumption and disposal. The inputs were classified into categories: electricity, natural gas, lubricant oils and refrigerant gases, chemicals, water and land. Packaging-related impacts arise from packaging grouped as secondary packaging (plastic and paper) and primary packaging (plastic). Transport-related impacts derived from raw milk transport to manufacturing plant, other purchased inputs' transport to the manufacturing plant, mozzarella delivery to distribution and retail centers, mozzarella transport to house, and waste transport.

In the cradle-to-grave perspective (Table 1), raw milk production is the main driver (48-98%) to overall impacts, except for OD where the main contributor is from the post farm gate stages (83%). Feed production and farm emissions (mainly enteric CH₄) are the main hotspots in raw milk production. Raw milk transport represents the second largest emitter for HT (15%), while it has minor importance (0.1-7%) for the others impacts. The manufacturing stage is the main driver for OD (70%), while it is second for CC and FE (5%), and third for CED (12%), TA, WD and LO. Packaging stage is the second contributor (1-20%) to TA, LO, WD, POF and CED, while it is third for HT (13%), FE and CC. Distribution, retail, consumption and disposal stages have a minor relevance.

Analyzing the farm gate-to-grave perspective (Figure 1), electricity usage is main driver (22-30%) for CED, CC, FE and TA, while it is second (7-19%) for OD, HT and POF. Raw milk transport is first driver for HT (30%) and POF (22%), meanwhile it is the second (17-18%) for TA and CC. Electricity and raw milk transport are the second main contributors (12%) for LO. Secondary paper packaging is main contributor to LO (51.5%), meanwhile it is the second (18%) for FE and CED, and it is third (15%) for HT and WD. Process water (33%) is the main contributor to WD, while wastewater treatment is the second largest for WD (32%) and FE (10%). Mozzarella transport post-plant delivery represents the third contributor (8-18%) for LO, CC, TA and POF. While primary plastic packaging is third contributor to CED (15%). Whereas refrigerant gas is main contributor for OD (76%).

At dairy plant, electricity usage, water consumption, primary and secondary packaging, wastewater treatment and refrigerant gas drive several impact categories (CC, CED, TA, FE, OD, WD). Meanwhile, the post dairy plant stages are mainly determined by electricity usage for cooling and storage of mozzarella, and secondarily by transport. Water usage and wastewater produced during mozzarella consumption stage (mainly the water to wash the dishes and the waste brine where the mozzarella was included to preserve the freshness) are relevant for WD. Instead transport activities are importance for HT and POF.

Mozzarella is frequently produced using curd, which is purchased from other dairy plants. The process to produce curd for mozzarella production is the same as the first stages to produce mozzarella, but after curd ripening, it is packaged into plastic bags and cooled, then delivered to a mozzarella manufacturing plant and the packaged curd is stored. After that, the curd is warmed in hot water, in order to soften it, so it is ready to be stretched and manufactured into mozzarella. In order to estimate the differences in environmental impacts between mozzarella from raw milk (mozz_1) and mozzarella from curd (mozz_2) a scenario analysis was performed using the information about curd production inventoried at mozz_1 dairy plant to model mozz_2 manufacturing. All resource flows were similar to mozz_1 production, and specific mozz_2 stages were added: curd packaging and waste, storage of curd after production and after delivery, transport to the mozzarella plant, warming of curd, and mozzarella cooking in the consumption stage. The main differences between the two mozzarella types arise from different manufacturing operations (64-98%). Not surprisingly, all the impacts increase in mozz_2 production and consumption, and the increases range from 20% to 49% in LO, WD, OD, FE, TA, POF and HT, respectively, and CC and CED increase by 28% and 40%, respectively. Curd transport is the main driver of the increase (48-88%), followed by curd warming (7-32%), storage of curd (4-16%) and packaging (1-7%). Moreover, another significant difference from mozz_1 which is generally consumed fresh (without cooking process), the mozz_2 is commonly used as an ingredient for pizza, so it requires heating and electricity to cook it (29% mass allocation factor to mozzarella), increasing its emissions at the consumption stage (2-36% of the total increase). CC and CED of mozzarella from curd reach about 8.5 kg CO₂eq and about 63 MJ per kg of consumed mozz_2, respectively.

Table 2: Percent contribution of the stages in mozzarella lifecycle, cradle-to-grave perspective.

	CC	OD	TA	FE	HT	POF	CED	WD	LO
Raw milk production	83.2	16.9	96.0	85.1	48.3	81.4	48.4	97.1	97.7
Raw milk transport	3.0	5.0	0.6	1.2	15.5	4.2	7.1	0.1	0.3
Manufacturing	4.8	69.7	1.0	5.3	8.4	2.9	12.2	0.2	0.3
Packaging	4.2	1.4	1.1	3.6	13.4	5.9	20.5	2.2	1.3
Distribution	2.5	4.1	0.6	1.3	7.3	3.4	6.1	0.1	0.2
Retail	1.0	1.4	0.3	1.0	2.7	1.0	2.7	0.1	0.1
Consumption and disposal	1.3	1.5	0.4	2.5	4.4	1.2	3.0	0.2	0.1

CC: Climate Change (kg CO₂ eq); OD: Ozone Depletion (kg CFC-11 eq); TA: Terrestrial Acidification (kg SO₂ eq); FE: Freshwater Eutrophication (kg P eq); HT: Human Toxicity (kg 1,4-DB eq); POF: Photochemical Oxidant Formation (kg NMVOC); CED: Cumulative Energy Demand (MJ); WD: Water Depletion (m³): LO: Land Occupation (m²a).

3.2 Allocation scenario

Different allocation approaches were tested across all impacts with respect to milk solids allocation (default allocation). They all increase the final emissions per kg of consumed mozzarella: No-allocation (54-76%), economic allocation (49-68%), fat allocation (28-40%) and protein allocation (26-37%). CC and CED reach the largest emissions using No-allocation (11.3 kg CO₂ eq and 70 MJ per kg of consumed mozzarella, respectively).

3.3 Normalization

We performed a normalization assessment using the ReCiPe Midpoint (H) V1.11 European normalization (Goedkoop et al., 2009). Average per capita emissions were compared with the per capita emissions from mozzarella consumption in Italy derived from annual Italian mozzarella consumption (288,100,000 kg by Cerved, 2015) divided by the Italian population.

Terrestrial acidification and freshwater eutrophication are the largest normalized impact categories and thus should be areas on which to focus environmental improvements. They both represent 1.3% of the annual acidification and eutrophication impacts, respectively, and derived mainly from use of nitrogen substances such as fertilizer at farm. Applying normalization in a farm gate to grave perspective freshwater eutrophication, human toxicity and terrestrial acidification are the most relevant impact categories, mainly driven by transport activities and electricity usage.

3.4 Uncertainty analysis

The impact results obtained per kg of consumed mozzarella were analyzed using 1,000 Monte Carlo simulation runs. The quality of individual data inputs was assigned using the Ecoinvent[®] 2.0 pedigree matrix (Frischknecht et al., 2007). The 95% confidence interval is 5.9 to 7.9 kg of CO₂ eq and 38 to 54 MJ for CC and CED per kg of consumed mozzarella, respectively. The average emissions are 6.7 kg CO₂ eq and 45 MJ per kg of consumed mozzarella. CC and CED are derived for 14% and 45% from dairy plant to grave prospective, respectively, while 0.5 kg CO₂ eq and 14 MJ are the impact per kg of mozzarella at dairy plant. The water depletion per kg of consumed mozzarella are 0.6 m³. The water depletion from dairy plant gate to grave perspective contributes 3% of total WD, which is 17 L per kg of consumed mozzarella and 83% of these are used during mozzarella manufacturing and packaging stages.

4. Discussion

The study highlights the environmental impacts of the mozzarella lifecycle. Other studies (Palmieri et al., 2016; Kim et al., 2013; EPD, 2011; Nielsen and Hoier, 2009) estimated the environmental impacts of mozzarella cheese supply chains, and they are in agreement with the present

study about the high contribution of raw milk production to the total environmental impacts. The importance of manufacturing stage, followed by distribution and retail stages has been reported by the same authors. Electricity and natural gas, used to produce steam and hot water at the manufacturing plant, and for mozzarella storage, drive several impacts in a farm gate-to-grave perspective, and this is in line with the studies of Palmieri et al. (2016) and Kim et al. (2013). CC, CED, WD and FE are in line with the values estimated for American mozzarella cheese (Kim et al., 2013), although the packaging has limited influence in the American study, while packaging is one of the major impact contributors in the present study. The same large contribution of packaging as a primary driver to several categories and to depletion of resources has been found by EPD (2011). In the present study, the impacts of the transport of raw milk and mozzarella are the result of the use of raw milk purchased from other European countries and the international export of mozzarella. Kim et al. (2013) determined milk transport impact on HT and POF, as in the present study, while Palmieri et al. (2016) and Nielsen and Hoier (2009) found a negligible impact from raw milk transport, due to the short distance from farm to plant. In agreement with this study, Palmieri et al. (2016) and Kim et al. (2013) estimated acidification and eutrophication as the main impact categories to focus environmental improvements when normalization is applied. No other study has been found on the estimation of impact associated to mozzarella produced from curd, so the present study represents the first case study to estimate the impacts associated to mozzarella produced using curd.

Obviously, a reduction of emissions at dairy farms should be the first effort to improve overall sustainability of the production chain. Our study suggests some opportunities to reduce the impacts after farm gate, mainly at the mozzarella manufacturing plant. Electricity and natural gas usage, together with plastic and paper (mainly cardboard boxes) packaging, are the greatest impact drivers. Implementation of energy efficiency measures into dairy plant should be the first focus for impact reduction including: sourcing renewable energy and implementing energy and heat recovery. Packaging could be optimized to reduce the amount per kg of mozzarella, and new more eco-sustainable packaging could be evaluated.

Allocation can significantly influence the reported impact per kg of mozzarella. Climate change ranged from 6 to 11 kg CO₂ eq and cumulative energy demand from 45 to 70 MJ per kg of mozzarella. Moreover attributing the resources among the products partially using product-specific information and partially with normative allocation approaches shows different results than only having resource consumption data at plant level, reconfirming the importance of having as much specific and detailed data in order to estimate the more real final emissions as possible.

Finally, in the preliminary results about the different impacts between the two types of mozzarella, mozzarella produced from purchased curd shows larger emissions than mozzarella produced directly from raw milk. The main differences arise from additional transport of curd, indeed human toxicity is the impact category with the larger impact variation between the two types of mozzarella. It is reasonable to suggest production of mozzarella from raw milk in order to promote sustainability, which avoids additional transport activities and manufacturing process inside the dairy plant. However, a further analysis is suggested to improve the estimation of impacts of mozzarella from curd, so more knowledge will be present about the environmental sustainability of the two types of mozzarella.

5. Conclusions

The study estimates the impact per kg of consumed mozzarella. As expected the raw milk production is the largest contributor to the impacts (48-98%) to climate change, terrestrial acidification, freshwater eutrophication, human toxicity, photochemical oxidant formation, cumulative energy demand, water depletion and land occupation, except for ozone depletion where manufacturing stage is the main driver. Excluding raw milk production from the analysis, electricity usage, packaging and transport are the main hotspots for impacts, except for ozone depletion where refrigerant leakage is the main driver. Wastewater treatment and water use are the main drivers for water depletion. Distribution, retail, consumption and disposal have relatively small contribution to final impacts; the main drivers are transport, cooling and storage of mozzarella. The normalization test suggests terrestrial acidification and freshwater eutrophication as the first impact categories to

focus efforts for improvements, while in a farm gate to grave perspective, eutrophication, human toxicity and terrestrial acidification are the most relevant impact categories related to mozzarella consumption. Mozzarella from curd presents greater emissions than mozzarella from raw milk mainly due to addition transport and manufacturing process inside the dairy plant.

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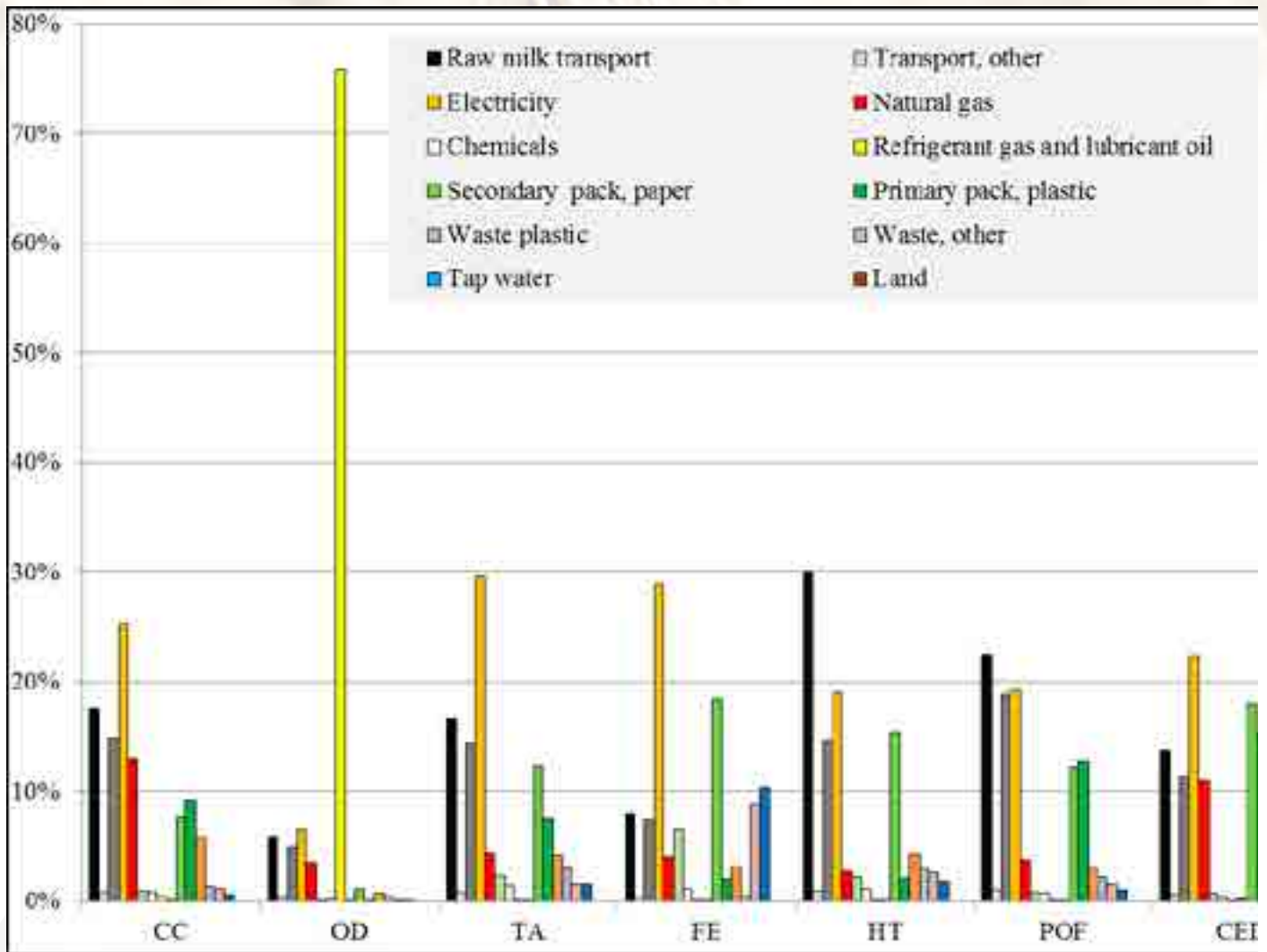


Figure 1: Percent contribution of the inputs in mozzarella lifecycle, post farm gate-to-grave perspective. CC: Climate Change (kg CO₂ eq); OD: Ozone Depletion (kg CFC-11 eq); TA: Terrestrial Acidification (kg SO₂ eq); FE: Freshwater Eutrophication (kg P eq); HT: Human Toxicity (kg 1,4-DB eq); POF: Photochemical Oxidant Formation (kg NMVOC); CED: Cumulative Energy Demand (MJ); WD: Water Depletion (m³); LO: Land Occupation (m²a).

78. Life Cycle Environmental Impacts of Global Consumption of Scotch Whisky

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ABSTRACT

The aim of this study is to provide an analysis of the life cycle environmental impacts of global consumption of Scotch whisky and to identify opportunities for improvements. The focus is on grain spirit, which represents the backbone of blended Scotch whisky consumed worldwide. Four environmental aspects are considered: primary energy demand (PED), carbon footprint (CF), volumetric water consumption (VWC) and stress-weighted water footprint (WF). It has been estimated that the annual global consumption of 513.4 million litres of Scotch whisky (40% v/v) requires 13,100 TJ of primary energy and generates 1.05 Mt CO₂ eq. Its VWC and WF are estimated at 218 million m³ and 1463 million litres eq., respectively. The production stage is the main hotspot for both the PED and CF, contributing 37% and 33%, respectively. Cultivation of grains used for whisky production is the next most significant contributor to the CF (32%) and packaging to PED (32%). The grains also account for 90% of the VWC, while the whisky production is responsible for 50% of the stress-weighted WF. Improvement opportunities explored in the paper include improved agricultural practices, implementing energy efficiency measures and switching to cleaner energy sources in distilleries, as well as increasing the recycled content and light-weighting of glass bottles. The results suggest that switching from fossil fuels to biomass in distilleries has the highest potential to reduce the CF (26%), while glass light-weighting and energy-efficiency measures could reduce the PED by 6% each.

Keywords: carbon footprint; grain whisky; life cycle assessment; primary energy demand; water footprint.

1. Introduction

The Scotch whisky industry is the UK's largest sub-sector within the food and drink sector. With an annual turnover of over £5 billion, it accounts for a quarter of the UK's food and drink exports (SWA, 2015a). Despite the importance of the whisky sector for the UK economy and to consumers worldwide, environmental impacts from Scotch whisky production and consumption are still largely unknown, with the exception of a study which was carried out in 2006, now ten years ago (SWA, 2009). This is the focus of this paper which presents life cycle environmental impacts of worldwide consumption of Scotch grain whisky, the backbone of blended Scotch whisky. Four environmental aspects are considered: primary energy demand (PED), carbon footprint (CF), volumetric water consumption (VWC) and stress-weighted water footprint (WF). A number of options for reducing the impacts are also considered. For simplicity, Scotch grain whisky is referred to as 'Scotch whisky' in the rest of the paper.

2. Methods

The main goal of this study is to estimate PED, CF, VWC and WF impacts of Scotch whisky production and consumption. The analysis is carried out for the annual global consumption of grain whisky using 2014 as a reference year. In 2014, 33.5 million litres (40% v/v) of Scotch whisky were released for consumption in the UK and 480 million litres (40% v/v) were exported worldwide (SWA, 2015b).

Life cycle assessment (LCA) is used to estimate PED and CF of Scotch whisky, following the ISO 14040/14044 guidelines (ISO, 2006a; 2006b). A further aspect considered is the water intensity of whisky production. Both the virtual water content (expressed as the volumetric consumption) and stress-weighted water footprint (WF) are estimated. The VWC of a product is defined as the volume of green and blue water used in its manufacture, measured across the supply chain (Hoekstra et al., 2011). The stress-weighted WF approach followed here is that proposed by Pfister et al. (2009), which incorporates the water stress index (WSI) as a mid-point characterisation factor. CCaLC 3.0 software (CCaLC, 2013) has been used to model the system and estimate the impacts.

As detailed in Figure 2, the system boundary of the study is from ‘cradle to grave’, comprising production of raw materials and utilities, whisky manufacture and its packaging, all life cycle transport and management of in-process and post-consumer waste (recycling, treatment and disposal). The data sources and the assumptions are detailed below.

Grain whisky is produced from three types of grain: wheat, barley and maize. The specific recipe for whisky will differ between producers but the values assumed here are representative of a typical contribution of the three types of grain. Wheat and barley are sourced from the UK and maize from France. Process steam and electricity requirements are met by natural gas-fired boiler and combined heat and power (CHP) plant. Excess electricity is exported to the grid. Typically, 70% of whisky is bottled in clear glass and 30% in green glass bottles (SWA, 2015c). Primary data have been obtained from whisky producers, including the quantities of raw materials, utilities, primary packaging, transport modes and distances as well as in-process waste management options. For the waste packaging, it is assumed that the 32% of bottles are landfilled, considering that in the UK 68% of packaging glass is recovered for recycling (DEFRA, 2016). System expansion has been used for crediting the system for the impacts avoided by displacing the production of the co-products (such as distillers’ dried grains with solubles, carbon dioxide and electricity). Depending on availability, the background LCA data have been sourced largely from Ecoinvent (2010) and CCaLC (2013) and supplemented by data from Cranfield (2007) and AGRIBALYSE (2015). The water requirements for cultivation have been calculated using the CROPWAT 8.0 software (FAO, 2013).

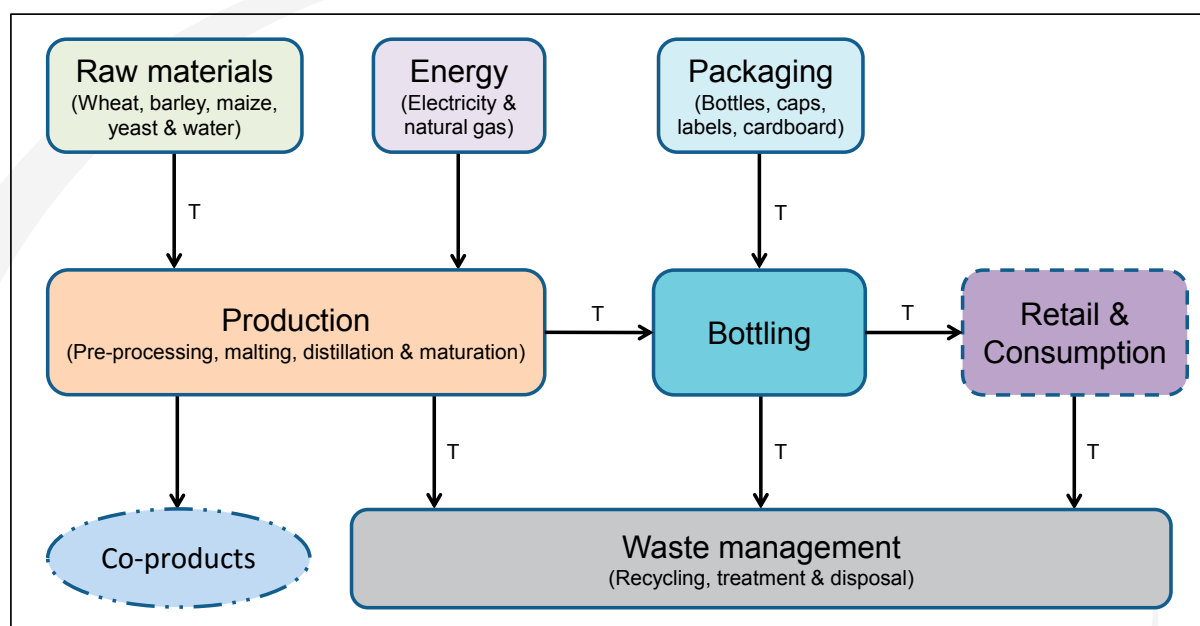


Figure 2 System boundary of the study

[---- Excluded from the analysis. - - - - System credits for co-products. T: Transport.]

3. Results and discussion

The environmental impacts of Scotch whisky consumption are shown in Figure 3. The CF is estimated at 1.05 million tonnes CO₂ eq./yr. As can be seen in Figure 3, the production stage and raw materials (grains) are the major contributor to the carbon footprint, each accounting for around 33% of the total (before the credits for the co-products). This is largely due to the GHG emissions from energy use and wheat cultivation. The contribution of packaging is 23% and is mainly due to the emissions from bottle manufacturing. The rest of the impact is from transport, including shipping (11%) and waste management (1%).

The annual global consumption of 513.4 million litres of Scotch whisky requires 13,100 TJ of PED. The production of whisky is a key contributor, accounting for 37% of PED, mainly due to the consumption of natural gas. The packaging is the second most significant contributor to PED, with 32%, largely due to the use of fossil energy sources in glass manufacturing.

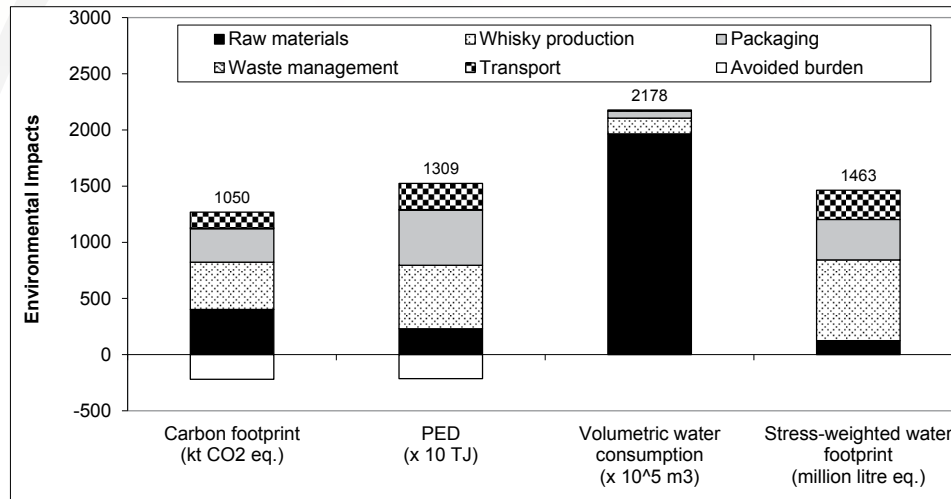


Figure 3: Life cycle environmental impacts of global consumption of Scotch whisky

The amount of water used in the life cycle of Scotch whisky is estimated at 218 million m³. Grains cultivation accounts for the vast majority of this (90%), while whisky production requires a relatively small amount of water (6%). However, the contribution of whisky production to the stress-weighted WF, estimated as 1463 million litres eq., is much higher (50%), while grains cultivation adds only 8%. This can be explained by the fact that in the regions (Scottish Borders) where wheat and barley are cultivated, irrigation of crops is not required, hence the water-stressed WF is low. Similar applies to maize cultivation in Aquitaine (France), which does require some irrigation, but the WSI is low (0.033). Finally, packaging accounts for 25% of the stress-weighted WF, 60% of which is associated with the secondary packaging (cardboard) and 40% with the glass bottle.

4. Improvement opportunities

This section explores improvements opportunities in the life cycle of Scotch whisky, informed by the hotspots identified in the study.

Raw material improvements: Given that the grains are a key contributor to the environmental impacts considered here, the influence of reducing agricultural impacts through better agricultural practices is considered. It is estimated that environmental impacts of the grain crops could be reduced by 10–20% by optimising application of fertilisers, reducing the use of pesticides and irrigation water as well as improving the crop yield (Jeswani et al., 2015). Here, a conservative value of 10% increase in crop yield and associated reduction in impacts is considered. As indicated in Figure 4, by improving agricultural practices, the CF of whisky could be reduced by 4%. This would also reduce PED by 2% and the VWC by 9%.

Process improvements: Scotch whisky manufacture has a long history and is reputed for the traditional process used to make the spirit. Therefore, any process improvement options need to be considered in that context. Nevertheless, one of the options that has been identified as potentially viable is very high gravity (VHG) brewing which could reduce steam consumption from distillation stage by up to 40% (Bell, 2001). Compared to the current operations, VHG brewing could reduce the FC by 4% and PED by 6%. However, this option would require some major investment and

technological changes in the production process, which in turn could lead to changes in product composition and potentially, the taste.

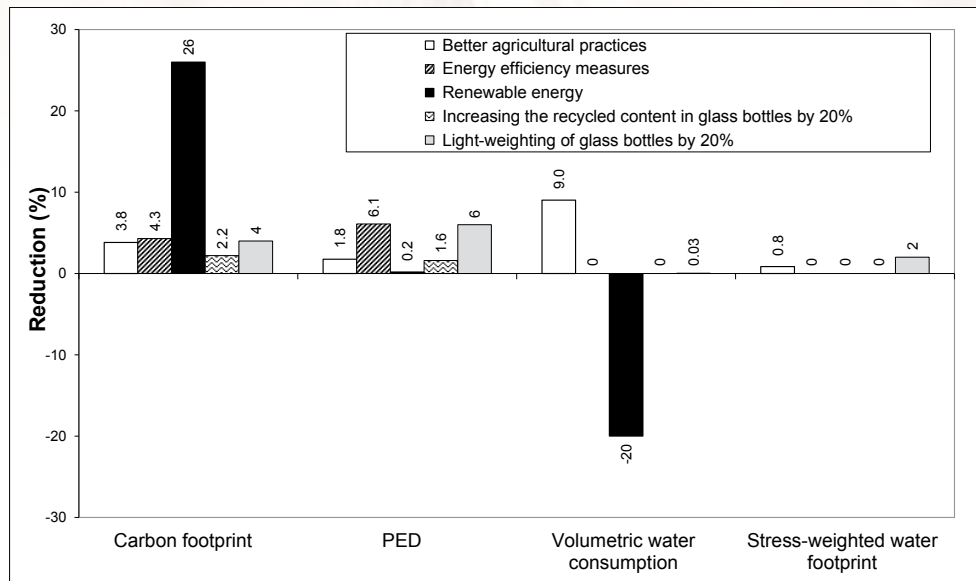


Figure 4: The effect of different improvement options on environmental impacts

There is also a growing interest in renewable energy in the Scotch whisky industry to reduce dependence on fossil fuels and GHG emissions (SWA, 2015c). To explore the potential effect on the impacts of using the renewables, it is assumed that natural gas used in the distillery is replaced by biomass (wood chips). The results shown in Figure 4 indicate that this change would lead to a 26% reduction in the CF. However, the VWC would increase by 20% although there would be no change in the stress-weighted WF. The latter is because the freshwater is not required for the production of biomass (wood chips).

Packaging improvements: As mentioned earlier, glass bottles contribute 23% to the CF and 32% to PED of whisky. Thus, the effect of two parameters on the environmental impacts – recycled glass content and bottle light-weighting – are examined. It can be seen in Figure 4 that with a 20% increase in the amount of recycled glass in the bottles, the CF and PED are reduced by around 2%, while a 20% reduction in the weight of bottles reduces the CF by 4% and PED by 6%.

5. Conclusions

This study has investigated life cycle primary energy demand, carbon and water footprints of global consumption of Scotch whisky. It has been estimated that the annual global consumption of 513.4 million litres of Scotch whisky (40% v/v) requires 13,100 TJ of primary energy and generates 1.05 Mt CO₂ eq. Its volumetric water consumption and stress-weighted water footprint are estimated at 218 million m³ and 1463 million litres eq., respectively. The production stage is the main hotspot for both the PED and CF, contributing 37% and 33%, respectively. Production of grains is the next most significant contributor to the CF (32%) and packaging to PED (32%). The grains also contribute the majority of water consumption (90%), while whisky production is responsible for 50% of the stress-weighted WF. It is recommended that measures, such as improving agricultural practices and energy efficiency in distilleries, switching to cleaner energy sources, modifying packaging and improving water management in direct operations and the supply chain, be adopted to help reduce the impacts from the Scotch whisky sector.

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154. Comparison of different data sources for the beverage industry

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ABSTRACT

Objective: Data on the inputs and outputs of the beverage industry were analysed and compared across three different data sources (detailed process data, industry association data, and input-output data) in order to identify their relative strengths and weaknesses. **Methodology:** The beverage industry is divided in five types of industries: wineries, breweries, distilleries, soft drinks producers, and mineral water bottlers. For each of these, we compared three different data sources on inputs and outputs:

- Average benchmark data from the international beverage industry association
- Data from a world-wide literature review of LCAs on beverages
- Average input-output data for the beverage industry for 14 countries on 5 continents, subdivided into five types of industries by the use of statistical data on national production volumes.

For international beverage industry association data, only energy and water use and data were available, so these were used for a cross-comparison. For other inputs and outputs, some comparisons were made between input-output data and process-data and qualitative observations were made with respect to the completeness and variability of the data sources. **Main results:** In general, the industry association averages were lower than the LCI process-data, and the input-output data gave the highest values, and the best coverage of related emissions (except for distillery for which industry association data were higher than the input-output average). The higher values in IO-data can be explained by incompleteness in data collection for process data and that both process data and industry benchmark data often have a selection bias, including relatively more enterprises with better management. On the other hand, process data provided more detail on by-products, specific fertilizer use, and many other highly specific inputs, while the industry benchmark data provided uncertainty ranges. The three data sources were combined to take advantage of the strength of each. **Discussion and conclusions:** The completeness of input-output data made these the preferred data source, but the detail of the industry benchmark and process-based data made these useful when subdividing the input-output data into data for each of the five industry types, to provide substance-specific data for chemicals, to correct aggregation errors in the input-output data, for example for by-products, and to indicate uncertainty ranges. The strengths of each data source can be used in a skillful combination of the three.

Keywords: Input-Output data, Industry data, Process-based data, Beverage industry

1. Introduction

Data on the inputs and outputs of the beverage industry were analysed and compared across three different data sources (detailed process data, industry association data, and input-output data) in order to identify their relative strengths and weaknesses.

2. Methods

For five types of beverage industries (wineries, breweries, distilleries, soft drinks producers, and mineral water bottlers), we compared three different data sources on inputs and outputs:

- IO-data: Average input-output data for the beverage industry for 14 countries on 5 continents, subdivided into five types of industries by the use of statistical data on national production volumes (see Table 1). The Input-Output data used is from EXIOBASE version 3.
- Industry averages: Average benchmark data from the international beverage industry association: For water and overall energy use, representative values are provided by the Beverage Industry Environmental Roundtable (BIER 2015), unfortunately neither specifying electricity separately, not how electricity enters into the overall energy value.
- Process-based data: Data from a world-wide literature review of LCAs on beverages: A literature review of existing process-based life cycle assessments on turned up 12 studies on wine, with 17 numerical datasets, from Australia, Canada, Italy, New Zealand, Portugal, and Spain (Amienyo et al. 2014, Aranda et al. 2005, Ardente et al. 2006, Barry 2011, Benedetto 2013, Bosco et al. 2009, Carta 2009, Gazulla et al. 2010, Neto et al. 2013, Point et al. 2012, Vázquez-Rowe et al. 2012 & 2013), 6 studies on beer production (Amienyo 2012, Climate Conservancy 2008, Cordella et al. 2008, Hospido et al. 2005, Koroneos et al. 2005, Novozymes 2009) giving rise to 5 numerical datasets (one study only covered the production

of wort, but could be used to complement some of the other studies that did not include this production step), 1 study on whisky (Amienyo 2012) and 2 studies on bottled water and soft drinks (Amienyo 2012, Quantis 2010; from U.K. and U.S.A.). In general, these data are not very complete and show a large variation. The datasets represent quite small samples, often only one producer, and can therefore not be said to be representative of their respective national productions.

Table 1: Country production volumes per beverage type

Country/region	Wine	Beer	Cider	Spirits	Bottled water	Soft drinks
Source	FAO stat	Eurostat	Eurostat	Eurostat	Eurostat	Eurostat
Unit	L	L	L	L	L	L
Check Republic	4.70E+07	1.68E+09	8.09E+06	1.82E+07	6.86E+08	1.75E+09
Germany	9.13E+08	8.71E+09	8.55E+07	1.11E+08	1.04E+10	9.44E+09
Spain	3.34E+09	2.96E+09	6.24E+07	1.74E+08	6.52E+09	6.30E+09
France	5.11E+09	1.58E+09	1.18E+08	1.91E+08	1.08E+10	5.51E+09
Ireland	0.00E+00	8.52E+08	6.84E+07	2.55E+07	6.93E+07	2.96E+08
Italy	4.67E+09	1.30E+09	0.00E+00	1.61E+08	1.56E+10	3.87E+09
Poland	8.32E+06	3.79E+09	1.57E+08	1.26E+08	3.44E+09	4.25E+09
Sweden	0.00E+00	4.59E+08	1.26E+07	3.45E+07	1.38E+08	5.84E+08
Great Britain	1.45E+06	5.46E+09	8.97E+08	7.47E+08	1.19E+09	6.46E+09
United States	2.78E+09	2.25E+10	3.48E+07	7.06E+08	3.40E+10	4.96E+10
Norway	0.00E+00	2.29E+08	1.02E+07	1.25E+06	8.60E+07	5.13E+08
Finland	0.00E+00	4.37E+08	8.21E+07	2.21E+07	7.30E+07	4.09E+08

3. Results

Table 2 shows the comparison results for energy use data, at the level of the aggregated beverage industry.

Table 2: Comparison of energy use values for IO-data, industry averages and process-based data

Input	Unit	Source	Average Beverage industry	Range
Energy	MJ _{elec+fuel} /L	Industry	1.16	0.61 – 4.41
		Process-based	1.74	0.89 – 3.09
		IO-data	4.43	0.46 – 10

The industry association averages are lower than the LCI process-data, and the input-output data give the highest values for energy use. The higher values in IO-data can be explained by incompleteness in data collection for process data and that both process data and industry benchmark data often have a selection bias, including relatively more enterprises with better management.

Figure 1 illustrates the geographical variations between IO energy data and the BIER (2015) industry energy data, when the latter is aggregated using the production volume data from Table 1. The process-based data were not sufficiently representative to allow a comparison at the national scale.

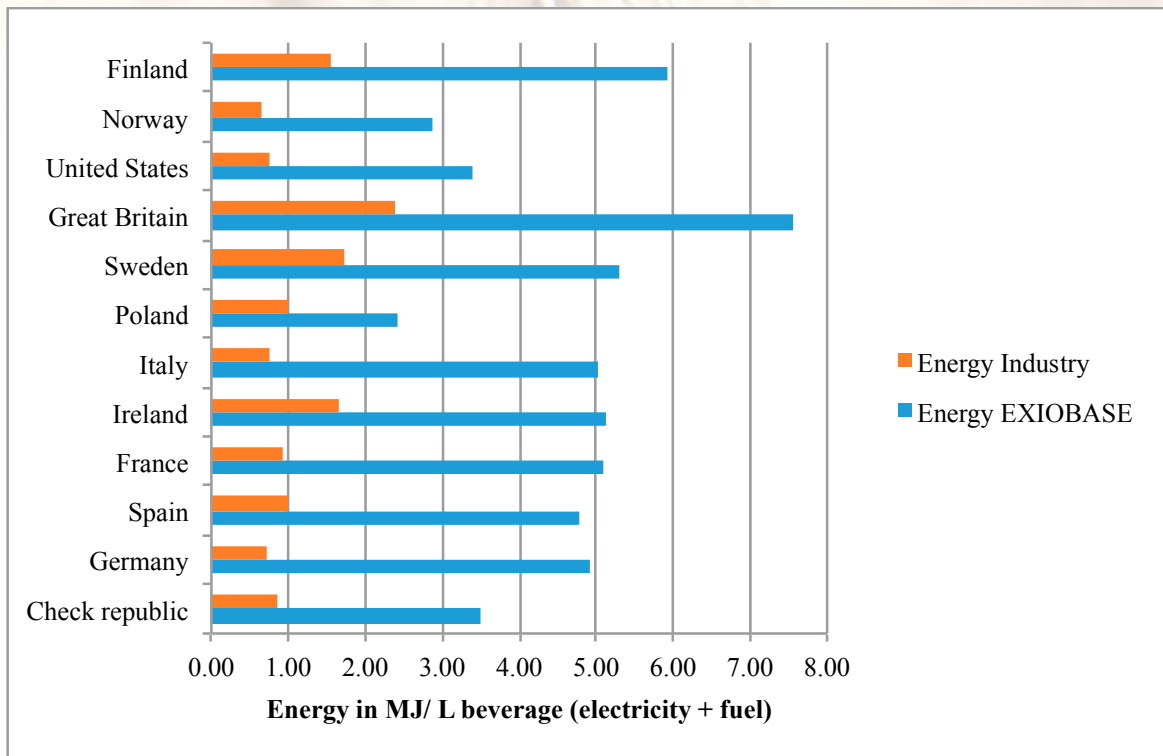


Figure 1: Comparison at country-level of energy data from beverage industry association sources and from IO-data

The values registered in the IO-database for energy consumption are higher than the aggregated energy values from industry averages. The IO energy values vary proportionally to the industry values, reflecting a pattern on energy consumption that follows the beverage industry production profile, for example the countries producing a large share of bottled water (see Table 1) show a lower energy consumption per Litre beverage reflecting the lower energy required to produce bottled water relative to the other types of beverages.

For agricultural inputs, some comparisons were made between input-output data and process-data. The process-based data on wineries, breweries and distilleries allowed for re-distribution of the agricultural inputs to the respective beverage industry types.

The Table 3 illustrates how process-based data from wineries can be used to provide substance specific data for chemicals.

Table 3: Comparison of level-of detail between IO-data and industry-based data on chemicals

	IO-data	Process-based data
Data source	Industry beverage EXIOBASE	Winery (PEF 2015)
Geographical location	Check Republic, Germany, Spain, Finland, France, Ireland, Italy, Poland, Sweden, Great Britain, United States, Canada, Australia, Norway, South Africa, Argentina/Chile, Rest of the World	France
Chemical inputs	Chemicals unspecified	Enzymes, Lactic acid, Malic acid, Tartaric acid, Potassium caseinate, Casein, Edible gelatine, Plant proteins, Ovalbumin, Classic filtration aids, Bentonite, Calcium tartrate, Potassium bitartrate, Yeast mannoproteins, Arabic Gum, Carboxymethylcellulose, Yeast cell walls, Yeast for wine production Diammonium phosphate, Ammonium sulphate, SO ₂ , potassium bisulphite, Nitrogen, Lysozyme, Ascorbic acid, Concentrated grape must, Rectified concentrated must, Sucrose, Lactic Bacteria, Potassium carbonate, PVPP, Oenological Charcoal, Oak chips, Metatartaric acid, Tannins
Amounts	0.01 – 10.88 g/L beverage	14.5 g/L wine

4. Discussion and conclusions

The completeness of input-output data made these the preferred data source, but the detail of the industry benchmark and process-based data made these useful when subdividing the input-output data into data for each of the five industry types:

- to correct aggregation errors in the input-output data, e.g. for by-products,
- to provide substance-specific data, e.g. for chemicals,
- to better specify uncertainty ranges.

The strengths of each data source can be used in a skillful combination of the three.

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31. Environmental Evaluation of Canadian Egg Production Systems

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ABSTRACT

In the last two decades, Canadian egg production systems shifted from deep-pit housing system to manure belt (or conventional cages) housing system while, in the next decade, they will most likely shifted to furnished cages, non-cage (e.g. slats/litter or aviaries) or free-range systems. In the last few years, progress has been made by Canadian producers to end the use of conventional cages.

The aim of this study was to compare the environmental impact of three cage layer housing systems: conventional cage (CC), furnished cage (FC), and cage-free aviary system (AV). This research was part of a multi-phase project, including an analysis of the hen welfare by, among others, an assessment of the quality of eggs, blood tests, and an assessment of the behaviour of hens in cages. At the end, a life cycle assessment approach, combined with an economic analysis, is used to compare the environmental footprint of the different housing systems.

In order to produce a more representative study of local production, an experiment was carried out to measure and compare gas emissions, manure characteristics, and egg production from the three cage housing systems. The experiment was replicated with 360 hens (Lohmann LSL-Lite) reared in twelve independent bench-scale rooms (mini-barn) during an 11-week period (23-34 weeks of age).

Since the purpose of the study was to assist producers in their efforts to reduce their environmental impacts, a gate-to-gate approach has been retained to assess the impacts of the production. The objective of this paper was to highlight the influence of the housings system on farm emissions. A simplified management cycle of egg production is presented in the paper. Processes whose emissions were accounted for included emissions from sources owned and control by the farm that are affected by the manure management strategy.

For comparison purposes, a typical farm, housing 30,000 hens aged from 19 to 70 weeks old, was used. The main characteristics of the farm represented the average egg farm found in the Province of Québec, Canada. Manure produced in the three scenarios was stored on the farm and land-applied on the soils near the farm following a nutrient management plan based on a phosphorus index.

For the entire farm, NH₃ emissions from the AV barn represented a loss of roughly 6,000 kg of N that was not available to fertilize the crops. This loss of nitrogen influences the amount of fertilizer needed for crops. Since a smaller quantity of phosphorus is produced in the AV system, a smaller area of crop is needed to spread all the manure produce on the farm. Finally, the amount of fertilizer needed for crops was also smaller limiting the environmental impact of the AV system.

Keywords: Laying hens, housing systems, ammonia emissions, greenhouse gas emissions.

1. Introduction

In the last two decades, Canadian egg production systems shifted from deep-pit housing system to manure belt (or conventional cages) housing system while, in the next decade, they will most likely shifted to furnished cages, non-cage (e.g. slats/litter or aviaries) or free-range systems. In the last few years, progress has been made by Canadian egg producers to end the use of conventional cages. From an environmental point of view and a hen welfare perspective, it appeared that no single housing system was ideal. A better understanding of those systems was a key factor in the identification of the environmental hot spots associated with laying hen production systems.

The aim of this study was to compare the environmental impact of three cage layer housing systems: conventional cage (CC), furnished cage (FC), and cage-free aviary system (AV). This research was part of a multi-phase project, including an analysis of the hen welfare by, among others, an assessment of the quality of eggs, blood tests, and an assessment of the behaviour of hens in cages. At the end, a life cycle assessment approach, combined with an economic analysis, will be used to compare the environmental footprint of the different housing systems.

Since the purpose of the study was to assist producers in their efforts to reduce their environmental impacts, a gate-to-gate approach has been retained in this paper to assess the impacts of the production. The objective of this paper was to highlight the influence of the housings system on farm emissions.

A previous analysis showed that the whole farm nitrogen cycle was identified as a major contributor to the environmental impact. Results showed that, in general, the major N loss was in the $\text{NH}_3\text{-N}$ form at every stage of the production and management cycles.

A simplified management cycle of egg production is presented in the paper. Processes whose emissions were accounted for included emissions from sources owned and control by the farm and affected by the manure management strategy. Emissions produced outside the farm, like feed and pullets' production and transformation of eggs, hens and crops were not included.

2. Methods

For comparison purposes, a typical farm, housing 30,000 hens aged from 19 to 70 weeks old, was used. The main characteristics of the farm represented the average egg farm found in the Province of Québec (CRAAQ, 2007). Manure produced in the three scenarios was stored on the farm and land-applied on the soils near the farm following a nutrient management plan based on a phosphorus index.

Typical diet for laying hens contained grain maize (55.3%), soybeans (10%), soybean meal (9.6%), limestone (9.7%), dried distillers grains (7.5%), and the other dietary supplement (7.9%). For the purpose of this study, grain maize and soybeans were all produced on the farm, and the remaining ingredients were purchased. In the three scenarios, the area of crops was fixed for grain maize and soybean to produce the amount needed to feed 30,000 hens year⁻¹.

Barn

Experimental Setup

In order to produce a more representative study of local production, an experiment was carried on to measure and compare gas emissions, manure characteristics, and egg production from the three cage housing systems. The experiment was replicated with 360 hens (Lohmann LSL-Lite) reared in twelve independent bench-scale rooms (mini-barn) during an 11-week period (23-34 weeks of age). The experiment was a completely randomized design with three housing systems and four repetitions. Values obtained from the experiment were subsequently used in the different scenarios. The hens were fed 100 g/day-hen of a commercial diet. Water was provided by a solenoid activated valve connected to a data logger to register the water flow through the nipple drinkers inside the cages. The lighting system was regulated to give 40 lux per room for 13.5 hours/day.

The temperature and relative humidity of the air in each individual room were measured using a probe. A data logger was connected to a computer to upload data coming from the temperature-relative humidity probe every 10 s and the average value was recorded every 15 min. The temperature in the rooms was set at 22.5°C. Ventilation rates were calculated from a 204 mm iris orifice damper installed in the exhaust duct of each room. The difference of pressure was measured across the damper every 10 s and a data logger recorded the average every 15 min.

Conventional Cages

Hens were reared in conventional cages (486 mm wide x 507 mm deep x 540 mm high) put 2 x 2 on three decks for a total of six cages. Each cage included five hens (492 cm²/hen). Manure dropped on a belt beneath each row of cages where it was dried with forced air and removed twice a week. The drying system was installed under all the decks of the battery cages. A perforated 7.5 cm diameter duct blew air (19.8 L/min-hen) from a 10 cm blower.

Furnished Cages

Hens were reared in furnished cages specifically built for the project. The system consisted of three cages containing 10 hens (780 cm²/hen) installed one on top of the other. Each cage was 1.3 m long x 0.60 m wide x 0.45 m height, and was equipped with 2 perches, 1 scratch pad, and 1 plastic-curtained nest box. Following the same specifications given for the battery cages, a belt with a drying system was installed underneath each tier.

Aviary System

The three-tier aviary system had dimensions of 1.42 m long x 1.2 m wide x 1.8 m high. The first floor was covered with bedding. The second and third tiers were equipped with a plastic-curtained nest box and hens had access to a total of six perches. A manure belt with a drying system was installed underneath the second and third tier.

Manure Sampling

Manure samples were collected every week by taking a fixed amount of manure from a random spot on each belt. Samples were analyzed for dry matter content (DMC), pH, total nitrogen (TN), ammonium nitrogen (NH₄-N) and minerals (P, K, Ca and Mg).

Gas Emissions

The sampling air was pumped to a mobile laboratory through Teflon™ tubing. In this laboratory, CO₂ and CH₄ were analyzed by gas chromatography and NH₃ was analyzed by a non-dispersive infrared analyzer. A data logger then recorded the values measured every 15 min. Concentration measurements were taken continuously during the entire experiment and were synchronized with the ventilation flow rate. Emissions were then calculated for each sampling period by multiplying the difference in concentration by the mass flow of the gas.

Storage

Manure storage emissions were derived from results obtained in a similar experiment carried on four years ago by the research team (Fournel et al., 2012a and b). The objective of that particular experiment was to measure and compare gas emissions from laying hens housed in conventional cages. Both experiments were made using the same methodology.

Soils and Crops

Manure produced by the hens was considered land-applied onto the fields near the farm with conventional spreading equipment following a nutrient management plan based on a phosphorus index. The crop area was set, based on the amount of phosphorus produced by the farm, the phosphorus concentration in the soil and crop needs. Phosphorus and potassium levels in the soils were set respectively at 90 kg P/ha and 150 kg K/ha.

Culture yields and nutrient application rates are presented in table 1. Nutrient application rates are based on the recommendations presented in CRAAQ (2003). The amount of phosphorus (11,250 kg P₂O₅) and potassium (10,950 kg K₂O) needed, came from the manure while additional nitrogen fertilizer was applied according to crop needs. For the CC, FC, and AV scenarios, the amount of additional nitrogen fertilizers applied were respectively 9,173 kg N, 9,647 kg N and 10,104 kg N.

After harvesting, the moisture content of grain-maize was set at 30% and the moisture content of the other crops at 20%. All the crops are dried using propane at 14% moisture content and stored on the farm.

Nitrogen losses included NH_3 , N_2O and NO_x emissions into the air and NO_3 leaching into groundwater. N_2O emissions from agricultural soils were calculated with the methods used in the Canadian GHG inventory (Environment Canada, 2010) and in Rochette et al. (2008). Nitrous oxide emissions from agricultural soils consisted of direct and indirect emissions. Direct sources were emissions from nitrogen that has entered the soil from animal manure applied as fertilizer and crop residue decomposition. Indirect sources are emitted off site through volatilization, redeposition, leaching and runoff of manure and crop residues. Other emissions factors depend on the amount of nitrogen applied. NH_3 and NO_x emission factors were set at 0.12 kg N- NH_3 /kg N for mineral fertilizers, 0.16 kg N- NH_3 /kg N for manure and 0.011 kg N- NO_x /kg N (Hamelin, Jørgensen, Petersen, Olesen, & Wenzel, 2012). Losses of NO_3 in groundwater were calculated using the SQCB- NO_3 model (Nemecek & Schnetzer, 2011).

Phosphorus losses included P and PO_4^{3-} losses by runoff to surface water and PO_4^{3-} leaching into groundwater. Phosphorus losses were estimated using a tool for monitoring environmental risk from diffuse exports of agricultural phosphorus (Michaud et al., 2009).

Table 1: Culture yields and nutrient application rates

Crops	Yield t ha ⁻¹	Nutrient application rates (kg ha ⁻¹)		
		N	P ₂ O ₅	K ₂ O
Grain-maize	8.7	170	80	75
Soybean	2.7	30	50	40
Barley	3.1	80	40	50
Wheat	3.5	100	40	50
Oat	2.9	50	35	40

Energy

Energy consumption was derived from economic analysis produced to evaluate the production cost of agricultural activities in the Province of Québec (CRAAQ, 2007): it included electricity consumption inside the building (lighting, manure management, egg collection) and diesel consumption for cultural operations and propane, and electricity consumption for drying and storage of crops. The values came from a mix of the farm-specific financial accounting and generic production budgets.

3. Life Cycle Inventory

Barn Emissions

Table 2 presents the laying hen performances for the three systems. Production in conventional and furnished cages was similar throughout the project, averaging 0.96 eggs day⁻¹ hen⁻¹. Egg production in the aviary system averaged 0.77 eggs day⁻¹ hen⁻¹. Composition (as fed-basis) of the diet is shown in Table 3, while the main characteristics of the eggs produced are presented in Table 4.

Gas emissions from the three cage layer housing systems are presented in Table 5. Emissions measured in the aviary system were almost 16th times higher than those of the conventional and furnished cages. Litter management, flock density on the first floor, airflow, T, HR, as well as manure decomposition, have contributed to ammonia production. CH_4 and CO_2 emissions were similar among the three systems, ranging respectively from 21 to 23 g CH_4 day⁻¹ AU⁻¹ and from 21.9 to 25.7 kg CO_2 day⁻¹ AU⁻¹.

The results presented in tables 2, 3, 4, and 5 show two primary factors affecting the LCI results: egg production and NH_3 emissions. Greenhouse gas emissions from the barn housing 30,000 hens

were almost identical for the three scenarios: 41 t CO₂e year⁻¹ for the CC housing system and 39 t CO₂e year⁻¹ for the FC and AS housing systems. Ammonia emissions varied from 0.63 t NH₃ year⁻¹ for the CC and 0.44 t NH₃ year⁻¹ for the FC to 7.9 t NH₃ year⁻¹ for the AV housing system. For the entire farm, NH₃ emissions from the AV barn represented a loss of roughly 6,000 kg of N that was not available to fertilize the crops.

Table 2: Laying hen performances

Parameters	Conventional cage	Furnished cage	Aviary system
Initial weight (kg)	1.516	1.524	1.519
Final weight (kg)	1.692	1.605	1.587
Egg production (egg day ⁻¹)	0.96	0.96	0.77
Egg weight (g)	59.6	59.3	58.6
Feed consumption (g day ⁻¹)	123.4	119.9	117.8

Table 3: Diet composition

Dry mater (%)	N (%)	NDF (%)	ADF (%)	P (mg/kg)	K (mg/kg)	Ca (mg/kg)	Mg (mg/kg)	Al (mg/kg)
91.8	3.2	22.2	8.0	6,058	7,649	54,428	1,668	347

Table 4: Egg quality

Parameters	Conventional cage	Furnished cage	Aviary system
Egg inside composition			
Dry matter (%)	23.7	23.5	23.0
N (g kg ⁻¹)	19.4	19.0	19.1
P (mg kg ⁻¹)	1,942	1,941	1,897
Eggshell composition			
Dry matter (%)	72.5	74.2	77.5
P (mg kg ⁻¹)	913.3	917.4	944.8
Ca (g kg ⁻¹)	342.9	342.5	344.7

Table 5: Gas emissions from the three cage layer housing systems

Parameters	Conventional cage	Furnished cage	Aviary system
Emissions (...hen ⁻¹ day ⁻¹)			
NH ₃ (mg)	60.0	42.5	759.2
CH ₄ (mg)	75.0	68.6	67.7
CO ₂ (g)	82.7	68.6	80.1
Emissions (...AU ⁻¹ day ⁻¹)			
NH ₃ (g)	18.7	13.6	242.4
CH ₄ (g)	23.3	21.8	21.6
CO ₂ (kg)	25.7	21.9	25.6
Emissions (...kg egg ⁻¹)			
NH ₃ (g)	1.08	0.77	18.68
CH ₄ (g)	1.30	1.23	1.45
CO ₂ (kg)	1.48	1.25	2.03

Storage Emissions

Characteristics of the manure produced by the hens are presented in Table 6 and gas emissions of the stored manure from the three cage layer housing systems are presented in Table 7. The quantity of manure produce influences the total gas emissions.

Table 6: Characteristics of the manure from the three cage layer housing systems

	Conventional cage	Furnished cage	Aviary system
Amount (g hen ⁻¹ day ⁻¹) ^a	70.8 ± 5.1	55.2 ± 3.1	49.2 ± 1.9
pH	6.27 ± 0.06	6.33 ± 0.03	8.38 ± 0.11
Dry matter (%)	50.5 ± 3.6	55.4 ± 3.8	73.3 ± 0.2
Organic matter (%)	36.5 ± 2.4	40.0 ± 2.9	55.9 ± 0.2
Total nitrogen (g N kg ⁻¹)	29.5 ± 0.5	35.0 ± 1.7	27.9 ± 0.6
Ammoniacal N (mg NH ₄ ⁺ -N kg ⁻¹)	3.25 ± 0.37	3.14 ± 0.12	3.56 ± 0.91
Phosphorus (g P kg ⁻¹)	8.0 ± 0.1	9.9 ± 0.7	10.0 ± 0.0
Potassium (g K kg ⁻¹)	11.5 ± 0.6	13.8 ± 0.8	14.2 ± 0.7
Calcium (g Ca kg ⁻¹)	31.9 ± 2.4	40.5 ± 1.6	33.7 ± 0.9

Table 7: Gas emissions of the stored manure for a 30,000 hens farm

Parameters	Conventional cage	Furnished cage	Aviary system
CO ₂ (t CO ₂ e year ⁻¹)	0.47	0.37	0.33
CH ₄ (kg CH ₄ year ⁻¹)	1.72	1.11	1.34
N ₂ O (kg N ₂ O year ⁻¹)	1.43	1.11	0.99
NH ₃ (t NH ₃ year ⁻¹)	1.15	0.89	0.80

Soils and Feed Production Emissions

Areas of crops in the three scenarios are presented in Table 8 while Table 9 presents the amount of manure and N fertilizer applied in the three scenarios. In the CC scenario, the amount of phosphorus (13,547 kg P₂O₅) and potassium (10,204 kg K₂O) needed, came entirely from the manure while additional nitrogen fertilizer (9,173 kg N) was applied according to crop needs. For the FC and AS scenarios, the amount of additional nitrogen fertilizer applied were respectively 9,647 kg N and 10,104 kg N.

Greenhouse gas emissions from crop production (including energy used for cultural operations and for drying and storage of crops) was 330 t CO₂e year⁻¹ for the CC housing system, 317 t CO₂e year⁻¹ for the FC, and 280 t CO₂e year⁻¹ for the AS housing systems. Ammonia emissions from crop production varied from 11.5 t NH₃ year⁻¹ for the CC, 11.1 t NH₃ year⁻¹ for the FC, and 9.9 t NH₃ year⁻¹ for the AS housing system. Greenhouse gas emissions and ammonia emissions from crop production were affected by the total area of land use.

Table 8: Crop areas for the three scenarios

Crop	Conventional cage	Furnished cage	Aviary system
Grain maize (ha)	88	84	80
Soybean (ha)	41	41	41
Barley (ha)	18	15	9
Wheat (ha)	14	15	8
Oat (ha)	13	14	8
Total (ha)	174	169	146

Table 9: Manure and N fertilizer applied for the three scenarios

	Conventional cage	Furnished cage	Aviary system
Manure applied (t)	739.4	576.5	513.8
N (kg)	19,829	18,437	13,032
P ₂ O ₅ (kg)	13,547	13,070	11,767
K ₂ O (kg)	10,204	9,547	8,756
N fertilizer (kg)	9,173	9,647	10,104

4. Discussion and Conclusions

The objective of this paper was to highlight the influence of the housings system on farm emissions from manure management. Values obtained to establish the environmental impact of three cage layer housing systems (conventional cage, furnished cage, and cage-free aviary system) were presented.

The two primary factors affecting the inventory results were egg production and NH₃ emissions. Ammonia emissions measured in the aviary system were almost 16th times higher than those of the conventional and furnished cages. For the entire farm, NH₃ emissions from the AV barn represented a loss of roughly 6,000 kg of N that was not available to fertilize the crops. This loss of nitrogen influences the amount of fertilizer needed for crops. Since a smaller quantity of phosphorus is produced in the AV system, a smaller area of crop is needed to spread all the manure produce on the farm. Finally, the amount of fertilizer needed for crops was also smaller limiting the environmental impact of the AV system.

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ORAL PAPERS

LAND USE AND EUTROPHICATION

13. Land Use and Eutrophication

18. Assessing Land Use Change Impact of Food Products Worldwide with the World Food LCA Database

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ABSTRACT

The World Food LCA Database (WFLDB) developed a unique approach to consistently include land use change (LUC) in life cycle inventory data for all crops and countries globally. In this approach, two allocation schemes are provided: the “crop-specific” and the “shared responsibility”, each corresponding to different “value systems”. The carbon footprint study of several crops (coffee, cocoa) and food products (milk, vegetal oils) from various countries demonstrates the uttermost importance of systematically including land use change in the life cycle assessment of food products, with differences up to a factor 12 in the most extreme cases. Standardisation through a global framework on land use change is needed to ensure a sound and consistent climate change policy.

Keywords: agriculture, deforestation, LUC, climate change

1. Introduction

In 2012, the sustainability consultancy Quantis and the Swiss Federal research institute for agriculture Agroscope joined their forces to create an exclusive international consortium with nine major, global, private companies from the agro-food value-chain (Bayer CropScience, General Mills, Kraft Foods, Mars, Mondelez International, Monsanto, Nestlé, PepsiCo, Syngenta and Yara) and two governmental agencies (ADEME and the Swiss Federal Office for the Environment) into what would become the World Food LCA Database (WFLDB) project (www.quantis-intl.com/wflldb). Three years later, 400 datasets for crops, animal products and food products in 40 countries were delivered to the project partners and in many parts published through the ecoinvent database. The objective of the WFLDB was to answer the need for transparent, consistent and disaggregated inventory data for agricultural and food products, using the best available science.

A major outcome was the development of the *Methodological Guidelines for the Life Cycle Inventory of Agricultural Products* (Nemecek et al. 2015), a detailed and comprehensive guidance for inventory modelling of agricultural and food systems. To date, these are among the very few peer reviewed documents providing operational guidance applying to a wide variety of agricultural systems and countries. Among all methodological issues relevant to agro-food systems which are addressed in the *Guidelines*, the innovative approach for modelling land use change (LUC) is a true breakthrough in the world of life cycle inventory (LCI) data. Applied consistently through all crops and countries, it provides a whole new level of understanding of land use change related impacts of food products and commodities, using two distinct “value systems”.

2. Methods

The *Methodological Guidelines for the Life Cycle Inventory of Agricultural Products* are compliant with the most recognised standards such as ISO 14040, 14044 and 14046 (ISO 2006a; 2006b; 2014), ILCD entry-level requirements (EC-JRC 2012), ecoinvent data quality guidelines (Weidema et al. 2013), International Dairy Federation Guide (IDF 2015) and IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006). They are also to a large extent aligned with the LEAP Partnership guidance (LEAP 2015) and draft Product Environmental Footprint Category Rules

(PEFCR¹, European Commission 2016) for several sectors. However, none of these standards yet provided an operational approach to include land use change in LCI data for a wide choice of products and at global level.

Building on the *Direct Land Use Change Assessment Tool Version 2013.1* (Blonk Consultants, 2013) and compliant with PAS 2050-1 protocol (BSI 2012), the LUC modelling approach followed in the WFLDB accounts for all carbon pools, i.e. above-ground biomass (AGB), below-ground biomass (BGB), dead organic matter (DOM) and soil organic carbon (SOC) (Table 1). The values for the relevant carbon pools are taken from the IPCC Agriculture, Forestry and Other Land Use (AFOLU) report (IPCC 2006) and FAO (2010), Annex 3, Table 11. Country climates and soil types are taken from the European Soil Data Centre (ESDAC 2010).

Three major modifications have been brought to the original tool to comply with the WFLDB methodological requirements: a) addition of the SOC-related emissions from peat drainage based on Joosten (2010) and IPCC (2013); b) inclusion of carbon capture in vegetation when occurring (e.g. when grassland is transformed into perennial cropland); c) addition of N₂O emissions related to SOC degradation according to IPCC (2006).

In crop production, global land transformation impacts are mainly driven by deforestation of primary forests. However, land use change from secondary forest or other types of land use (grassland, perennial or annual crops) to arable land are also addressed (Table 1).

Table 1: Carbon pools accounting in land transformation

Carbon pool	Land transformation to annual or perennial crop				
	From primary forest	From secondary forest	From perennial crop	From annual crop	From grassland
AGB ⁽¹⁾	8% harvested and stored 92% emitted (20% burned, 72% by decay)			100% emitted by decay Net carbon capture may occur in certain cases (and is taken into account)	
BGB ⁽²⁾	100% emitted by decay				
DOM ⁽³⁾	100% emitted by decay			Ignored	
SOC ⁽⁴⁾	SOC change according to IPCC 2006, including peat drainage emissions. Net carbon capture may occur in certain cases (and is taken into account)				

⁽¹⁾ Aboveground biomass; ⁽²⁾ Belowground biomass; ⁽³⁾ Dead organic matter; ⁽⁴⁾ Soil organic carbon

WFLDB does not formally distinguish direct land use change (dLUC) and indirect land use change (iLUC). The same approach as the one applied in the original LUC tool has been adopted: land use is inventoried at national level per crop and per type of land use based on FAO annual data (FAOSTAT 2012, FAO 2010). Changes are calculated over the period 1990 - 2010.

The LUC impact assessment follows the framework defined in ecoinvent v3 (Nemecek et al. 2014), which is based on IPCC (2006) methodology. The fate of the carbon in each pool is described in Table 1. A time period of 20 years is used for the amortisation of the emissions, which is aligned with PAS 2050-1 (BSI 2012) and FAO guidelines for feed supply chains (LEAP 2015).

Furthermore, and unique to WFLDB, two allocation schemes corresponding to different “value systems” are provided: the “crop-specific” and the “shared responsibility” approaches. Each system uses its own key (Table 2). The default system is “crop specific”.

¹ http://ec.europa.eu/environment/eussd/smgp/ef_pilots.htm

Table 2: Value systems and respective LUC allocation keys

Value system	Allocation key
Crop-specific approach	Land use change is allocated to all crops and activities which production area grew during the last 20 years in a given country, and only to them, according to their respective area increase.
Shared responsibility approach	Land use change during the last 20 years is evenly distributed among all crops and activities present in the country, based on current area occupied.

3. Results

The impact on global warming (100 years), or carbon footprint, of several products was calculated based on WFLDB LCI data, using the IPCC 2013 emission factors. This paper shows the outcome of the analysis for a selection of products, some in their raw form at the farm gate, and others after transformation or inclusion of more complex food products. In each case, three scenarios were assessed:

- Carbon footprint of 1 kg of product, from the cradle-to-gate, excluding land use change
- Carbon footprint of 1 kg of product, from the cradle-to-gate, including land use change with the crop-specific allocation
- Carbon footprint of 1 kg of product, from the cradle-to-gate, including land use change with the shared-responsibility allocation

The analysis for green coffee beans (Figure 1) and sun-dried cocoa beans (Figure 2) cultivated in the leading exporting countries shows the critical importance of including LUC for agricultural systems in Latin-American, African and Southeast Asian countries. The influence on the carbon footprint often reaches a factor 2 for green coffee (i.e. in Honduras, Indonesia and Vietnam) and up to a factor 13 in the most extreme case of cocoa beans production in Indonesia (Table 3), from 3 kg CO₂-eq/kg to 40 kg CO₂-eq/kg. Such differences are explained by very low yields and low level of mechanization, combined to large increase of cultivated area at the expense of primary forest.

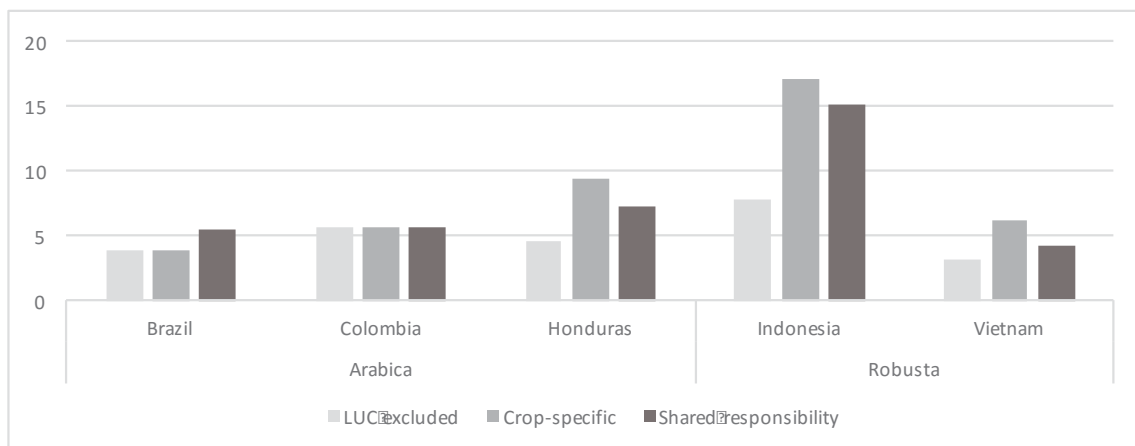


Figure 1: Impact on global warming (100 y), in kg CO₂-eq per kg green coffee at the farm

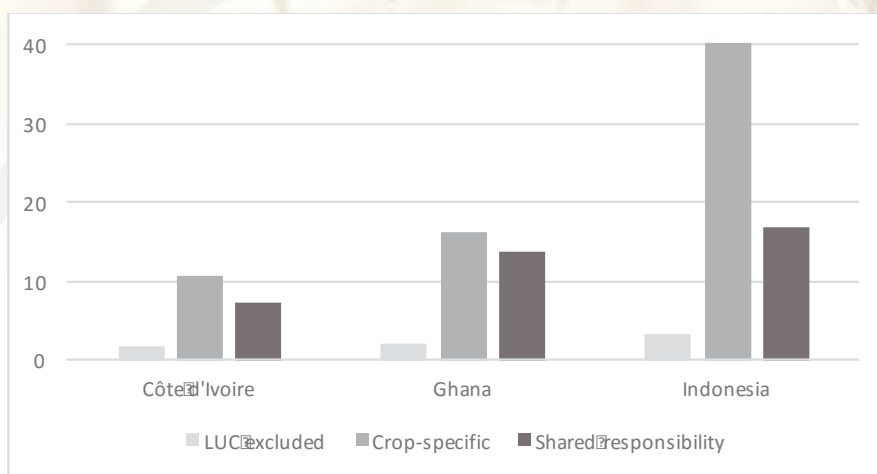


Figure 2: Impact on global warming (100 y), in kg CO₂-eq per kg sun-dried cocoa beans at the farm

Table 3: Influence of including land use change on the carbon footprint of green coffee beans and sun-dried cocoa beans, at the farm

Crop	Country	% of global production (FAOSTAT 2010)	Carbon footprint increase due to LUC inclusion	
			Crop-specific	Shared responsibility
Green coffee beans	Brazil	34.5%	0%	+44%
	Colombia	6.4%	0%	0%
	Honduras	2.7%	+102%	+58%
	Indonesia	8.1%	+122%	+96%
	Vietnam	13.1%	+99%	+34%
Cocoa beans	Côte d'Ivoire	29.2%	+506%	+305%
	Ghana	10.4%	+688%	+568%
	Indonesia	16.0%	+1198%	+438%

The study also demonstrates that inclusion of LUC can also be critical for animal products and transformed food products. The assessment of the carbon footprint of fat and protein corrected raw milk produced in the United-States illustrates the influence of LUC within animal feed (Table 4). In this example, the same dairy production system (i.e. a mix of grazing and non-grazing systems) was modelled following two scenarios. First, we considered that all soybean included in the feed ration as compound feed was supplied from US producers. Second, we considered that all soybean was supplied from Brazil and Argentina, which are typically the main suppliers to the European feed market. The inclusion of LUC in that case influences the conclusions a dairy farmer might draw when willing to reduce the impact of raw milk through changing his compound feed supply chain from US to South American soybean, going from a 14% carbon footprint reduction to a 21% increase.

Table 4: Carbon footprint of US raw milk with different supply chains for soybean in the feed ration

Soybean origin	LUC excluded	Crop-specific		Shared responsibility	
	kg CO ₂ -eq/kg	kg CO ₂ -eq/kg	Increase	kg CO ₂ -eq/kg	Increase
USA	1.23	1.23	0%	1.24	+0.3%
Argentina & Brazil	1.05	1.50	+42%	1.26	+19%
<i>Delta</i>	-14%	+21%	-	+2%	-

The comparison of different vegetal oils provides a different insight into the relevance of considering LUC in agricultural systems. In this example, coconut oil, palm oil and rapeseed (or canola) oil from various countries are compared in regard to their carbon footprint per kilogram of oil

at the mill exit gate (Figure 3). While the impact of Indonesian coconut oil (from 1.5 to 4.5 kg CO₂-eq/kg, or a factor 3) and palm oil (from 0.6 to 4.6 kg CO₂-eq/kg, or a factor 8) is very sensitive to LUC, the carbon footprint of the same products in India, or rapeseed oil in Canada and France is not. Basically, LUC allocation is sensitive in countries where large surfaces of natural land were transformed to arable land in the last 20 years, and is not where LUC is completed since long. It appears that palm oil seems to have a lower impact than other oils when LUC is not considered, due to the high yields of palm trees. However, its impact can also be the highest of those presented here when considering their specific role in LUC. It therefore appears critical to account for LUC in comparative assessments, especially when this might lead to erroneous claims.

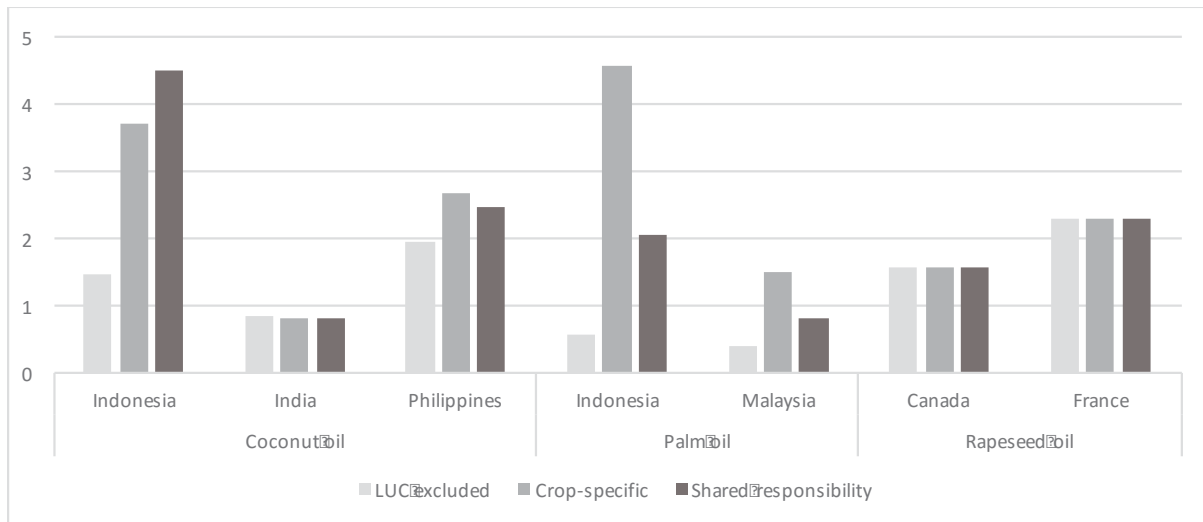


Figure 3: Impact on global warming (100 y), in kg CO₂-eq per kg vegetal oil, at the oil mill

4. Discussion

When analysing the results of the current study, one should bear in mind that the WFLDB provides LCI data for typical production systems at national scale. In other words, each LCI considers an average yield, set of fertilisers and pesticides inputs, irrigation amount and technologies and machinery, as well as average soil and climate conditions in the country of reference. The impact assessment results are therefore not representative of specific production systems (e.g. organic, very high productivity or local climatic conditions).

Nevertheless, the analysis of the WFLDB LCI data shows the high significance of land use change over the carbon footprint of many food products and the large variability among crops and countries. The analysis shows that LUC has a major influence in countries where deforestation is still occurring such as Indonesia and Brazil, while it does not in countries where agriculture is no longer expanding at the expense of natural land, such as the United-States, India or France. It also demonstrates the need to consistently assess LUC through all agricultural systems, as to enable fair comparison of different food products fulfilling a similar function. The example of soybean used as feed for dairy cattle further shows that not accounting for LUC in the complete supply chain might lead to incorrect decisions and policies when it comes to climate change.

The choice of different “value systems” for allocating the total LUC to specific crops can alone have a major influence on the total carbon footprint and may in some cases revert conclusions of a comparative assessment. It remains a subjective choice to decide which of the crop-specific and the shared responsibility approach is most relevant. This choice yet relates more to a philosophical view of land availability (or land pressure) than it relies on scientific principles or hard data. For this reason, the authors of the WFLDB decided to provide both but emphasises the need to a) always account for LUC through all transformation schemes and accounting for all carbon pools, and b) apply a consistent methodology and common value system through the entire supply chain.

5. Conclusions

The WFLDB demonstrates the uttermost importance of including land use change in the LCA of food products and addressing allocation to different crops in a consistent way. Choosing one value system over another currently remains a subjective choice, and available standards are yet neither sufficiently detailed with regards to this decision, nor operational enough. It is now critical that the LCA community works on providing a clear land use change allocation framework – or standard – supported by scientific evidence and global consensus. This is key to ensure comparability of LCI data and LCA studies in the agro-food sector.

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190. Quantifying Land- Use Change Associated with US Agricultural Production.

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Objective: Many studies in the food and agricultural sector in the United States have assumed that since the US agricultural landscape has been relatively stable for many decades, that land-use change is a relatively unimportant contributor to life cycle impacts of the food supply chain. In this work, we investigate land-use change in US agriculture using publicly available remotely sensed data. The major sources of information are the National Land Cover Database (NLCD) published by the Multi-Resolution Land Characteristics Consortium, which differentiates 16 land-use classes, of which 2 are agricultural. The 2nd source of information is the Crop Data Layer (CDL) published by the USDA, which differentiates, at 30 m resolution, all of the major crops produced in the US.

Methods: Because of the high level of aggregation of the NLCD for agricultural use, we have focused on preliminary work on using the CDL to identify land changes over 5 and 10 year periods. The CDL was initiated in the upper Midwest, and the state of North Dakota has the longest continuous data record. Therefore we have focused on initial analysis in this area. Since 2008, the CDL has expanded and provides nationwide coverage. However because the possibility of accounting for land-use change is important our approach is to evaluate detailed crop level land-use change and then correlate this with data available from the NLCD, which has a longer history of nationwide coverage, and NASS data on crop areas to extrapolate the crop specific land transformation at national scale. To simplify the analysis, as a proof of concept, we aggregated the approximately 130 specific land use classifications reported into 9 (Table 1). To calculate the area transformed from non-agricultural land to a specific crop, we assigned pixel values in the transformed map with the formula $(LU1*10 + LU2)$. Thus a pixel assigned as Forest in the first analysis year which was planted in corn in the second analysis year would be assigned a value of 53 (i.e., forest to corn transformation).

Results and Implications: Figure 1 presents initial results for North Dakota showing individual locations where change in land use from forest or grassland to corn/soy occurred. While the detailed evaluation at this resolution is not normally required for LCA, the aggregation of the information and the ability to see bi-directional transformations at regional scale (Table 2) represents an important step in inventory. Because the impacts of the reverse transformations do not fully offset the initial transformation, a detailed accounting such as enabled by this analysis will provide a better platform for the inclusion of LUC in biodiversity and ecosystem services assessments which are being discussed in the international community.

Table 1. Aggregated LU Classifications for analysis

Land Use	Classification	Land Use	Classification
No data	0	Forest	5
Alfalfa	1	Grass	6
Wheat	2	Other Ag	7
Corn	3	Soybeans	8
Developed	4	Water	9

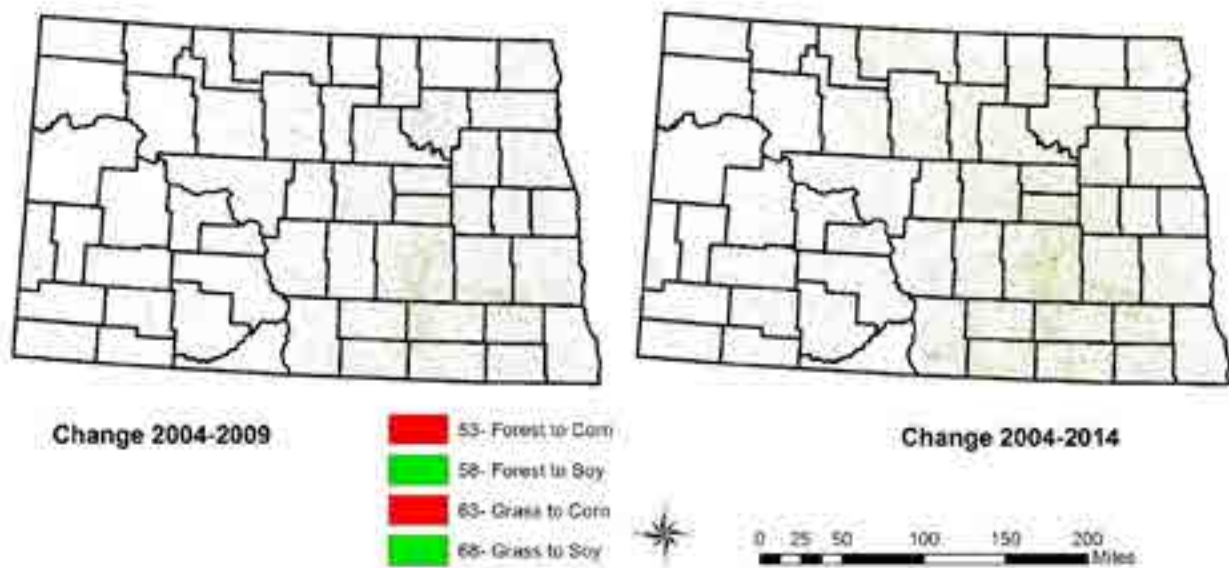


Figure 4. Map of LUC associated with conversion of forest and grassland/pasture to corn and soy between 2004-2009 and 2004-2014

Table 2. Aggregate land-use changes for North Dakota between 2004 - 2014			
Transformation	ha	Transformation	ha
Forest-corn	438	Grass-wheat	901,973
Forest-soy	1,123	Ag -grass	776,152
Forest-wheat	5,571	Ag-forest	40,091
Grass-corn	311,989	Grass - forest	311,989
Grass-soy	220,827	Forest - grass	28,361

56. Country Specific Land Use Characterization Factors for Life Cycle Impact Assessment

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ABSTRACT

In recent years, methods for the consideration of impacts of land use on the environment have successfully been developed and applied in LCA. In order to enhance their usability, the availability of characterization factors (CF) based on scientifically acknowledged methods is crucial. A set of country and land use type specific characterization factors was calculated using detailed GIS input data representing impacts on ecosystem services according to LANCA[®] (Land Use Indicator Value Calculation in Life Cycle Assessment). The calculation method of the CFs for several indicators of ecosystem services is presented in this paper. These characterization factors are published and can be applied and integrated into Life Cycle Impact Assessment to address the impact category land use in LCA studies. Furthermore, in using the CFs for LCA studies as well as in sensitivity analyses and consistency checks it turned out, that the provided CFs objectify the ongoing discussion within the scientific community regarding the choice of the reference situation as a crucial factor influencing the results. Depending on the choice of the reference situation the results can be completely different. One possible solution for this discrepancy could be to provide one set of CFs for each reference situation. This way, the decision about the reference state can be made by the practitioner, taking into consideration the goal and scope of the study.

Keywords: reference situation, land use impact assessment, LANCA[®], ecosystem services

1. Introduction

More than half of the earth's terrestrial land is actively being used by humans. For these purposes an area between 5,000 and 15,000 ha (Schmidt 2010) of natural land is sealed per day around the globe. The resulting loss of biodiversity and ecosystem services is not only of scientific but also of political, societal and economic concern. In order to assess all relevant impacts of a product or process on the environment, also land use aspects have to be considered within methods such as Life Cycle Assessment (LCA).

Lately, methods for the consideration of impacts of land use on the environment have successfully been developed and applied in LCA (Lindeijer 2000, Baitz 2002, Milà i Canals et al. 2007, Koellner et al. 2013). For the advancement of these already existing and well established methods the availability and further development of characterization factors (CF) based on scientifically acknowledged methods is crucial.

The LANCA[®] (Land Use Indicator Value Calculation in Life Cycle Assessment) method was developed at the department Life Cycle Engineering in 2010 and refined in 2016 (Beck 2010, Bos 2016). It addresses several indicators for ecosystem services. Using this method we calculated a set of country and land use type specific characterization factors representing impacts on ecosystem services and published this set for the integration into Life Cycle Impact Assessment. The application of the characterization factors in LCA studies showed that the choice of the reference situation is crucial in the calculation of land use aspects.

2. Methods

For the consideration of land use aspects in LCA and the calculation of CFs from ecosystem service quality indicators the concept of transformation and occupation is decisive. This general concept is explained in the next paragraphs. Subsequently, the land use impact categories and their calculation methods are explained briefly.

Transformation and Occupation

There is a consensus amongst scientists that land use can be separated into a transformation phase and an occupation phase (Lindeijer 2000, Baitz 2002, Milà i Canals et al. 2007). According to Koellner et al. (2013) and the ELCD flow list (European Commission, JRC Ispra 2015) the transformation phase is further separated into "transformation from" and "transformation to". The published characterization factors are calculated as follows within the LANCA[®] framework:

$$CF_{\text{transformation from}} = -\Delta Q_{\text{transformation from}} = -(Q_{\text{ref}} - Q_{\text{LU,previous}}) \quad (1)$$

$$CF_{\text{transformation to}} = -\Delta Q_{\text{transformation to}} = -(Q_{\text{LU,prospective}} - Q_{\text{ref}}) \quad (2)$$

$$CF_{\text{occupation}} = -\Delta Q_{\text{occupation}} = -(Q_{\text{LU,current}} - Q_{\text{ref}}) \quad (3)$$

The transformation and the occupation values represent the ecosystem quality difference (ΔQ) between the reference system (Q_{ref}) and the chosen type of land use (Q_{LU}), respectively. The characterization factors are multiplied by (-1) since within life cycle thinking the decline in the ecosystem quality is equal to an impact on the environment and is therefore expressed in positive values. Alike to this, the improvement of the ecosystem quality equates to a negative impact, showing a benefit for the environment. The reference situation in Life Cycle Impact Assessment describes a reference in a region in relation to the current use of a piece of land. An example can be the area of natural land without any anthropogenic influences. The reference situation is used as a basis to calculate the quality differences and the corresponding CFs. With this approach it is possible to calculate the transformation phase regardless of the intermediate stages. The characterization factors “transformation from”, “transformation to” and “occupation” are explained in detail within the next paragraphs.

“Transformation from” means the transformation from a previous land use type and respective ecosystem quality $Q_{\text{LU,previous}}$ (e.g. from grassland) to the reference situation Q_{ref} .

Positive values are equal to an improvement in ecosystem quality, whereas negative values show a decline (see Figure 1). Related to the differences in the ecosystem quality within Life Cycle Impact Assessments, positive values imply an additional impact on the environment (see axis “I”), whereas negative values are equal to an improvement for the environment. Thus, as described above the ecosystem quality difference is multiplied by (-1). “Transformation to” means the transformation from a reference situation Q_{ref} to a prospective land use type and ecosystem quality $Q_{\text{LU,prospective}}$ (e.g. artificial areas).

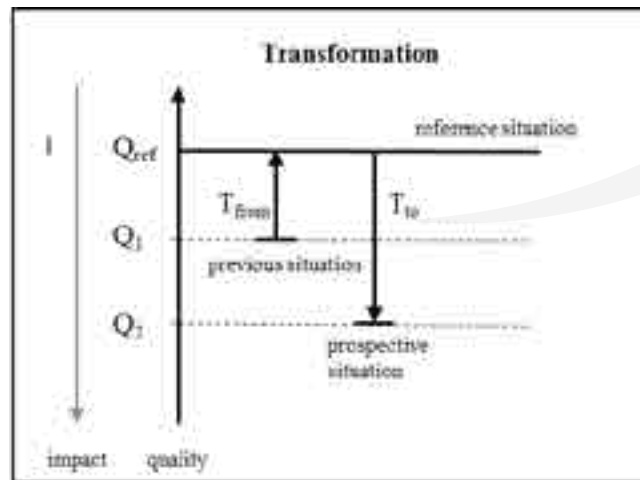


Figure 1: Schematic representation of the land transformation calculation for the LANCA[®] indicators

Occupation represents the difference in ecosystem quality, between the reference situation compared to the current type of land use and its respective ecosystem quality $Q_{\text{LU,current}}$ (e.g. the excavation area for occupation, mineral extraction). As shown in Figure 2, positive quality levels mean degradation, negative values an improvement of the ecosystem quality. And again, positive values on the “I” axis mean an additional impact and the ecosystem quality difference has to be multiplied by (-1).

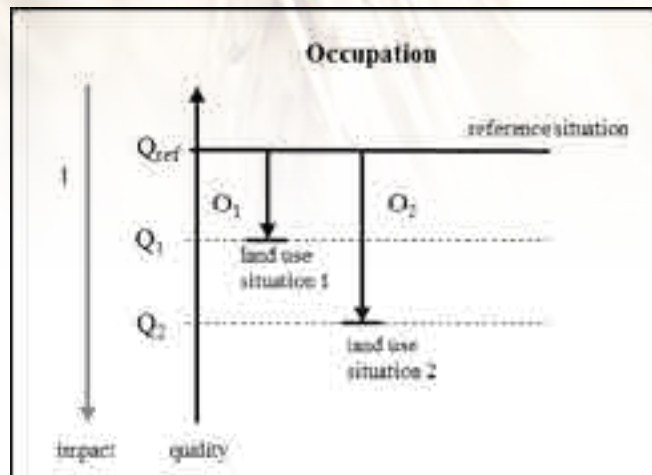


Figure 2: Schematic representation of the land occupation calculation for the LANCA[®] indicators

Ecosystem Service impact categories

Within the LANCA[®] framework five land use impact categories are calculated: Erosion Resistance, Mechanical Filtration, Physicochemical Filtration, Groundwater Replenishment, and Biotic Production. These are described briefly in the following paragraphs:

Soil erosion describes the process of removing and transporting soil particles by the means of water or wind, which occurs if the inherent resistance of the soil against mechanical influences no longer exists (Blume et al. 2010). The resulting loss of soil implies serious effects on the environment including impacts on the water and nutrient cycle, the root depth and the productivity of the soil (Yang et al. 2003). The resistance to soil erosion constitutes an important function of natural ecosystems and should therefore be considered as an indicator for impacts due to transformation and occupation of land. In order to estimate the soil erosion rate, a revised version of the Universal Soil Loss Equation (RUSLE) has been developed by Renard et al. (1997) including modified calculation methods for the different factors. The RUSLE model is used as a basis for the calculation of Erosion Resistance in LANCA[®].

Mechanical Filtration is being described as the capacity of the soil to be mechanically infiltrated by a suspension (Marks et al. 1989). The impact category Mechanical Filtration differs hereby from the category Physicochemical Filtration. Mechanical Filtration is being modeled as the amount of water that can be infiltrated into a specific soil, whereas Physicochemical Filtration describes the amount of adsorbable cationic pollutants. According to Beck et al. (2010), Mechanical Filtration is determined based on soil texture, distance from surface to groundwater and surface sealing.

Physicochemical Filtration of a soil is characterized by its ability to fix and exchange cations to clay and humus particles, also considering the pH dependency of the adsorption intensity to humus. This quantity is called effective cation exchange capacity. The Physicochemical Filtration is calculated using both the potential and the effective cation exchange capacity based on the properties of the soil as well as the degree of surface sealing (Bos 2016).

Groundwater Regeneration represents the capacity of the soils to regenerate ground water sources. This ability is dependent on three different factors: the surface vegetation, the climatic zone as well as the structure of the soil. The Groundwater Regeneration is calculated based on information on soil, slope and land use type, causing the runoff, as well as precipitation and evaporation (Bos 2016).

Biomass Production or primary production represents the ability of an ecosystem to continuously create spare biomass. This leads to an increasing amount of biomass, which is available at a location over a certain period of time. Biotic Production is determined based on soil properties as well as the surface sealing determining the net primary biomass production (Beck 2010).

3. Results

CFs for the land use impact categories Erosion Resistance, Mechanical Filtration, Physicochemical Filtration, Groundwater Regeneration, and Biotic Production are calculated and the respective CFs are deviated and published in Bos et al (2016). The calculation of the CFs is in line with the ILCD handbook (European Commission, JRC 2010) principles for characterization factors as well as the

demands for the integration into life cycle databases such as GaBi or ecoinvent. The respective inventory flows for the CFs are part of the ELCD – European reference Life-Cycle Database (European Commission, JRC 2015).

All characterization factors have been calculated according to the basic structure depicted in Figure 3 and are provided as country-specific CF values. These can be applied within LCA studies. The characterization factors have been calculated with various spatial datasets: For each impact category, all spatially differentiated input quantities required have been provided as global GIS datasets. These data served as a basis for the calculation of country-averages for each CF, that are used as input values for the LANCA[®] tool.

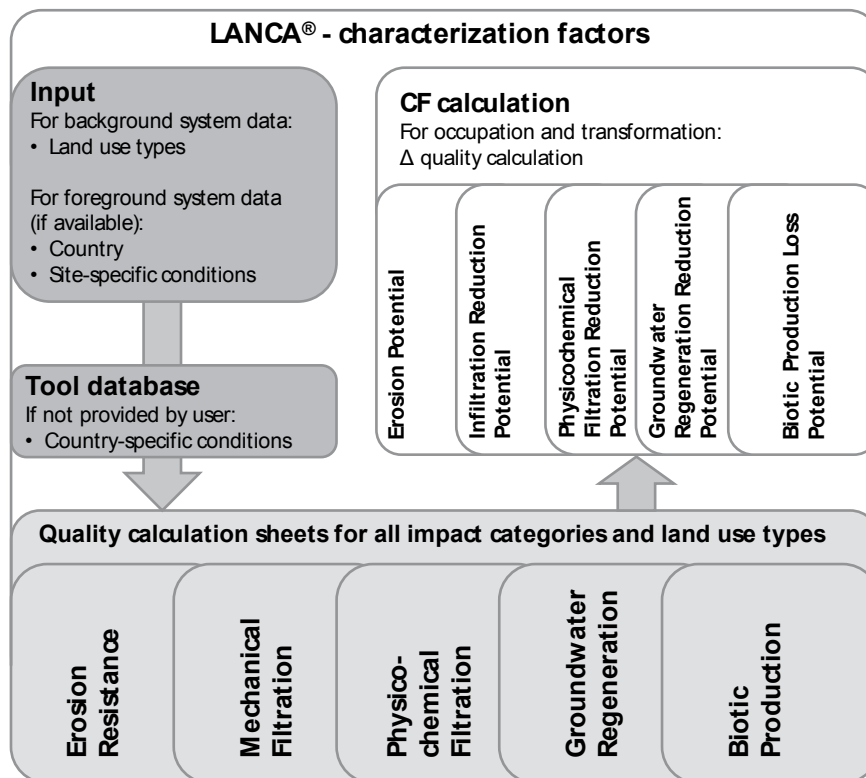


Figure 3: structure for the LANCA[®] characterization factor calculation

Furthermore, respective datasets have been superimposed on another in GIS in order to determine the main climate and biome for the particular country. The country-specific climate zone or biome has then been determined by the largest share within the country. As reference situation a natural state of land use has been presumed based on WWF terrestrial biomes provided by Olson et al. (2001) similar to the recommendation of Milà i Canals et al. (2007). WWF distinguishes 867 different ecoregions that cover the overall terrestrial land surface. The same classification has already been used also as reference situation for other impact assessments in the LCA framework.

The following spatial datasets are used: land use types are derived according to Arana Benitez (2015) for the inventory flows provided by European Commission, JRC Ispra (2015). Soil properties are calculated based on the Harmonized World Soil Database (Nachtergaele et al. 2012). Furthermore, climate zones have been used according to the Köppen-Geiger climate classification (Rubel and Kottek 2010) and main biomes according to Olson et al. (2001).

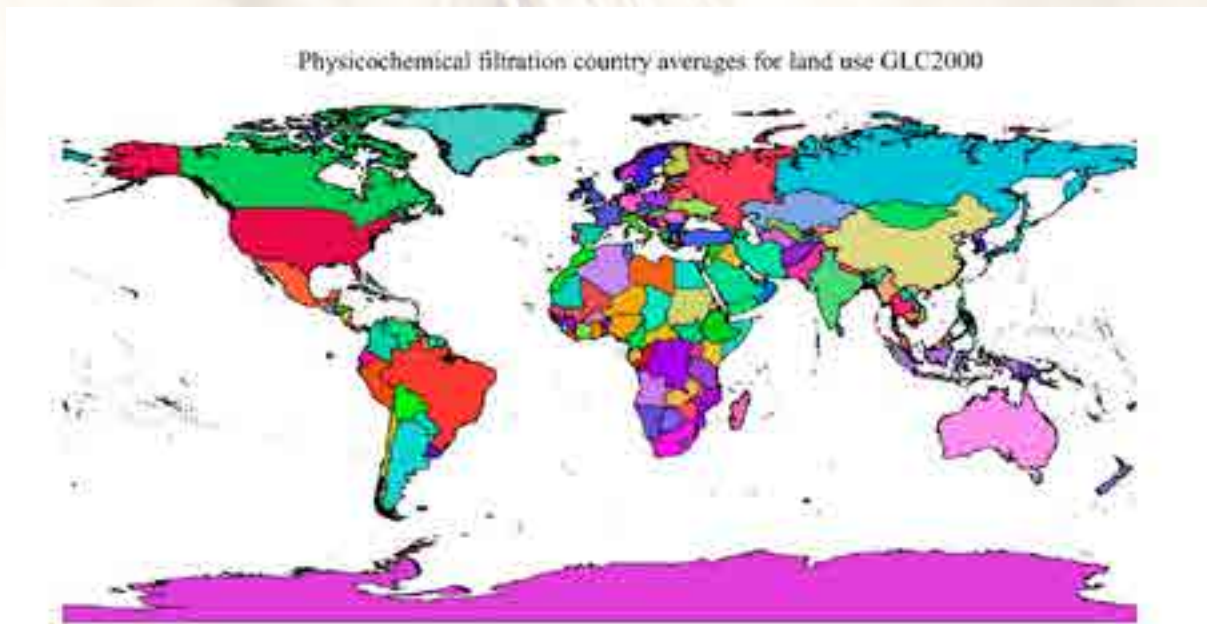


Figure 4: Country averages for the Physicochemical Filtration for the GLC2000 land use

Figure 4 shows a map of the world representing the ecosystem quality indicator Physicochemical Filtration using GLC2000 as land use type classification. Each country has its own specific quality value for the indicator Physicochemical Filtration and is therefore in a different color. Overlaying this map with a map using the WWF Terrestrial Ecoregions of the World as land use type the transformation and occupation impacts can be calculated.

The provided CFs objectify the ongoing discussion within the scientific community regarding the choice of the reference situation as a crucial factor influencing the results. The question remains: Which reference situation is the most appropriate one? Koellner et al. (2013) suggest three different options of reference situations: Option 1 is the concept of PNV (potential natural vegetation). Option 2 is the (quasi-)natural land cover in each biome/ecoregion. Option 3 is a current mix of land uses. Option 1 and Option 2 can be seen as similar, so we calculated an example using the natural land cover as well as an actual or desired land cover situation as reference. Figure 5 shows an example addressing a current discussion on organic farming as the desired land use situation for the CF Erosion Potential, Occupation by using two alternative reference situations for non-irrigated intensively used arable land: First, "mixed forest, primary" as natural status according to the WWF Terrestrial Ecoregions of the World; second, "arable, non-irrigated, extensive" as a politically desired land use situation. Using the reference situation "mixed forest" instead of the politically desired situation "arable, non-irrigated, extensive" leads to far higher calculated impacts regarding the impact category Erosion Resistance. The example clearly shows that depending on the reference situation, the same use of a piece of land may lead to very different results.

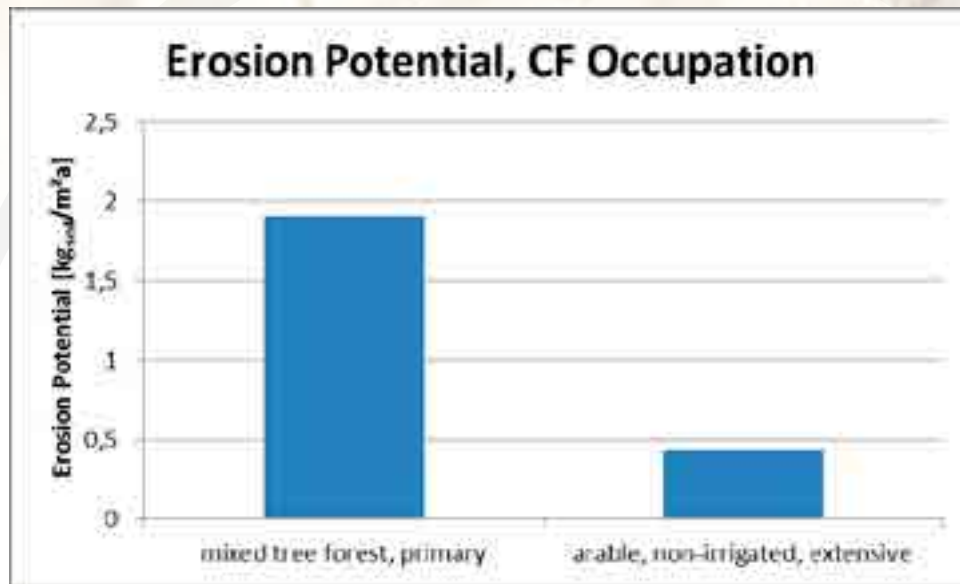


Figure 5: characterization factor for the erosion potential occupation for the land use type arable, non-irrigated, intensive using two different reference situations: mixed forest, primary and arable, non-irrigated, extensive

4. Discussion and Conclusion

Through the publication of the characterization factors ecosystem services can be addressed consistently in LCA studies and results are comparable.

However, the choice of the reference situation is not finally discussed: Especially for the use of a piece of land as arable land, the PNV concept might not be appropriate because if it weren't this specific crop, another crop would realistically be planted on the same piece of land. Thus, the specific piece of land would be arable land in any case. Nevertheless, when thinking of the concept of LCA every unit of emission that is emitted is compared to a virtual "zero emission process". Thus the reference situation is "zero emissions" or an environment without emissions that can be transferred for land use impact assessment to a natural environment. This thinking strengthens the argument for using PNV as the reference situation.

One possible solution for this discrepancy could be to provide one set of CFs for each reference situation. This way, the decision about the reference state can be made by the practitioner, taking into consideration the goal and scope of the study.

This open issue is subject for further research but will certainly be addressed soon in order to provide an as realistic as possible and consistent picture of land use aspects in LCA.

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94. Past and future soybean land transformation impacts in Mato Grosso, Brazil

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ABSTRACT

The state of Mato Grosso is the largest soybean producer in Brazil with production in both Amazon tropical forest and Cerrado savanna biomes. In this study, we test how LCA might help evaluate an agricultural product's environmental performance considering past and future land transformation impacts to biodiversity and ecosystem services. We apply the 2013 UNEP-SETAC land transformation guidelines to derive impacts of soybean production in Mato Grosso's Amazon and Cerrado biomes considering past expansion (2001-2014) and three distinct expansion scenarios for the 2015-2020 period considering business-as-usual increase in cropland. Results show that a total soybean area of 14.8 Mha produced in the state of Mato Grosso could lead to a total cumulative impact to biodiversity as high as $2.6 \cdot 10^{12}$ PDF $m^2 y$ (Amazon) and $4.1 \cdot 10^{12}$ PDF $m^2 y$ (Cerrado), and a total cumulative damage to ecosystem services of $\$1.36 \cdot 10^{11}$ (Amazon) and $\$1.22 \cdot 10^{11}$ (Cerrado). Cropland expansion into pasture showed little improvements in the Cerrado compared to the Amazon biome which could see its cumulative end-point impacts minimized to that of the Cerrado's for the 2011-2020 period. Our study provides some insight into the use of LCA focusing specifically on land use, and could inspire similar studies in other regions seeking to improve product environmental performance.

Keywords: agricultural expansion, deforestation, land use change, biodiversity, ecosystem services.

1. Introduction

Brazilian soybean production has grown considerably since the 1990s, especially in the state of Mato Grosso located in the country's Central Western region (Figure 1). Planted area more than doubled between 2000 and 2010 from 3 Mha to 6 Mha, and reaching 8.6 Mha in 2014 (IBGE, 2016). This increase in production was possible through cropland expansion in the Cerrado and Amazon biomes where natural vegetation cover respectively comprises a mixture of savanna landscapes (shrubs, scrub forest, woodlands) and deciduous and semi-deciduous transition forest. The link between land use change and Brazilian policies, international markets, and demand in commodities has been of interest in the land use sciences (Gibbs et al., 2015; Nepstad et al., 2014), with studies using high resolution remote sensing information to understand deforestation dynamics with agricultural expansion. These studies have been successful in relating the effects of land use and land cover change to impacts on the carbon and water cycles, but oftentimes cannot make a direct link to a specific production system at scales larger than field level.

Recent supply-chain initiatives have contributed to the decrease in deforestation, such as the Soybean Moratorium in 2006, in addition to greater law enforcement of the Brazilian Federal Forest Code and access to credit, in order to increase opportunity costs of deforestation (Nepstad et al., 2014). The use of life cycle assessment (LCA) can inform such initiatives by assessing the soybean's environmental performance based on a list of options within the regional production context. Given the availability of high resolution data for the region, we propose to carry out a LCA focusing specifically on land transformation impacts with two objectives: (1) to establish the impacts to biodiversity and ecosystem services of soybean produced over a 20-year timeframe (2001-2020); (2) to inform the best land use option for soybean production in the region given future land use considerations (2015-2020). Combining past and future impacts could provide further indication on the ability to improve environmental performance of soybean in Mato Grosso considering planned expansion.



Figure 1: The state of Mato Grosso, located in Central-western Brazil and containing the Amazon and Cerrado biomes of interest for this study

2. Methods

We apply 2013 UNEP-SETAC guidelines for land transformation impacts (Koellner et al., 2013) along with high resolution remote sensing information to assess the total impacts of soybean production in the state of Mato Grosso between 2001 and 2020, considering three production pathways for the 2015-2020 period. Land transformation mid-point impacts were calculated following equation (1) (Koellner et al., 2013)

$$I_{mid} = \frac{1}{2} CF_{occ} t_{regen} A_{trans} \quad (1)$$

where I_{mid} is the total land transformation impact, CF_{occ} (impact/m²) is the characterization factor of land occupation, t_{regen} (y) is the time required for land to regenerate back to natural vegetation (159 years and 117 years for the Amazon and Cerrado biomes, respectively, according to Curran et al. (2014)), and A_{trans} is the area transformed (m²) for soybean production. Regeneration processes are assumed to be linear over t_{regen} at a constant rate expressed by the factor of 1/2 in equation (1). Land transformation damage to ecosystem services were calculated following Cao et al. (2015), shown in equation (2)

$$I_{end} = ECF(CF_{trans}) XF AC A_{trans} \quad (2)$$

where $ECF(CF_{trans})$ is the economic conversion function (\$/physical parameter) calculated as a function of the characterization factor CF_{trans} (\$/(m² y)), XF is the exposure factors (equal to 1) and AC is the adaptation capacity (equal to 0.84 for Brazil).

Our functional unit is the soybean production system represented annually by the amount of land used for production within the political boundaries of the state of Mato Grosso between 2001 and 2020. Impacts were summed cumulatively between 2001 and 2020, meaning that impacts of transformation occurring in 2001 were added to those between 2002 and 2020, while impacts of transformation occurring in 2015 were added to those between 2016 and 2020. We take 2000 as the first year from which to sum land transformation impacts due to the lack of spatial information prior to this date.

Land transformation impacts considered were assessed at the mid-point level for Biodiversity Damage Potential (BDP, PDF m² y, with PDF being the potentially disappeared fraction of species) (de Baan et al., 2013), and at the end-point level for Erosion Resistance Potential (ERP, ton), Water Purification Potential (Mechanical Filtration, WPP-MF, and Freshwater Regulation Potential, FWRP,

m³) (Saad et al., 2013), Biotic Production Potential (BPP, ton C) (Brandão and Milà i Canals, 2013), and Climate Regulation Potential (CRP, ton C) (Müller-Wenk and Brandão, 2010) following equations (1) and (2). Characterization factors were calculated for both Amazon and Cerrado biomes constrained to the state of Mato Grosso using global soil information (Shannguan et al., 2014), as well as local information on biomass (above and below ground), soil organic carbon (Maia et al., 2010), topography (Jarvis et al., 2008), evapotranspiration on land (Lathuillière et al., 2012), and local information on species richness (e.g. Salórzano et al., 2012).

Values of A_{trans} were obtained by combining agricultural production data (IBGE, 2016) with high resolution remote sensing information (Gibbs et al., 2015). Remote sensing information provided details on natural vegetation transformed following 2000, while transformation of pasture into soybean was deduced using pasture area derived by Lathuillière et al. (2012). Pasture is often seen as a transition land cover following transformation of natural vegetation in both Amazon and Cerrado biomes, with large areas of soybean having replaced pasture between 2000 and 2010 (Macedo et al., 2012). Due to lack of information prior to 2000, we assumed that pasture being replaced by soybean had been utilized as pasture for more than 20 years such that land transformation impacts were first allocated to pasture before soybean.

We assumed business-as-usual soybean expansion following 2014, representing 15 %/y (Amazon) and 5 %/y (Cerrado) since 2010 (IBGE, 2016). For each annual new expansion between 2015 and 2020, we considered three possible scenarios: (A) 100 % expansion into natural vegetation, (B) 50 % expansion into natural vegetation and 50 % into pastureland (Figure 2).

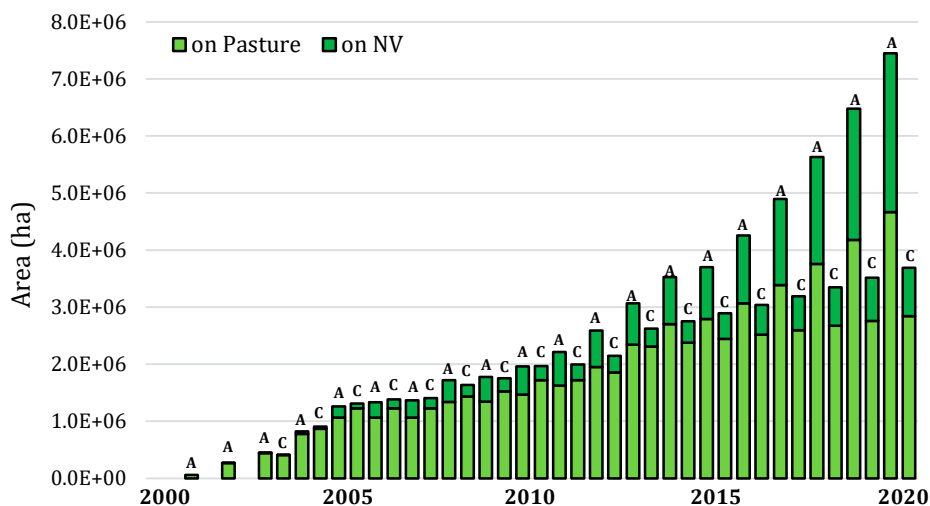


Figure 2: Total cumulative land transformation since 2000 for soybean in Mato Grosso's Amazon (A) and Cerrado (C) biomes (2000-2014) and predicted area transformed following scenario B for the 2015-2020 period (50 % expansion into natural vegetation (NV) and 50 % into pasture). Each year is assigned both an Amazon and Cerrado transformation.

3. Results

3.1. Twenty years of land transformation impacts of soybean in Mato Grosso

Total land transformation impacts were more prominent in the Cerrado than in the Amazon due to the larger expansion for soybean in the biome until 2010 (Figure 3), with impacts carrying into the 2010-2020 period. A business-as-usual cropland expansion would lead to a total cumulative biodiversity impact of $2.6 \cdot 10^{12}$ PDF m² y and $4.0 \cdot 10^{12}$ PDF m² y, and a total cumulative damage to ecosystem services as high as $\$1.36 \cdot 10^{11}$ and $\$1.22 \cdot 10^{11}$ in the Amazon and Cerrado biomes, respectively.

The largest contributor to total damage to ecosystem services came from WPP-MF representing up to 59% (Amazon) and 47% (Cerrado), followed by BPP at 13% (Amazon) and 19% (Cerrado), and FWRP at 11% (Amazon) and 17% (Cerrado). Despite greater transformation in the Cerrado, CRP damage was greater in the Amazon biome (up to $\$1.84 \times 10^{10}$ compared to $\$5.29 \times 10^9$ for the Cerrado), and within the range of BPP ($\$1.46 \times 10^{10}$ (Amazon) and $\$2.23 \times 10^{10}$ (Cerrado)).

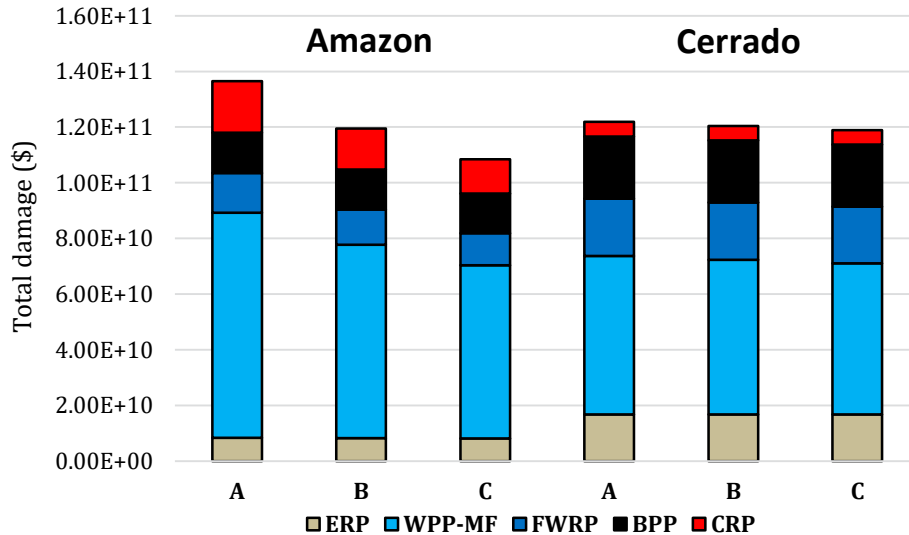


Figure 3: Total cumulative damage to ecosystem services from land transformation in Mato Grosso's Amazon and Cerrado biomes for the 2001-2020 period considering three production pathways for the 2015-2020 period: (A) 100% expansion into natural vegetation, (B) 50% expansion into natural vegetation and 50% into pasture, (C) 100% expansion into pasture. ERP: Erosion Resistance Potential; WPP-MF: Water Purification Potential – Mechanical Filtration; FWRP: Freshwater Regulation Potential; BPP: Biotic Production Potential; CRP: Climate Regulation Potential.

3.2. Hotspots of production for the 2001-2020 period

Expansion in the 2011-2020 period showed greater cumulative damage to ecosystems services when compared to the 2001-2011 period for both the Amazon and the Cerrado. In 2010, total impacts to biodiversity had reached 1.11×10^{11} PDF m^2 y (Amazon) and 2.65×10^{11} PDF m^2 y (Cerrado) compared to 1.36×10^{12} PDF m^2 y (Amazon) and 1.14×10^{12} PDF m^2 y (Cerrado) in 2011-2020 (scenario A). Similarly, total cumulative damage to ecosystem services had reached $\$5.29 \times 10^9$ (Amazon, Figure 4) and $\$7.37 \times 10^9$ (Cerrado, Figure 5) in 2010, compared to $\$7.83 \times 10^{10}$ (Amazon) and $\$4.08 \times 10^{10}$ (Cerrado) in 2011-2020 (scenario A). Despite larger expansion into the Amazon biome, scenario C showed a similar cumulative damage to ecosystem services when compared to all expansion scenarios in the Cerrado, but a biodiversity damage of 8.50×10^{11} PDF m^2 y one order of magnitude lower than in the Cerrado (1.05×10^{12} PDF m^2 y).

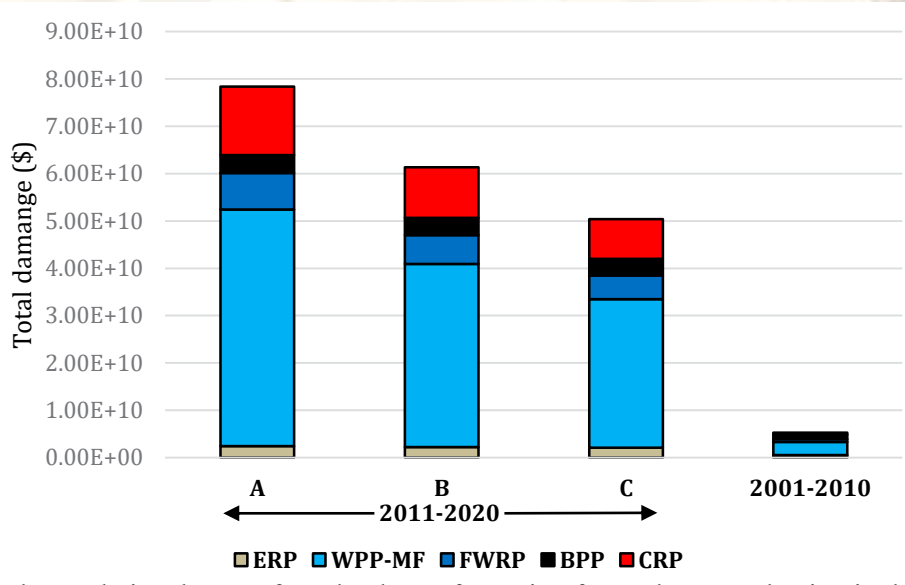


Figure 4: Total cumulative damage from land transformation for soybean production in the 2011-2020 period compared to 2001-2010 in the Amazon biome considering three production pathways for the 2015-2020 period: (A) 100% expansion into natural vegetation, (B) 50% expansion into natural vegetation and 50% into pasture, (C) 100% expansion into pasture. ERP: Erosion Resistance Potential; WPP-MF: Water Purification Potential – Mechanical Filtration; FWRP: Freshwater Regulation Potential; BPP: Biotic Production Potential; CRP: Climate Regulation Potential.

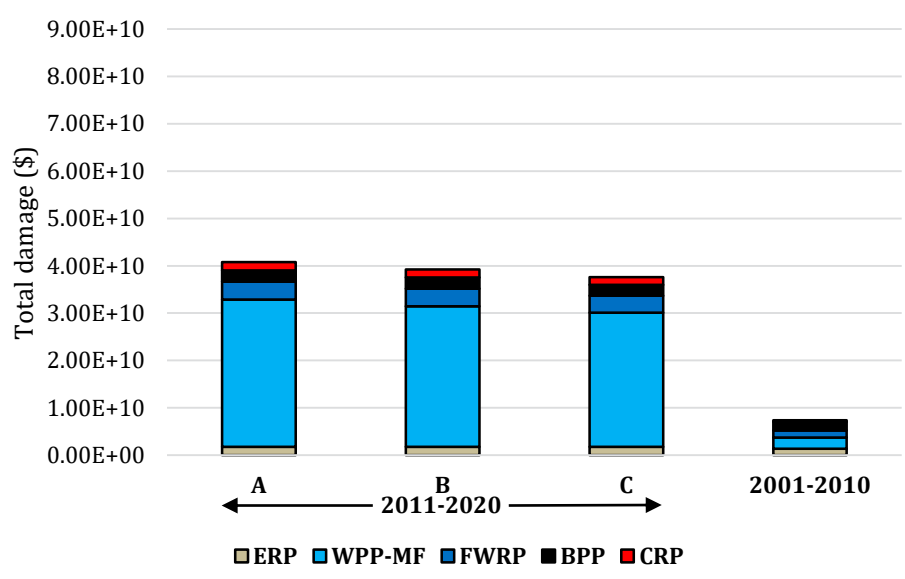


Figure 5: Total cumulative damage from land transformation for soybean production in the 2011-2020 period compared to 2001-2010 in the Cerrado biome period considering three production pathways for the 2015-2020 period: (A) 100% expansion into natural vegetation, (B) 50% expansion into natural vegetation and 50% into pasture, (C) 100% expansion into pasture. ERP: Erosion Resistance Potential; WPP-MF: Water Purification Potential – Mechanical Filtration; FWRP: Freshwater Regulation Potential; BPP: Biotic Production Potential; CRP: Climate Regulation Potential.

4. Discussion

4.1. Land transformation impacts in the Amazon and Cerrado biomes

The land transformation history of Mato Grosso has given some indication on the expected total environmental impact of soybean production within the 20-year timeframe proposed for allocation (Koellner et al., 2013). Historically, soybean has been more established in the Cerrado biome of Mato

Grosso due to more suitable soils, production costs, infrastructure, and the predominance of pasture in early settlements, but also looser restrictions on deforestation: the Brazilian Federal Forest code of 1965 (updated in 2012) mandated the retention of 80% of natural vegetation cover for properties located in the Amazon, but 20-30% for properties in the Cerrado (50% for the Cerrado/Amazon transition area) (Fearnside and Barbosa, 2004; Brannstrom et al., 2008). The importance of CRP and BPP, apparent in the total damage from the Amazon biome, also reflects some of the initiatives that Brazil has taken to reduce deforestation and increase soil organic carbon content as a national strategy to mitigate greenhouse gas emissions.

4.2. Using LCA to improve land use environmental performance

Our results could help with the land use selection as a means to improve Mato Grosso's soybean environmental performance considering the 20-year horizon. Without the cumulative effects of land transformed prior to 2000, cropland expansion into already established pasture would reduce total damage. Given current policies and incentives (Nepstad et al., 2014), cropland expansion will likely continue onto natural vegetation and pasturelands as described in our scenario B. An initial attempt to improve the environmental performance of soybean could focus on a pasture to cropland conversion. This conversion would prevent further harm to biodiversity while limiting total damage to ecosystem services. A cropland to pasture expansion in the Amazon would reduce such damage, but would also carry additional impacts that would be counteractive to Brazil's goals to reduce greenhouse gas emissions (CRP, BPP). On-farm water management (WPP-MF and FWRP) and strategies to reduce losses of soil organic carbon (BPP) would greatly improve soybean production in all biomes. While water management still needs to be addressed in the exclusively rain-fed soybean production systems, no-till planting has been highly promoted in the region as a way to reduce losses of soil organic carbon.

Our proposed option to expand 2015-2020 production on pastureland (scenario C) assumes an increase in cattle density on current pasture, as opposed to future pasture expansion onto natural vegetation. In the early 2000s, cropland expansion into pasture in the Cerrado biome led to pasture expansion into the Amazon biome (Barona et al., 2010). Such indirect land use change has not been quantified in this study but would lead to exactly the same impacts as scenario A if natural vegetation were transformed for all pasture displaced by cropland. Other options may include cropland intensification on current land as a way to improve yield, but such options will rely on agricultural inputs, especially fertilizer and the possible future use of irrigation. These options were not considered here, but should also be included when considering future input related environmental performance scenarios for the region.

4.3. Considerations for similar studies using land use in LCA

This study has highlighted some current limitations to LCA to improve environmental performance both in the methodology and decision-making aspects. The 20-year time horizon in our study was chosen to coincide with convention for allocation of land transformation impacts (Koellner et al., 2013). Thus, the impacts allocated to soybean in the early 2000s would no longer be included post-2020, making this type of study difficult to have an impact on decision making at the product level. Should the time horizon be extended to 30 years, land use optimization in our study could become more meaningful as more improvements could be made to the production system along the longer time horizon, and allocation of impacts to pasture. However, a longer timeframe also suggests greater uncertainty in the projected land use, which could also complicate decision-making for the production system. Here, we have made a business-as-usual assumption for the 2015-2020 period, but other, non-conventional cropland expansion scenarios, could be considered (including yield increase through greater agricultural input).

The study results depend greatly on the information available for the timeframe under consideration with the final impact also changing with the choice of the initial year of impact assessment. These differences are apparent when comparing the 2001-2010 and 2011-2020 time frames as each 10-year interval clearly displays differences in total cumulative land transformation impacts. Such information highlights additional challenges for impact assessment and the need for detailed guidelines for LCA practitioners that are considering land use change in their studies.

The use of remote sensing to determine A_{trans} as our inventory has been beneficial for our study, and similar information should be considered for other studies. Publicly available remote sensing products are increasingly being available through platforms that are made easy to use (e.g. Google Earth Engine, Google Inc., <https://explorer.earthengine.google.com>). Similar to our values of A_{trans} , the biome averaged characterization factors could benefit from high resolution biophysical information available through global databases and remote sensing available from land use sciences such as soil type and organic matter content (Shannguan et al., 2014), above- and belowground biomass, or evapotranspiration (*sensu* Lathuillière et al., 2012 with MODIS). More regionalized characterization factors might help identify land use strategies similar to the ones described here, but defined within a specific biome.

5. Conclusions

We have applied LCA to assess total impacts to biodiversity and damage to ecosystem services for soybean production in Mato Grosso for the 2001-2020 period. A 20-year time period of allocation for the impacts of land transformation has been used, corresponding to the recommended timeframe from the 2013 UNEP-SETAC guidelines (Koellner et al., 2013). LCA could inform land use decision-making in the production system considering enough information is available on land transformation with scenarios that are representative for the region. In a business-as-usual soybean expansion scenario for Mato Grosso, our results suggest that 2015-2020 expansion should focus on cropland expansion into pasture areas with an increase in cattle density on current pastureland, especially in the Amazon in order to minimize environmental impacts, along with considerations from soil erosion, soil organic carbon and changes in soil compaction as an important intervention in the production system.

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227. How EU27 is outsourcing the vast majority of its land and water footprint

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ABSTRACT

Water consumption and scarcity problems are mainly caused by agricultural production. Highly populated and economically developed regions such as the EU27 import a large share of their products from other regions. Thereby they induce environmental impacts in these regions and put the resource availability at risk, affecting their future food security. In order to trace the water consumption and related impacts of EU27 final demand, we combine multi-regional input-output data with detailed water consumption estimates of crop production. This allows tracing back the water consumption through the full supply chain of final demand to the originating watersheds. In a second step, impact assessment methods are applied to evaluate water scarcity effects on this level of spatial detail. The results show that the EU27 states are outsourcing the majority of their water scarcity impacts (65-75%) to other regions. While different methods draw a variant picture, they consistently show the high dependency on other regions, which leads to responsibilities of the EU27 in these regions in order to comply with the polluter pays principle and also secure future food supply.

Keywords: Agriculture, international trade, environmental outsourcing, life cycle impact assessment

1. Introduction

Global agriculture is responsible for ~85% of water global consumption (Shiklomanov and Rodda 2003) and therefore the main driver water scarcity and related impacts, while food products are traded in a highly globalized market. This has been shown in several studies, indicating the tele-connections of producer and consumer regions (Hubacek et al. 2014). In order to connect impacts of producer to final consumption, we applied the EXIOBASE multi-regional input output (MRIO) dataset, which is accounting for water consumption and trade of 43 individual countries (~95% of the global GDP) and 5 rest-of-the-world (RoW) regions (Wood et al. 2014). Each region has 163 industrial sectors and for agriculture, 8 different crop sectors are available in Exiobase.

Water footprints have emerged as an area of high public interest, which finally led to the creation of an ISO standard (ISO 2014) as a result of international consensus building. Many methods exist (Kounina et al. 2013), while most methods use either a water scarcity index (WSI, Pfister et al. 2009) or an approach from the NGO “water footprint network” (Mekonnen and Hoekstra 2011). More recently, an international working group was formed to harmonize the different approaches and recommend a method. The preliminary recommendation is the so called Aware method, which combines natural water scarcity and water stress induced by humans in one single number through land use equivalents required to regenerate the water consumed sustainably (Boulay et al. 2015).

MRIO facilitates a more complete water footprint assessment of final consumption than bottom-up approaches using trade data (Feng et al. 2011). Therefore, this work builds upon a recently published study that combines EXIOPOL data and detailed information on water consumption of 160 related to final consumption by the EU27 states (Lutter et al. 2016), which only reported water consumption and impacts in terms of the water footprint network approach.

2. Methods

Since water consumption and impacts vary regionally, we created a spatial disaggregation matrix that allows allocating the consumption of a crop group in each region to >10'000 watersheds as a function of the production pattern and subsequently the application of water scarcity characterization factors (Figure 1). Water consumption estimates of 160 crops on high spatial detail is taken from Pfister and Bayer (2014) and combined with MRIO data EXIOBASE (Wood et al. 2014). Details on how these results are calculated are reported in detail by Lutter et al. (2016).

In this work we present and extended this analysis in terms of LCIA by using Aware (with a range from 0.1 to 100) and water stress index (WSI) ranging from 0.01 to 1 on top of the blue water scarcity (BWS) method which ranges from 0-12 months per year (count of months under water scarcity). Everything is calculated on watershed level.

3. Results

Figure 1 shows “green water” (rain-fed water; as a land use indicator) and “blue water” (irrigation) footprints of EU27 final consumption. The “blue water” footprint of EU27 final consumption of products is mainly located in Europe, the US, China, India, Pakistan and Brazil, while scarce water originates mainly from Europe, India, Pakistan, the US, China, and Egypt. For green water, Sub-Saharan Africa and Latin America have a much higher contribution, indicating high land use impacts caused by EU27 production, since green water is a potential proxy for land use impacts.

The share of green, blue and scarce water consumed within the EU27 for its final consumption is between 25 and 35% of the total, reflecting the high dependency of EU states on foreign land and water resources. This is supported by the fact that different water scarcity methods, including the recommended method from the recent UNEP-SETAC Pellston workshop on LCIA methods, identify Indus as the highest contributor to water scarcity impacts, followed by the Guadalquivir (shown in figure 1 for BWS). However, different stress indicators result different hotspots, such as shown for the relevance of Nile and Mississippi river, where a high discrepancy is observed among the methods (Table1). Other rivers of high relevance and high discrepancy include Tagus, Danube, Po, Ganges, Ebro and Gaudiana.

The differences among the methods become also visible when comparing the total amount of scarcity covered by the top 15 watersheds presented in Table 1: while BWS and Aware attribute 71% and 67%, respectively, of all scarcity to these rivers, WSI only attributes 50% to these rivers. A large portion of this difference can be attributed to relevance of the Indus watershed for water scarcity in EU27 final demand. This indicates the importance of testing different indicators for assessing hotspots in the supply chain.

While the origins are of most interest, it is also relevant through which product group the impacts are caused. Water footprint is mainly caused by agricultural products, which are mainly imported as processed bio-based products through other sectors, incl. processed food and leather.

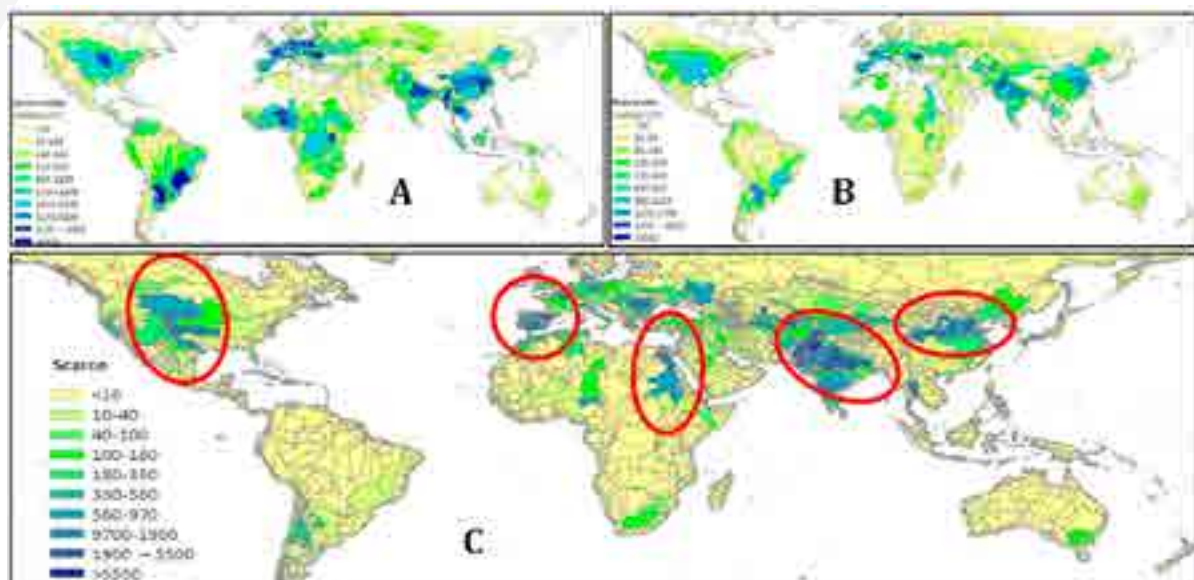


Fig. 5: Water consumption in million m³ (Mm³) per year and watershed from EU27 final consumption: (A) Green water consumption (rain-fed); (B) blue water consumption (irrigation); (C) Scarce water consumption representing midpoint impacts for water scarcity based on Mekonnen and Hoekstra (2011). Red circles indicate water scarcity hotspot (cf. Table 1). Adjusted from (Lutter et al. 2016)

Tab. 1: Top 15 producer watersheds for EU27 final consumption, sorted by blue water consumption. The shares are also presented in terms of impact after characterization with three different methods (incl. CFs): Blue water scarcity (BWS, Mekonnen and Hoekstra 2011), Aware (Boulay et al. 2015) and WSI (Pfister et al. 2009).

Watershed	Water consumption		Blue water scarcity		Aware		WSI	
	[Mm3]	Share	CF [month/year]	share	CF [-]	share	CF [-]	share
Indus	25'107	12%	12	40.2%	60.8	32.9%	0.83	20.9%
Danube	9'485	5%	0	0.0%	1.2	0.2%	0.07	0.7%
Mississippi	8'895	4%	4	4.8%	11.2	2.2%	0.24	2.1%
Quadalquivir	5'459	3%	7	5.2%	60.2	7.1%	1.00	5.5%
Nile	4'850	2%	2	1.3%	100.0	10.5%	0.98	4.8%
Parana	4'472	2%	0	0.0%	0.5	0.0%	0.01	0.0%
Po	4'276	2%	2	1.2%	1.1	0.1%	0.05	0.2%
Amu Darya	4'040	2%	5	2.7%	37.3	3.2%	1.00	4.0%
Ganges	3'658	2%	7	3.5%	17.9	1.4%	1.00	3.7%
Ebro	3'478	2%	3	1.4%	42.6	3.2%	0.26	0.9%
Guadiana	3'259	2%	7	3.1%	22.6	1.6%	0.99	3.2%
Douro	2'732	1%	5	1.8%	24.4	1.4%	0.17	0.5%
Tagus	2'584	1%	5	1.7%	4.3	0.2%	0.53	1.4%
Hai river	1'853	1%	12	3.0%	80.9	3.2%	1.00	1.9%
Chao Phraya	1'512	1%	7	1.4%	3.5	0.1%	0.48	0.7%
Other rivers	115'531	57%		28.7%		32.5%		49.6%

4. Discussion

The results show the effect of our globalized markets: the majority of food impacts are occurring outside the consumer region in the case of EU27. Many reasons might exist, but clearly affluent countries import from less affluent countries, with the main exception of the US. One reason is the low economic revenue of the agricultural sector but also the high population density and competition for land in Europe. However, the vast dependency on imported products and high share of external impacts highlights the need for policy actions to mitigate impacts in producer countries.

Several limitations need to be highlighted. First of all, the MRIO data includes high sector aggregation of the 160 crops into 8 groups. Furthermore, aggregation outside Europe combines many important producers in Africa, Asia the Middle East and Latin America into large regions. While only ~5% of total GDP is affected, a large fraction of water scarcity is located in such areas. Therefore, the level of detail is hampered, since water consumption of aggregated sectors and countries is traced back by the relative production shares, which might not be representative for actual trade.

Additionally, water consumption estimates have high uncertainty (Pfister et al. 2011), as well as the model to estimate water scarcity do (Laura and Stephan 2016; Scherer et al. 2015). Therefore also the results of this analysis are highly uncertain. However, the results are largely consistent among the different stress indicators applied.

Another limitation is that we applied the current water scarcity indicators to total water consumption. Since the indicators are showing current scarcity suitable for assessing marginal changes, the impacts are overestimated. In Theory, the non-marginal water consumption of EU27 final demand needs to be assessed by integrating the water scarcity indicators from water consumption without EU demand to current water consumption (Pfister and Bayer 2014). However, since the correlation is very high between marginal and non-marginal index, this effect is assumed to be lower than those mentioned above.

5. Conclusions

The study shows, that EU27 is outsourcing the vast majority of water scarcity related to its consumption to other regions. A high share is originating from highly water stressed river systems, many situated in poor countries. It is therefore concluded that the EU27 has a high responsibility to improve the situation in these regions to comply with the polluter pays principle, but also to ensure future supply of its agricultural products. For robustness of result it is recommended to use more than one method to assess water scarcity.

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191. Identifying Land Use and Land-use Changes (LULUC): a global LULUC matrix

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ABSTRACT

Land Use and Land-use Changes (LULUC) information are essential to determine the environmental impacts of anthropogenic land-use and conversion. However, existing global satellite imagery provide information on land cover but rarely large scale mapping of land-use type, distinguishing for example between managed and unmanaged land. National statistics show limitation in terms of data quality and consistency between countries and quantify land occupation per land-use type rather than providing information on land-use changes. In order to overcome these limitations, we combined the strengths of the remote sensed global land cover dataset of MODIS Collection 5 and FAOSTAT land-use data. The aim was to obtain a LULUC database including a collection of country-specific LULUC matrixes, as suggested by the IPCC. We produced two versions of the LULUC matrix: version 1 (V1) based on the International Geosphere-Biosphere Program land classification system; version 2 (V2), a refined version disaggregating the land identified as forest by MODIS - based on the canopy cover - into the FAOSTAT forest classes, to distinguish primary, secondary, planted forests and permanent crops. The outcome was a first country-based consistent set of spatially-explicit LULUC matrixes quantifying the land undergoing a change between two reference years, from 2001 to 2012. The database facilitates a more holistic assessment of land-use changes, explicitly quantifying changes occurring between land classes. It could support global scale land-use change analyses, requiring a distinction between land types based not only on land cover but also on land-uses. The matrixes provided a good overview of where and what type of LUC occurred, a crucial information to assess environmental impacts caused by LULUC. The spatially explicit dataset may serve as a starting point for further studies aiming at determining the drivers of land-use change supported by spatial statistical modelling.

Keywords: Land-Use Changes, Agriculture, Forestry, Life Cycle Assessment

1. Introduction

Land plays a fundamental role for human subsistence. Between 42 and 68% of the land surface has been subjected to land use activities during the last 300 years (Hurt et al. 2006). In 2005 Land-use Changes were responsible for 12,2% of the 2005 global GHG emissions (Herzog 2005). Since land is a finite resource with multiple functions, the increasing population and consumption (Garnett et al. 2013), the expansion of bioenergy crops, and the effects of climate change have recently increased the stress on global land resources (Alexander et al. 2015). The study of Land Use and Land-use Changes (LULUC) have therefore become crucial in environmental science (Foley et al. 2011). The European Commission considers Life Cycle Assessment (LCA) as *“the best framework for assessing the potential environmental impacts of products currently available”* (European Commission 2015). Nevertheless, the debate on LULUC modeling has become central in the LCA community. Modelling on a global scale the environmental, economic and social impacts of LULUC has proven controversial in economy-wide models as in input/output and LCA analyses (Ahlgren and Di Lucia 2014, Warner et al. 2014): LUC analyses are often limited in scope to a determined product or region due to lack of global LUC datasets and challenges in identifying the land affected as a consequence of changes in products demand (De Rosa et al. 2015). Particularly disputed is the modelling of indirect Land-use Changes (iLUC): iLUC *“refers to shifts in land use induced by a change in the production level of an agricultural product elsewhere, often mediated by markets or driven by policies”* (Allwood et al. 2014). What seems to be lacking are therefore consistent global LULUC data, providing spatially explicit information on both total area for each land-use type and land-use changes between them (IPCC 2003). In recent years, there have been significant developments in measuring and monitoring LULUC data: remote sensing technics provide global land cover maps and datasets (Friedl et al. 2010, Hansen et al. 2013); advancements in collecting and reporting international and regional land data allows development of more consistent databases for comparisons over time and between countries (MacDicken 2015). Yet, neither remotely sensed data nor country-based land-use statistics can alone provide spatially-explicit information on LULUC by land-use types (Keenan et al. 2015). On one hand, using remote sensed land cover classifications is challenging to distinguish between managed and unmanaged land, e.g. distinguishing by canopy cover criteria between large scale primary forests and planted forests or managed tree plantations. Satellite imageries in fact, typically identify land according to characteristics such as canopy cover, vegetation height, soil type and temperature rather than land-uses. On the other hand, national statistics present limitation in terms of data quality and

consistency between countries (Hansen et al. 2013) and quantify land occupation per land-use type rather than providing explicit information on land-use changes.

In order to overcome these limitations, we combined the strengths of both state-of-the-art remote sensing global land cover datasets (Friedl et al. 2010), the Forest Resource Assessment dataset (MacDicken 2015) and FAOSTAT country-based land-use statistical database (FAOSTAT 2015). The aim was to obtain a LULUC database including a collection of country-specific globally consistent set of spatially-explicit LULUC matrixes, as suggested by the IPCC 'Good Practices Guidance for LULUCF' (IPCC 2003). The matrixes show LULUC from 2001 to 2012, 2005 to 2010, 2010 to 2012, and year by year from 2001 - 02 to 2011 - 12, 2). While examples of regional LUC matrixes exist (Luo et al. 2008, Sohl and Sayler 2008, Versace et al. 2008), the purpose of the LULUC database presented here was to identify globally and nationally the land undergoing a change from one land category to another between two reference years, from 2001 to 2012, based on global datasets. The paper also presents and discusses more in detail the results for countries where the most significant changes in forest and cropland area occurred from 2001 to 2012 as an example to guide through matrixes' interpretation and illustrate their potential applications.

2. Materials and Methods

We produced two versions of the LULUC matrix: the first version - V1 - was based on the MODIS Collection 5 Land Cover Type product (Friedl et al. 2010) and the International Geosphere-Biosphere Program (IGBP) land classification system. The area undergoing a transition between land classes is represented by grid cells geographically identified (spatially-explicit). The second version - V2 - combined the remotely sensed data with country-based land-use statistics. To do this we aggregated the IGBP forest classes based on land cover types in a generic forest class and then disaggregated it in forest-related land-use categories according to FAOSTAT land categories.

2.1 LULUC matrix version 1

The source of land cover information was the MODIS Collection 5 Land Cover Type product (Friedl et al. 2010) generated in a 461 m spatial resolution covering a temporal range between 2001 and 2012 and based on the IGBP DISCover Data Set Land Cover Classification System (Loveland and Belward 1997). To provide a LULUC-matrix for each country we used the TOOL tabulate area in ArcGIS.

TOOL calculates the cross-tabulated area between two datasets - in this case land cover data from 2001 and 2012 - and provides a table with area information within defined zones. For each country, land covers from 2012 defined the zones and land covers from 2001 defined the raster from which the areas were to be summarized within each zone. This provided a table with information on how much area remained within the same land cover class from 2001 to 2012, how much changed, and what it changed into. This procedure was repeated for each country (Global administrative areas <http://www.gadm.org>) – for a total of 231 country-specific LULUC matrixes. Analogously, matrixes were produced for 2005-2010, 2010 – 2012, and year by year from 2001 - 02 to 2011-12.

Furthermore, we aggregated the values to generate 6 continental matrixes for the same time frames. The 16 land cover classes generated 225 land cover change, that is the number of cells in matrix V1 excluding the diagonal values. In order to univocally identify each of these spatially explicit changes, the land cover in 2001 was multiplied with 100 and then summed with the land cover in 2012 (2001 MODIS land cover data \times 100) + 2012 MODIS land cover data). In this way it was possible to detect the 240 different land cover changes distinguished by a integer number code (e.g. 203 coding for land cover value = 2 in 2001 became land cover class 3 in 2012).

The limits of the IGBP land cover categories lies in not providing information on the actual land-use. The categories are in fact designed to univocally identifying land according to characteristics such as canopy cover, vegetation height, soil type and temperature but do not inform on whether the land is managed or unmanaged. Most importantly, none of the categories classifies planted forest or tree plantations. The classification is formally correct, since it classifies the land type in accordance to the definition of the land categories. Nevertheless, it does not provide sufficient information to support environmental impact analyses related to LUC.

2.2 LULUC matrix version 2

The second version - V2 - of the matrix was designed to represent the tree-covered land, i.e. evergreen/deciduous needleleaf/broadleaf forest, mixed forest and woody savanna, according to the FAOSTAT forest land categories and to include the FAOSTAT category *Permanent Crops* while

maintaining the remaining land categories as in IGBP classification (Fig. 1). Forest land cover types, from the remotely sensed data produced in V1, were first aggregated into an aggregated *Forest* category, a provisional working category, and then disaggregated as shown in Fig. 1 using FAOSTAT and Forest Resources Assessment (FRA) data (MacDicken 2015).

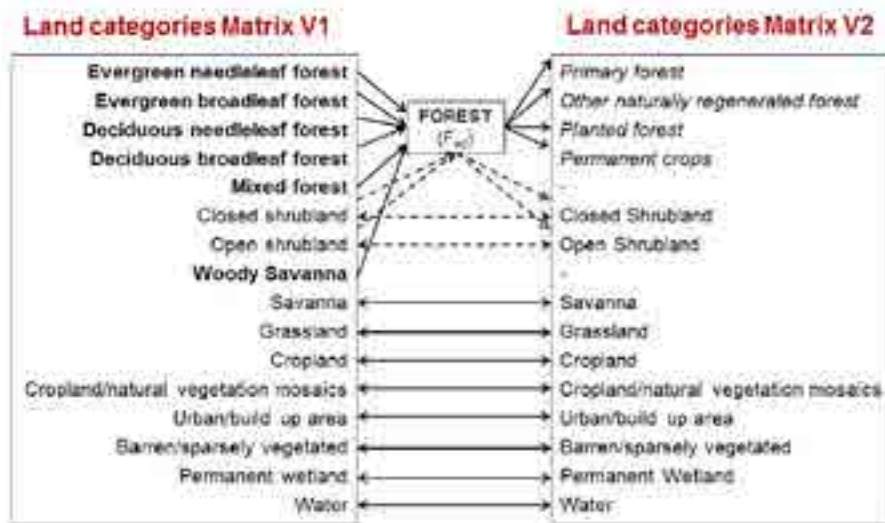


Fig. 1- Land categories in LULUC matrix version 1 (V1) and matrix version2 (V2). V1’s categories corresponded to the IGBP DISCover categories land categories. In both version of the matrix Land-use Changes could occur between any land categories. The arrows represent the relationships between the categories of matrix V1 and V2. Continuous arrows indicate a direct correspondence. Dashed line indicate that closed and open shrub land turning into any of the forest category above in matrix V1 was accounted in matrix V2 as being already a forest land type, for consistency with the FAOSTAT definition of forest. FAOSTAT (2016) in fact defines as a forest also land cover by trees not reaching yet the forest canopy cover or height threshold but “able to reach these thresholds in situ”.

Although the names of the forest land categories of matrix V2 echo FAOSTAT forest land categories, the forest area may be different because the FAOSTAT definition of forest land do not correspond to the IGBP definition (see Loveland and Belward 1997, and FAOSTAT 2016). The generic *FOREST* working category of Fig. 1 was disaggregated in matrix V2 by applying the ratio of the FAOSTAT forest subcategories *Primary forest*, *Other naturally regenerated forest* and *Planted forest* out of the total forested land reported in FAOSTAT. The IGBP classification system of forest in matrix V1 as land with canopy cover >60% and trees’ height >2mt was assumed to include permanent crops, e.g. fruit trees and plantations. Since IGBP land cover types do not make explicit the area of permanent crops, V2 included the FAOSTAT land category *Permanent crops* (Pc). Fig. 2 illustrates the structure of matrix version 2 as drawn from matrix version 1 and FAOSTAT data. The aggregated FOREST category F_{ag} in Fig. 1 is represented by the red dashed line in Fig. 2. The area of permanent crops in year y (Pc_y) was drawn by FAOSTAT and subtracted from the area identified as forest not undergoing changes by satellite imagery in Matrix V1.

The dashed lines in Fig. 2 indicate that non-forest land with woody vegetation (i.e. land classified as open and closed shrubland) not reaching yet the forest thresholds in year y but “able to reach these threshold in situ” (FAOSTAT 2016) was accounted in matrix V2 as forest land cover type, consistently with FAOSTAT definition. Consequently, cells from G3 to G7 and H3 to H7 in matrix V2 (Fig. 2) cannot assume any value, because they were already accounted as forest lands.

land classes.

Initial area \ Final area	Primary forest	Other naturally regener. forest	Planted forest	Permanent crops	Closed shrublands	Open shrublands	Savannas	Grasslands	Permanent wetlands	Croplands	Urban build-up areas	Cropland/nat. vegetat. mosaics	Barren or sparsely vegetated	Water	Final Area
	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]	[km ²]
Primary forest	11,085,982	1,386,089	-	-	-	-	-	-	-	-	-	-	-	-	12,472,072
Other naturally regener. forest	8,260	19,358,181	0	5,889	-	-	1,322,478	405,100	234,728	272,263	0	1,135,637	9,473	67,346	22,819,353
Planted forest	254	0	1,661,299	647	-	-	148,431	82,766	49,074	97,051	0	282,956	2,141	11,196	2,335,816
Permanent crops	104	0	0	1,179,046	-	-	94,221	37,283	27,933	61,972	0	161,009	816	2,106	1,564,492
Closed shrublands	12,553	21,504	2,283	1,452	54,098	58,175	23,631	31,655	2,286	10,785	0	9,566	710	902	229,600
Open shrublands	167,347	300,970	15,532	5,548	123,913	15,312,763	550,403	2,284,286	78,132	208,534	0	131,263	845,726	49,674	20,074,093
Savannas	420,556	857,227	27,832	36,898	114,626	462,990	6,342,051	474,149	20,501	171,946	0	655,283	1,492	6,140	9,591,690
Grasslands	177,637	314,185	25,032	17,503	79,151	1,798,447	545,493	13,015,117	33,902	1,067,628	0	542,418	936,574	250,356	18,803,442
Permanent wetlands	159,116	307,447	23,834	20,454	160,769	99,300	92,787	587,119	50,791	1,034	65,151	12,896	36,236	1,633,139	
Croplands	118,838	259,768	36,026	25,802	27,649	329,611	284,939	961,214	13,039	8,724,172	0	1,181,444	31,668	6,243	12,000,414
Urban build-up areas	0	0	0	0	0	0	0	0	543	0	652,075	0	0	43	652,660
Cropland/nat. vegetat. mosaics	354,083	778,066	79,624	62,871	23,487	94,776	757,100	556,613	14,952	1,242,996	0	4,773,456	46,169	3,653	8,787,847
Barren or sparsely vegetated	3,127	5,396	657	382	943	344,480	2,302	356,466	4,803	9,264	0	33,056	17,932,573	71,643	18,765,094
Water	48,498	75,522	6,259	3,210	2,889	42,787	14,272	130,526	34,746	7,118	76	7,263	180,479	4,360,737	4,914,384
Initial Area	12,556,356	23,664,356	1,878,379	1,355,453	447,210	18,604,800	10,184,621	18,427,961	1,101,758	11,924,522	653,185	8,978,501	20,000,718	4,866,276	134,644,096
NET change	-84,284	-845,002	457,437	209,039	-217,610	1,469,292	-592,931	375,482	531,381	75,892	-525	-190,655	-1,235,624	48,108	0

Between 2001 and 2012 the majority of the world forests were evergreen broadleaf forests and the most extensive land cover type on Earth was barren and sparsely vegetated land followed by open shrubland (Tab. 1). These were also the land cover types undergoing most changes globally from 2001 to 2012. Matrix V2 (Tab. 2) enables to distinguish natural forest from planted forest and permanent crops. Overall, the aggregate matrix (Tab. 2) shows:

- a substantial decrease of natural forest land, caused by forest degradation (transformation into shrubland, savanna and mosaics of natural vegetation and cropland), cropland and grassland expansion;
- a rather stable cropland area though with an increasing area of mosaics of cropland and natural vegetation which in turns changes into forested land and woody vegetation;
- a global increase of open shrubland, mainly at the expense of barren land and grassland;
- a global expansion of grassland in barren or sparsely vegetated land, cropland and forest land.

The matrixes aggregated per continent (available in the full version of the paper) showed that the decrease of Primary forest occurred especially in Africa and South America in contrast to a slight increase in Europe. A more stationary forest trend is visible in North America. Vice versa, cropland and grassland have been expanding markedly in Africa and Asia while Europe, North America and Oceania experienced a significant decrease in cropland.

The database was designed to support spatially-explicit analyses of national, regional and global LULUC and advanced LUC analyses. The novelty appeared to be the outstanding possibility to visualize land-uses and land-use transitions for a specific country in a specific timeframe, distinguishing between managed and unmanaged land as primary forest from secondary forest, planted forests and permanent crops. The matrixes also allow to identify and quantify deforestation and land-use type following land clearing activities; areas where cropland is expanding and where is decreasing; identifying land not in use or unproductive arable land; the location and therefore the potential productivity of land in transition etc. There was a substantial difference between the two versions of the matrixes since they provided complementary but not comparable results and therefore required to be analyzed in pair. Matrix V1 showed for example that in Asia between 2001 and 2012 the most diffuse forest land class was evergreen broadleaf forest which increased of almost 170.000 km²; matrix V2 however, showed that most of the forest land type identified by matrix V1 was non-primary forest or planted forest and broadleaf crops, e.g. palm oil plantations: in eleven years the amount of Asian natural forest decrease of around 115.000 km² while planted forests and permanent crops increased approximately 430.000 km².

The database could support a number of applications: it facilitate a more holistic assessment of land-use changes, explicitly quantifying changes occurring between land classes – 240 LUC for matrix V1 and 183 for matrix V2. It could support global scale land-use change analyses, requiring a distinction between land types based not only on land cover but also on land-uses. The matrixes provided a good overview of where and what type of LUC occurred. This information is key when calculating LULUC

emissions for e.g. national and global emission accounts. Furthermore, the information contained in the LUC matrixes also provided an important input for economy-wide models that focus on iLUC. Economy-wide models can include LCA, input-output (IO) models and general and partial economic equilibrium models. The spatially explicit dataset constituted by matrixes V1 may serve as a starting point for further studies aiming at determining the drivers of land-use change supported by spatial statistical modelling. The drivers of land abandonment for example, could be stepwise assessed globally, regionally, and locally. This could shed light on potential dependencies between geographical characteristics, LUC drivers and their intensity (Odgaard et al. 2014). The single-year matrixes could also allow assessing not only the location but also the rate of change. The rate can be calculated by detecting the change over a time period for each cell (Sandel and Svenning 2013). The database was designed to be modified and include MODIS Collection 6 product when available and FAOSTAT updates. The accuracy of matrixes V2 was affected by the lack of FAOSTAT data (e.g. data on primary forest or permanent crops); analogously FAOSTAT proxy data based on manual estimation with high uncertainties implied a higher uncertainty also of the data reported in matrix V2.

4. Conclusion

This paper proposed a first attempt to globally generate country-based top-down LULUC matrixes. The matrixes identified the area undergoing a transition between 2001 and 2012 and intended to support spatially-explicit analyses of national, regional and global LULUC and advanced LUC analyses. The novelty appeared to be the outstanding possibility to visualize land uses and land-use transitions for a specific country. The two versions of the matrix also allowed to identify weaknesses in FAOSTAT national statistics and to interpret MODIS satellite images. Matrix version 2 distinguished between managed and unmanaged land, as primary forest from secondary forest, planted forests and permanent crops, to identify and quantify deforestation and land-use type following land clearing activities; areas where cropland is expanding or decreasing; identifying land not in use or unproductive arable land; the location and therefore the potential productivity of land in transition etc.

Globally, the results show: a substantial decrease of natural forest land, caused by forest degradation (transformation into shrubland, savanna and mosaics of natural vegetation and cropland) cropland and grassland expansion; a rather stable cropland area though with an increasing area of mosaics of cropland and natural vegetation which in turns changes into forested land and woody vegetation; a global increase of open shrubland, mainly at the expense of barren land and grassland; a global expansion of grassland in barren or sparsely vegetated land, cropland and forest land. Between 2001 and 2012 the decrease of natural forest land took place especially in Africa and South America, with a minor decrease in Oceania and Asia and a slight increase in Europe. A more stationary forest trend is visible in North America. Vice versa, cropland and grassland have been expanding markedly in Africa and Asia while Europe, North America and Oceania experienced a significant decrease in cropland. The LULUC database could serve as a starting point for further global and country-specific detailed LULUC analyses: it intended to support more holistic assessments of land-use accounting for differences in land-use type and their location. The matrixes provided an overview of where and what type of LUC occurred, a crucial information to calculate LULUC-related environmental impacts as emissions accounts. It also provided an important input for economy-wide models – e.g. LCA, input-output models - that focus on iLUC impacts. The database could be modified to include updates as MODIS Collection 6 product and periodic FAOSTAT updates.

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ORAL PAPERS

FOOD PROCESSING

14. Food Processing

128. Reducing Environmental Impact of Greek Yogurt is the Whey to Go

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ABSTRACT

In the United States, traditional Greek yogurt production and consumption has higher environmental impact than set or stirred-types including human toxicity (10%), climate change impact (40%), and water depletion (up to 65%) due to a higher quantity of raw milk input per functional unit caused by straining. The straining process results in a thicker yogurt, which has become consumers' favorite, but it also produces liquid acid whey. Potential environmental impacts of the traditional Greek yogurt production and acid whey were not yet evaluated. Due to unavailability of surveyed processing data for Greek yogurt production and different treatment options, we used SuperPro Designer[®] models developed by the United States Department of Agriculture (USDA) to simulate whole milk Greek yogurt processing and anaerobic digestion of acid whey. Acid whey is mostly water, thus we developed additional filtration options including reverse osmosis, microfiltration, and ultrafiltration. The output data from the SuperPro Designer[®] models were used as input data to SimaPro for a cradle-to-yogurt processing life cycle assessment (LCA). The results showed that anaerobic digestion has the lowest environmental impact. Under membrane scenarios, production of value added products including fat and casein, whey concentrate, and lactose increased electricity and natural gas use, but compared to acid whey transport, it reduced both the size and number of trucks by half, which reduced cost of transportation, but also avoided cost from farmers to use it as fertilizer or animal feed.

Keywords: acid whey, anaerobic digestion, whey-to-Greek acid whey concentrate, whey-to-lactose, reverse osmosis, ultrafiltration.

1. Introduction

Greek yogurt success among U.S. consumers was rapid with 50% increase of market share since its introduction in 2007. Thick and creamy consistency of traditional Greek yogurt is achieved by a straining process, which creates acid whey. Processing and disposal of acid whey adds to overall yogurt manufacturing costs and has larger environmental impacts compared to set or stirred-style yogurts (Thoma et al., 2016). In the United States, raw acid whey is transported to farms and used as animal feed or fertilizer. But, land application of acid whey and use as animal feed are not considered good long term solutions because of their low profitability, for example, the producer covers the acid whey transport cost and pays the farmer to use acid whey. Existing dairy wastewater treatment (WWT) plants cannot treat acid whey because of high biological oxygen demand, chemical oxygen demand, lower pH, and high phosphorous level compared to typical dairy effluent or cheese whey (Table 1).

Table 1: Typical fluid milk manufacturing plant wastewater composition compared to cheese and acid whey composition (Chatzipaschali and Stamatis, 2012; Rao, 2008)

Composition	Fluid milk plant wastewater	Cheese whey	Acid whey
pH	7.2	6.4-6.3	4.6-4.8
Total solids	1,060 mg/L	63,000-70,000 mg/L	63,000-70,000 mg/L
BOD	1,240 mg/L	35,000 – 55,000 mg/L	45,906 mg/L
COD	84 mg/L	50,000-70,000 mg/L	61,138 mg/L
Phosphorus	11.7 mg/L	319-957 mg/L	2,000–4,500 mg/L

Despite environmental concerns and popularity of Greek yogurt in the United States, the availability of yogurt processing data is limited to set and stirred curd yogurt. Milk production is driving environmental impacts of whole milk Greek yogurt and transport of raw acid whey is costly; thus, it does not provide an optimal solution for the acid whey problem (Thoma et al., 2016). We built upon Thoma et. al., (2016) work to include production of high value consumer acid whey products, i.e., whey concentrates and lactose. Raw acid whey has higher phosphorous (Table 1) and two times

higher calcium content than cheese whey; thus, dried Greek acid whey concentrate (GAWC) possibly can be added to animal feed and if approved as such, for food products as bulking, sweetening, and as nutrient fortifying agent. Lactose is typically used as animal feed.

2. Methods

We used SuperPro Designer[®] Greek yogurt processing plant models provided by the USDA to prepare life cycle inventory (LCI) (Intelligen Inc., 2015; Tomasula and Yee, 2016). The models included a whole milk Greek yogurt production (1) without acid whey treatment, named whey-to-feed scenario, (2) whey-to-biogas scenario using anaerobic digestion, and (3) no-whey scenario of non-traditional Greek yogurt production (Tomasula and Yee, 2016). Survey data for stirred and set yogurt were used to verify Greek yogurt models (Thoma et al., 2016). We expanded SuperPro Designer[®] models to include (4) whey-to-Greek acid whey concentrate (GAWC) using reverse osmosis (RO) and (5) whey-to-GAWC and lactose scenarios using RO and ultrafiltration (UF). When whole milk is used to produce Greek yogurt, the first step in membrane treatment of acid whey is microfiltration (MF), which retains fat and casein. These results were not individually reported because of lower contribution in total plant impacts.

Cradle-to-processing plant and gate-to-processing plant LCAs were used to evaluate Greek yogurt and acid whey LCA scenarios, respectively. Environmental impacts were calculated using IPPC 2007 100a GWP characterization factors for climate change, Cumulative Energy Demand (CED) for non-renewable fossil energy indicator, USEtox for freshwater ecotoxicity and human toxicity, and ReCiPe 2008 for water depletion, marine eutrophication (MEU), freshwater eutrophication (FEU), and photochemical ozone formation (POF) (Goedkoop et al., 2009; Henderson et al., 2011; Huijbregts et al., 2006; Solomon et al., 2007).

SuperPro Designer[®] models provided process specific data for plant energy, material, water use, and emissions; thus, no allocation was needed for the gate-to-gate LCA. The LCI for scenarios (1), (2), and (3) was reported in Thoma et al., (2016), and for scenarios (4) and (5) in Table 2. Raw milk impact was allocated to Greek yogurt, but not to the GAWCs, and lactose, which enabled assessment of the increase in environmental impact of the whole yogurt processing plant.

(1) Whey-to-farm scenario:

The system boundary included a cradle-to-gate LCA of whole milk Greek yogurt production. Raw acid whey was shipped 340 km to a farm (Economic Census, 2012; Thoma et al., 2016; Tomasula and Yee, 2016).

(2) Whey-to-biogas scenario:

The system boundary included a cradle-to-gate LCA of whole milk Greek yogurt production. The yogurt plant used an anaerobic digester to treat acid whey and took credit for produced biogas (Thoma et al., 2016; Tomasula and Yee, 2016).

(3) No-whey scenario:

The system boundary included a cradle-to-gate LCA of non-traditional Greek yogurt production. Non-traditional Greek yogurt is produced by adding milk by-product powders such as milk protein concentrates and stabilizers and no acid whey is removed (Thoma et al., 2016; Tomasula and Yee, 2016). However, traditional method is superior in product consistency and taste, so many processors have already switched to traditional production.

(4) Whey-to-GAWC scenario:

The system boundary is shown in Figure 1. This scenario included microfiltration (MF), which reduced bacteria and spores resulting in lower heat treatment requirements and removed fat and casein. GAWC is retained by reverse osmosis (RO) and water is removed by spray drying (Figure 1). The GAWC is spray dried and shipped on average 1,196 km to animal feed processor (Economic Census, 2012).

(5) Whey-to-GAWC and lactose scenario:

Following microfiltration (MF), the permeate was ultrafiltered (UF) to retain the GAWC and spray dried (Figure 1). The UF permeate is rich in lactose, which is retained by the RO and spray dried.

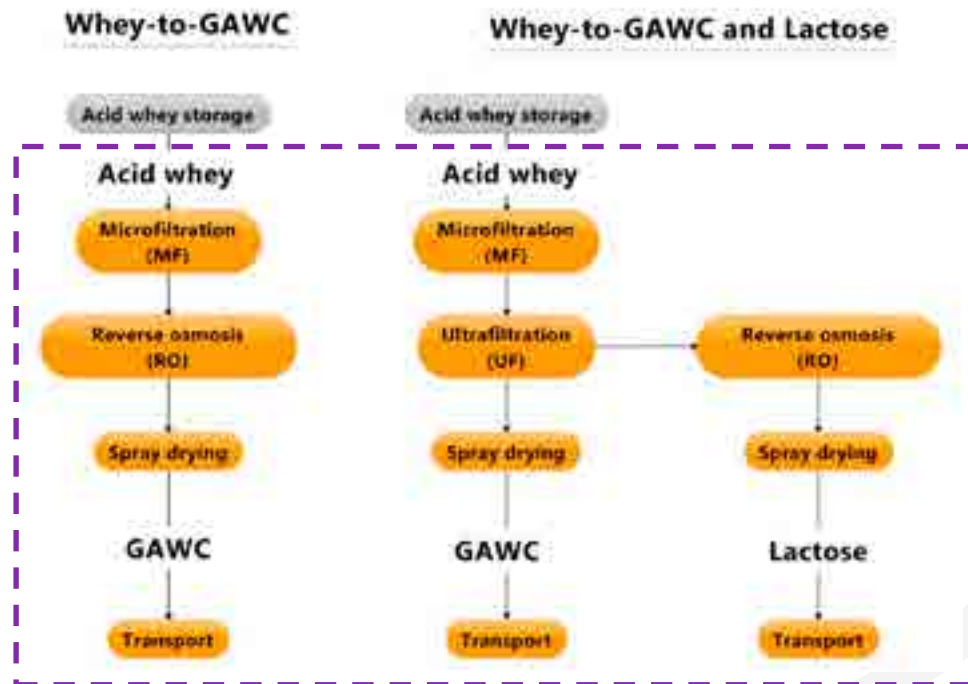


Figure 1: Membrane acid whey treatment scenarios including (4) whey-to-GAWC (RO) and (5) whey-to-GAWC (UF) and lactose (RO). Purple dash line shows system boundary.

Table 2: The LCI for on-site acid whey treatments (per kg of GAWC and lactose)

Greek yogurt plant	Unit process	GAWC (RO)	GAWC (UF)	Lactose (RO)
GAWC, dried (kg)	new	1	1	-
Lactose, dried (kg)	new	-	-	1
Electricity (kWh)	DataSmart	0.393	0.771	0.423
Natural gas (m ³)	DataSmart	1.12	2.05	0.396
Membrane material (kg)	DataSmart	3.88E-04	2.92E-04	5.01E-04
Water CIP (m ³)	DataSmart	1.37	5.23	4.07
Sodium hydroxide (kg)	DataSmart	0.003	0.020	0.007
Chlorine (kg)	DataSmart	0.004	0.010	0.009
Nitric acid (kg)	DataSmart	0.005	0.019	0.013
WWT (L)	DataSmart	6.37	5.23	19.8
Membrane disposal (kg)	DataSmart	3.88E-04	2.92E-04	5.01E-04
Transport of dried product (tkm)	DataSmart	1.20	0.304	0.278

RO – reverse osmosis, UF – ultrafiltration, GAWC – Greek acid whey concentrate, DataSmart – LCI database (EarthShift, 2014)

4. Results

Non-traditional production of Greek yogurt has the lowest environmental impact. Reduction of environmental impact of traditional Greek yogurt was achieved using anaerobic digester (Figure 2), but it will cost to implement digester (Thoma et al., 2016). Climate change impact of dried GAWC (RO), dried GAWC (UF), and dried lactose (UF) was estimated to be 3.36, 5.83, and 1.57 kg CO₂e/kg product, respectively. Yogurt plant natural gas and electricity consumption associated with the addition of membrane treatment increased by 70% and 17%, respectively, but the cradle-to-grave LCA impact increased up to 7% (Figure 2). Figure 3a, 3b, and 3c show contribution of different processes to environmental impacts of dried GAWC (RO), GAWC (UF), and lactose (UF), respectively. Natural gas used for spray drying was primary impact driver including climate change, non-renewable energy, ecotoxicity, and human toxicity (Figure 3a, 3b. and 3c). Thin-film composite material used for RO contributed to climate change impact, freshwater water depletion, and FEU (Koch, 2016). Water used for clean-in-place (CIP) had the largest contribution to water depletion. WWT treatment of permeate was primary impact driver of MEU and FEU. Clean-in-place chemicals used in scenario (5) whey-to-GAWC and lactose had higher contribution to environmental impacts due to additional system requirements (Figure 3b and 3c). Dried product transport impact was up to 12% for RO systems. But, transport of acid whey required 0.213 tkm per kg yogurt due to large output volume. Thus, production of value added products including GAWC and lactose reduced number and size of trucks necessary to transport products by half despite larger distances (0.003-0.031 tkm per kg yogurt).

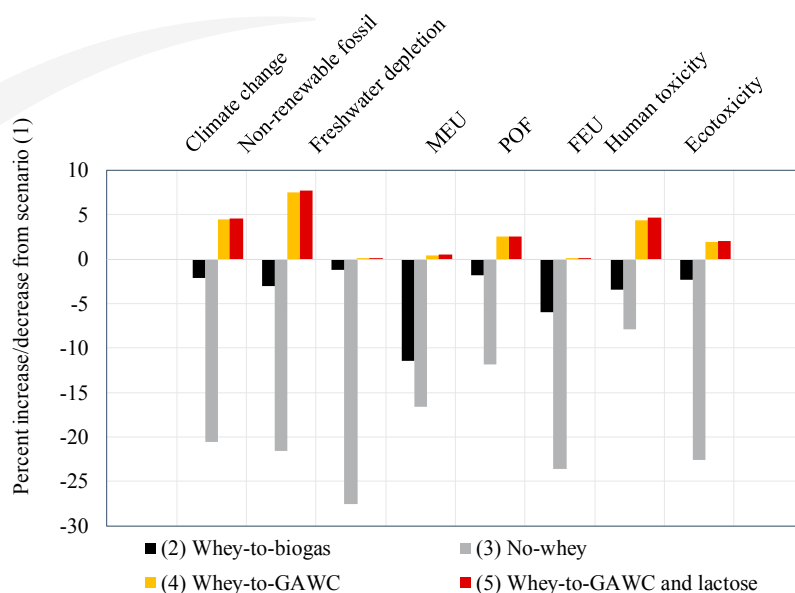


Figure 2: Percent environmental impact increase/decrease of cradle-to-gate Greek yogurt scenarios with acid whey treatment compared to the environmental impact of cradle-to-gate Greek yogurt (1) whey-to-feed scenario (zero green line) including climate change impact, non-renewable energy, freshwater depletion, marine eutrophication (MEU), freshwater eutrophication (FEU), photochemical oxidant formation (POF), human toxicity, and freshwater ecotoxicity.

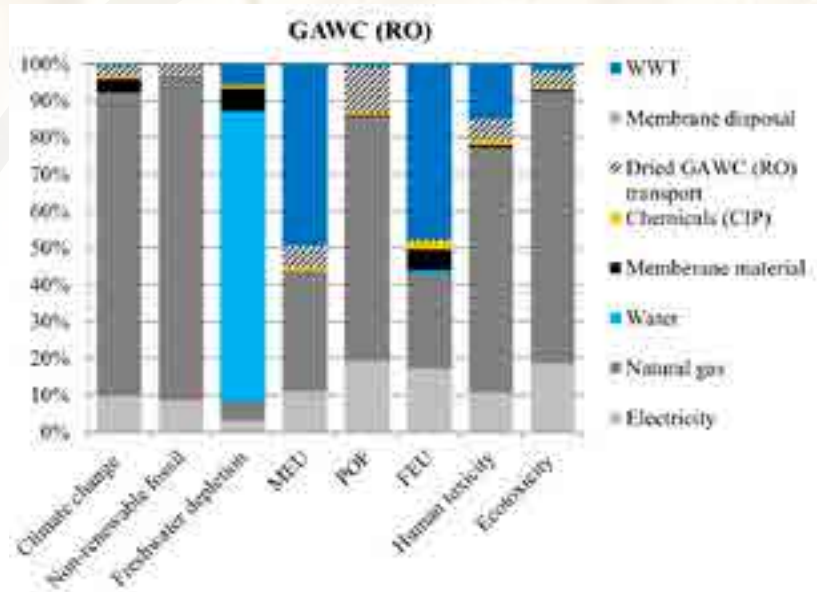


Figure 3a: Relative LCA results for GAWC (RO) including climate change impact, non-renewable energy, freshwater depletion, marine eutrophication (MEU), freshwater eutrophication (FEU), photochemical oxidant formation (POF), human toxicity, and freshwater ecotoxicity.

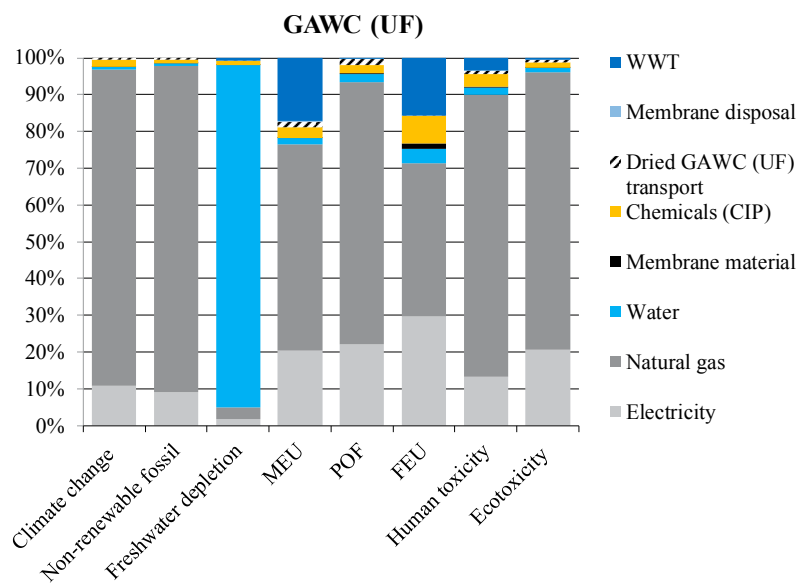


Figure 3b: Relative LCA results for GAWC (UF) including climate change impact, non-renewable energy, freshwater depletion, marine eutrophication (MEU), freshwater eutrophication (FEU), photochemical oxidant formation (POF), human toxicity, and freshwater ecotoxicity.

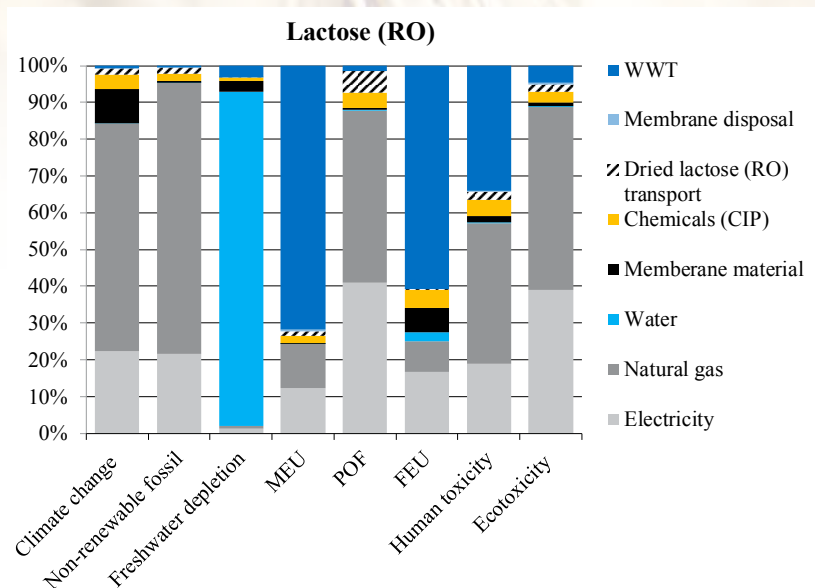


Figure 3c: Relative LCA results for lactose (UF) including climate change impact, non-renewable energy, freshwater depletion, marine eutrophication (MEU), freshwater eutrophication (FEU), photochemical oxidant formation (POF), human toxicity, and freshwater ecotoxicity.

5. Discussion

Even though non-traditional Greek yogurt production had the lowest environmental impact, most producers prefer traditional production to assure product taste and quality. EPA mandates that BOD is reduced to a certain level assessed each watershed, for example, New York State Island BOD surface discharge limits are less than 10 mg/L (Chobani, 2014). According to one Greek yogurt producer, the installed reverse osmosis membrane is capable to reduce COD and BOD values below discharge limits of adjacent river (Chobani, 2014). But, scenario (4) and (5) reduced BOD of the permeate to 1,000 mg/L and 50 mg/L, respectively. These values were above the discharge limit, but in the range of typical dairy effluents that can be treated in the dairy WWT (Table 2).

5. Conclusions

Anaerobic digestion of acid whey has potential to reduce environmental impacts of traditional Greek yogurt production. Dry GAWC and lactose production increase total impact of the yogurt processing plant, but reduce cost of raw acid whey transport and can be used in food and feed preparations.

Our next step is to optimize current models to ensure that permeate is below the discharge limits. Further research will focus on developing other options of acid whey treatment including WWT, enzyme treatment, and nanofiltration, but also uncertainty based comparison of different acid whey treatment options, and building the consequential LCA model.

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20. Pasteurized versus UHT Milk: Trade-offs between Environmental Impacts and Nutritional Quality

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ABSTRACT

Milk needs to undergo a sanitation treatment to destroy pathogenic and part of the spoilage microorganisms to make it fit for human consumption. To achieve this, milk can either be pasteurized or treated at ultra-high temperatures (UHT). Although these treatments help with the safety and shelf-life of milk, they also affect its nutritional quality and cause various environmental impacts. Therefore, this study aims to compare pasteurized and UHT milk from both environmental and nutritional perspectives. The results show that UHT milk is environmentally more sustainable than the pasteurized, but has lower nutritional quality due to a greater loss of vitamins. The choice between the two types of milk would, therefore, depend on stakeholder priorities and preferences.

Keywords: life cycle assessment, milk, nutritional quality, environmental sustainability.

1. Introduction

Although nutritional aspects are essential to understanding impacts of food on human health, they are not generally considered in life cycle assessment (LCA) of food products (Ernststoff et al., 2014). As a consequence, evaluation of nutritional and environmental aspects of food is normally carried out separately, which in turn may lead to misleading results and unfair comparisons between different food items. To address this issue, this study illustrates by an example of milk how nutritional parameters could be considered alongside LCA impacts.

To make it fit for human consumption, milk undergoes a sanitation treatment, which destroys pathogenic and part of the spoilage microorganisms. To achieve this, milk can either be pasteurized or treated at ultra-high temperatures (UHT). Pasteurized milk undergoes a milder heat treatment compared to UHT and needs to be kept under temperature-controlled conditions across its life cycle, while UHT milk can be stored at ambient temperature. Although the treatment helps with the safety and shelf-life of milk, it also affects its nutritional quality while at the same time increasing the impacts of milk production.

2. Goal and Scope

The goal of this study is to compare pasteurized and UHT milk from both environmental and nutritional perspectives. The environmental assessment has been carried out using LCA and the nutritional quality has been evaluated in terms of vitamin loss resulting from the heat treatment at the processing stage (based on Corradini, 1995). The reason for considering vitamins rather than the other milk constituents is that they are sensitive to heat (Corradini, 1995) while fat, carbohydrates, proteins and minerals are essentially unaffected by heat treatment (Rolls and Porter, 1973). The scope of the study is from 'farm gate to grave' (Figure 1); the production of milk at farm, common to both milk products, is excluded. The functional unit is defined as '1 litre of milk consumed at home'.



Figure 1: System boundaries for pasteurized and UHT milk from ‘farm gate to grave’

3. Life Cycle Inventory

Data have been sourced from literature and adapted to UK conditions. This section specifies the assumptions and data sources used in the different life cycle stages.

For the processing stage, the following inputs and outputs have been taken into consideration:

- electricity and heat (natural gas) consumption for milk treatment (based on FAO, 1992; Nielsen et al., 2003);
- electricity consumption during pasteurized milk storage at plant (Brush et al., 2011);
- refrigerant use and leakage during pasteurized milk storage at plant (DEFRA, 2008);
- water consumption (WRAP, 2013);
- sanitizers used for cleaning of production appliances (NaOH and HNO₃) (Thomas and Sathian, 2014);
- wastewater treatment (EC, 2006); and
- packaging material: HDPE for pasteurized and poly laminate for UHT milk (based on DairyCo, 2014).

A distance of 100 km has been assumed from the milk-processing plant to the retailer. The life cycle inventory data for transport have been sourced from Ecoinvent (Frischknecht et al., 2007). For pasteurized milk, they have been modified to include the additional amount of fuel used by the on-board refrigeration unit as well as the production and leakage of refrigerants, with the latter assumed at 22.5% of the annual charge (DEFRA, 2008; UNEP, 2003).

The electrical and thermal energy consumed at the retailer for heating, lighting and ventilation has been calculated in accordance with DEFRA (2008). The same data source has also been used for electricity and refrigerant consumption and leakage for storing pasteurized milk in supermarkets.

The consumer transport distance and the related impacts associated with milk shopping have been calculated assuming UK conditions, based on:

- the composition of the UK weekly food basket by weight (DEFRA, 2014);
- the average distance covered in the UK per week for food shopping (Pretty et al., 2005); and
- the share of distance travelled by car and by bus for food shopping (Pretty et al., 2005).

The data for the consumption stage have been calculated based on:

- the volume of domestic refrigerators/freezers and their average daily energy consumption (Zimmermann et al., 2012); and
- the average food storage time (WRAP, 2010).

For waste management, the amount of waste and its disposal have been assumed based on the UK statistics for food waste (WRAP, 2009). As shown in Figure 2, two sources of milk spoilage are considered at the consumer level: overcooked milk and milk not used in time. Since UHT milk has a much longer shelf-life compared to the pasteurized (6 months vs. 4 days), it is assumed that 50% less waste is generated from UHT milk related to the milk not used before the expiry date. This is a conservative assumption as the amount of waste could be lower due to the much longer shelf-life of UHT milk.

The LCA impacts have been estimated using GaBi V6.11 (Thinkstep, 2015), following the CML method (Guinée et al., 2002); in addition, primary energy demand has been also estimated using the method in GaBi.

For the nutritional quality, the following vitamins are considered:

- vitamin B1, B6, B9 and B12; and
- vitamin C.

These vitamins are considered due to their nutritional relevance, particularly of vitamin B, for which milk is a good source (Miller et al., 2000).

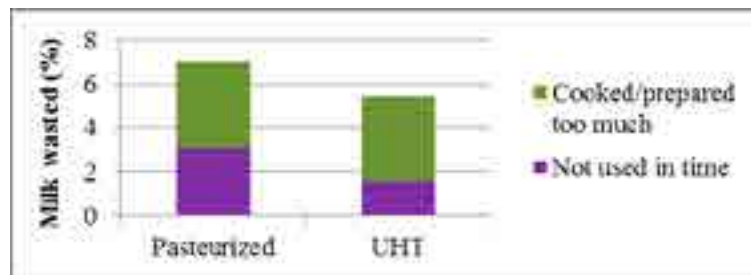


Figure 2. Percentage of milk wasted at the consumer level.

4. Results and Discussion

Figure 3 shows the environmental impacts of 1 litre of pasteurized and UHT milk from farm gate to grave. As can be seen in the figure, UHT milk is environmentally more sustainable than the pasteurized, with the difference in impacts ranging from 15% for depletion of elements to 92% for ozone layer depletion (ODP). Global warming potential is around 20% higher for pasteurized milk. The high difference for ODP between the two types of milk is due to the use of refrigerants across the life cycle of pasteurized milk, while no refrigerants are required in the case of UHT. The gap for the other impact categories is mainly due to the different energy requirements for the treatment of two types of milk and, in the case of ADP fossil, also due to the different packaging materials.

However, when considering the nutritional quality, pasteurized milk appears to be a better option because of lower vitamin losses. The greatest difference is found for vitamins B6 and B1: 0.4% loss vs 10% and 1% loss vs 10%, respectively (Figure 4). This is due to the milder heat treatment used for pasteurized milk (72°C) than for the UHT (135°C). The difference in the loss of vitamins B9, B12 and C between the two treatments is smaller but still noticeable.

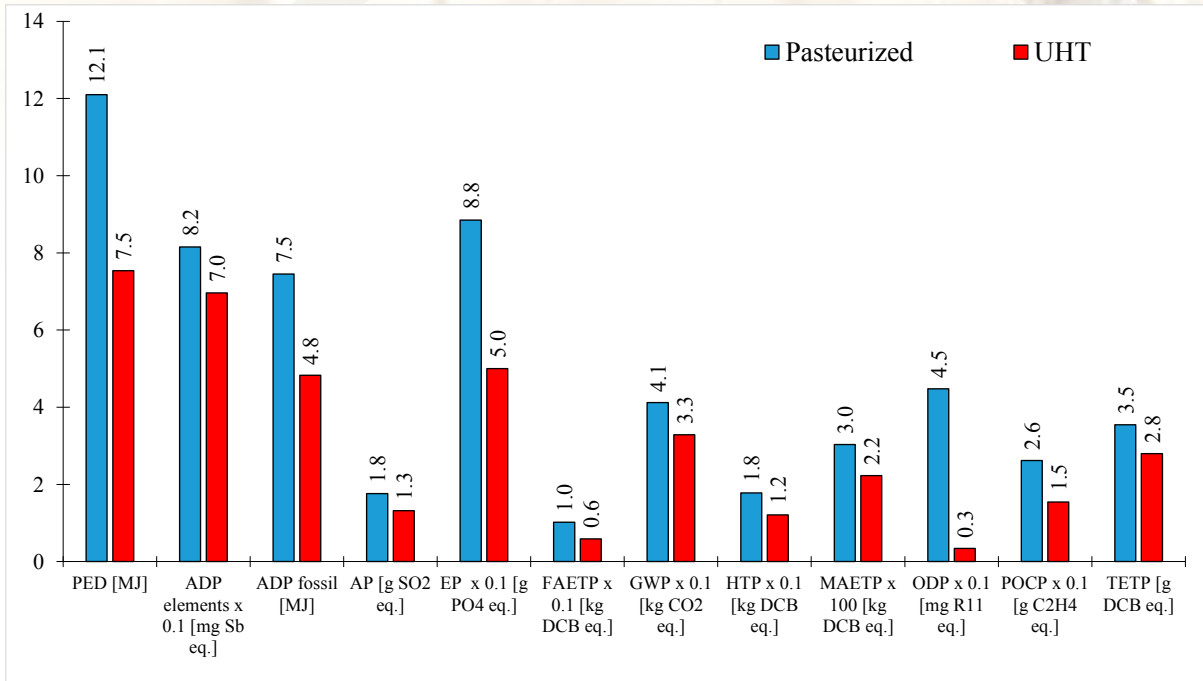


Figure 3. Environmental impacts of 1 litre of pasteurized and UHT milk from farm gate to grave. [PED: Primary energy demand; ADP: Abiotic depletion potential; AP: Acidification potential; EP: Eutrophication potential; FAETP: Freshwater aquatic eco-toxicity potential; GWP: Global warming potential; HTP: Human toxicity potential; MAETP: Marine aquatic eco-toxicity potential; ODP: Ozone layer depletion potential; POCP: Photochemical oxidants creation potential; TETP: Terrestrial eco-toxicity potential].

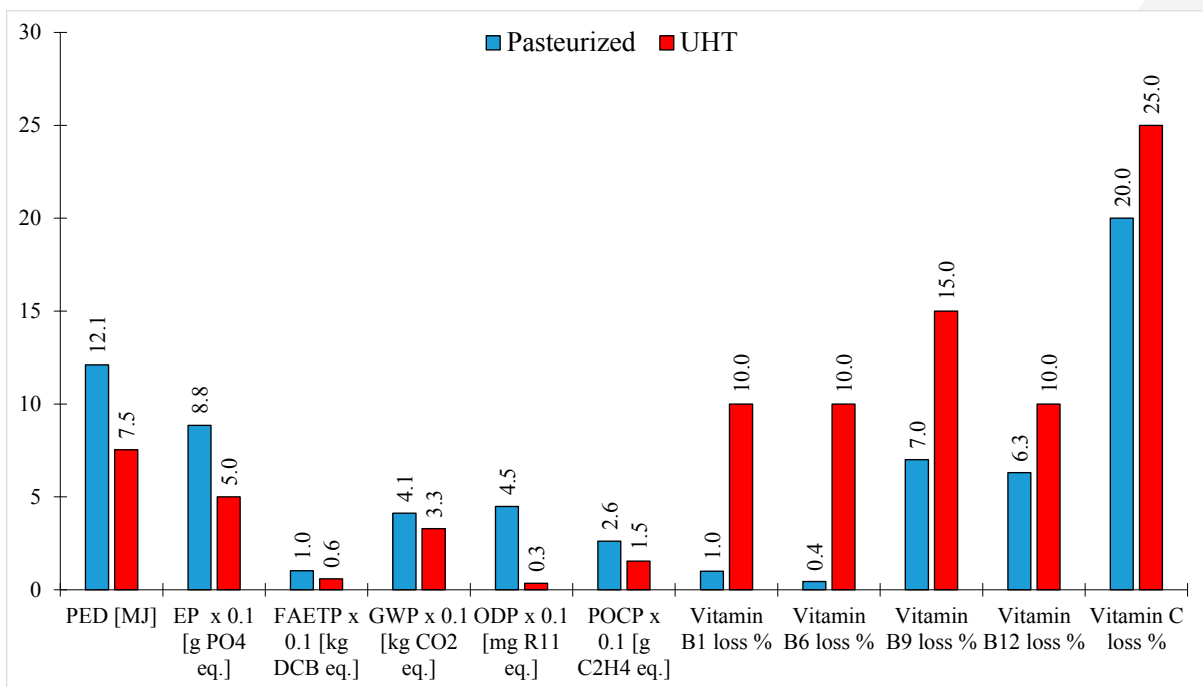


Figure 4. The environmental and nutritional performance of 1 l of pasteurized and UHT milk. [Environmental impacts are from farm gate to grave. Select environmental impacts shown, focusing on those with the greatest difference between the two types of milk. For impacts nomenclature see Figure 3.]

5. Conclusions

This study has demonstrated how nutritional parameters can be considered alongside life cycle environmental impacts using milk as an example. The results show that from the environmental point of view, the energy intensity of the UHT treatment is lower than the consumption of electricity and refrigerants in the cold chain of pasteurized milk. In addition, the prolonged shelf-life of UHT leads to a lower generation of waste. However, the intensive heat treatment of UHT causes a greater loss of vitamins, which from a consumer perspective suggests a lower-quality product. The best option between the two types of milk would, therefore, depend on stakeholder preferences and priorities. These could be explored in future studies through multi-criteria decision analysis. Future studies could also consider sensorial properties as another important parameter for assessing quality and consumer acceptability of food.

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107. Life cycle greenhouse gas emissions of ready-made baby food

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ABSTRACT

The ready-made baby food market is growing fast but little is known about its environmental impacts. In an attempt to provide further information and stimulate the debate about the sustainability of baby food, this study considers global warming potential (GWP) of one of the most-consumed types of baby food in the UK: oat porridge. Life cycle assessment has been used to estimate GWP. The functional unit is defined as '125 g of oat porridge consumed at home', equivalent to one meal. The scope of the study is from 'cradle to grave'. The results suggest that the GWP is equivalent to 155 g CO₂ eq. per functional unit. The main hotspots are the raw materials (70%) and manufacturing (15%). For the former, the most relevant ingredient is milk powder and for the latter, drying of the product. Considering different product formulations, the GWP could be reduced by up to 23%. These results could be used by baby food manufacturers to mitigate climate change impacts from their products.

Keywords: Baby food; global warming potential; life cycle assessment; oat porridge.

1. Introduction

Diet is very important for infants since what they eat in their early developmental stages determines to a degree their health later in life. However, baby diet has been changing over the past decades, moving away from traditional, home-made food to commercially produced meals. For example, a survey of infants, aged 6 to 12 months, found that 81% consumed ready-made baby food (Nestle Nutrition Institute 2008). The reason for this shift is largely related to modern lifestyles, with a lot of parents preferring convenience to traditional methods of cooking.

As a result, the ready-to-eat baby food market is growing fast globally. For example, in the UK it grew by around 30% since 2009 and it was worth an estimated £656 million in 2014 (Intel 2015). However, despite its socio-economic importance, studies of environmental impacts in the baby food sector are rare, with only one study found in the literature from the late 1990s, based in Sweden (Mattsson 1999). Therefore, this study aims to fill this knowledge gap by evaluating the implications for climate change of this growing market. As an illustration, the focus here is on one of the most-widely consumed products by babies in the UK – oat porridge.

2. Methodology

The goal of the study is to estimate the global warming potential (GWP) of oat porridge and to identify the hotspots across the supply chain. Life cycle assessment has been used as a tool and the system been modelled in GaBi software (Thinkstep 2015). GWP has been estimated following the CML impact assessment method (Guinée et al. 2002). The scope of the study is from 'cradle to grave' and the functional unit is defined as '125 g of oat porridge consumed at home', equivalent to one baby meal.

The system boundary encompasses the production and processing of raw materials (ingredients), the manufacturing of the ready-made baby food, the production of packaging materials, product distribution, retail, consumption and end-of-life (EoL) waste management. The consumption stage involves boiling water to prepare the porridge using one part of cereals and three parts of water, as recommended by manufacturers. EoL waste management

includes consumer waste and packaging; wastes generated in other life cycle stages are accounted for in the respective stages.

The product formulation has been defined following own market research for different brands of cereal-based baby foods. The average product has then been defined based on that information. As shown in Table 1, it contains oat flakes, rice flour, milk powder, sugar, palm oil and barley malt extract. The manufacturing stage has been modelled using data from the literature (Gantwerker & Leong 1984). Data for this and the other life cycle stages can be found in Table 2. The ready-made product is packaged in a 125 g bag in box with the plastic inner bag made of low density polyethylene film (5 g) and the outer packaging of cardboard box (25 g). For EoL, current UK waste management practices have been applied (DEFRA 2015). Composting has been assumed for waste ingredients or porridge generated in manufacturing due to spillage or poor quality.

For transport, a distance of 50 km has been assumed in all life cycle stages where relevant, except for consumer shopping, for which a round trip of 7 km has been assumed.

Table 1: Product formulation and inventory data for the raw materials (ingredients)

Ingredients	Contribution (%)	Description	Country of origin	Source of LCI ^a data
Oat flakes	35	Oats cultivation	NL	Van Zeist et al. (2012) [Barley seed used as proxy]
		Field operations, grain drying	UK	McDevitt & Canals (2011); Nemecek & Kagi (2007)
		Oat flakes milling	UK	Nielsen et al. (2003)
Rice flour	11	Rice cultivation	US	Ecoinvent Centre (2015)
		Rice flour milling	UK	Nielsen et al. (2003) [Wheat milling used as proxy]
Milk powder	30	Dairy farming	UK	Williams et al. (2006)
		Milk powder production	UK	Nielsen et al. (2003)
Sugar	20	From sugarcane, at sugar refinery	BR	Ecoinvent Centre (2015)
Palm oil	3	At oil mill	MY	Ecoinvent Centre (2015)
Malt extract	1	Barley grain production	DE	Ecoinvent Centre (2015)
		Barley malt extract powder production	UK	Own calculations

^a LCI: life cycle inventory

Table 2: Data for energy, water and waste in different life cycle stages (expressed per functional unit)

Parameter	Life cycle stages					
	Ingredients	Manufacturing	Packaging	Retail ^a	Consumption ^b	End of life
Electricity (MJ)	2.63	1.01	0.31	1.36x10 ⁻²	1.44x10 ⁻¹	2.45x10 ⁻³
Natural gas (MJ)	0.196	0.32	0.10	1x10 ⁻³	3.3x10 ⁻²	4x10 ⁻⁴
Water (l)	10.8	0.37	0.13	1x10 ⁻³	1.42x10 ⁻¹	9x10 ⁻⁴
Waste (g)	6.25x10 ⁻²	0.7-	- ^c	3.93x10 ⁻¹	2x10 ⁻³	2.3

^a Source: Brunel (2008)

^b Source: Electricity: own calculations; water: Defra (2008); waste: Holding et al. (2010)

^c Waste packaging is considered in the end of life stage.

3. Results and Discussion

Global warming potential (GWP) is estimated at 155 g CO₂ eq. per functional unit (f.u). As shown in Figure 1, the main hotspots are the raw materials (ingredients), contributing 70% to the total and manufacturing which adds 15%. In the raw materials stage, milk powder is the most important ingredient, causing 73% of the impact from this stage. This is due to the need to use of 7.8 kg of raw milk for the production of 1 kg of milk powder (Nielsen et al. 2003). The drying process contributes 73% to the GWP from the manufacture stage.

Analysing the contribution of individual greenhouse gases across the whole life cycle, 71% of the emissions are associated with the raw materials, of which 19% is carbon dioxide, 20% nitrous oxide and 32% methane. In the manufacturing stage, emissions from electricity generation add 14% to the total emissions along the supply chain (largely carbon dioxide, with minor contributions from nitrous oxide).

Given the high contribution of the raw materials to the total impact, a scenario analysis has been carried out to consider how different product formulations could reduce the GWP. Four scenarios are examined, considering 5-20% higher amount of cereals and 5-20% lower quantity of milk powder compared to the base case (Table 3). The results in Figure 2 suggest that the GWP reduces to 119 g CO₂ eq./f.u. in the best case (Product 4), representing a 23% reduction on the base case; in the worst case, the reduction of 6% could be achieved. Therefore, reducing the proportion of milk in the product while increasing the share of cereals could help to mitigate climate change impacts from the product. However, in that case some of the impact is transferred to the manufacturing stage due to the changes needed in the drying and mixing processes which depend on the composition of the product.

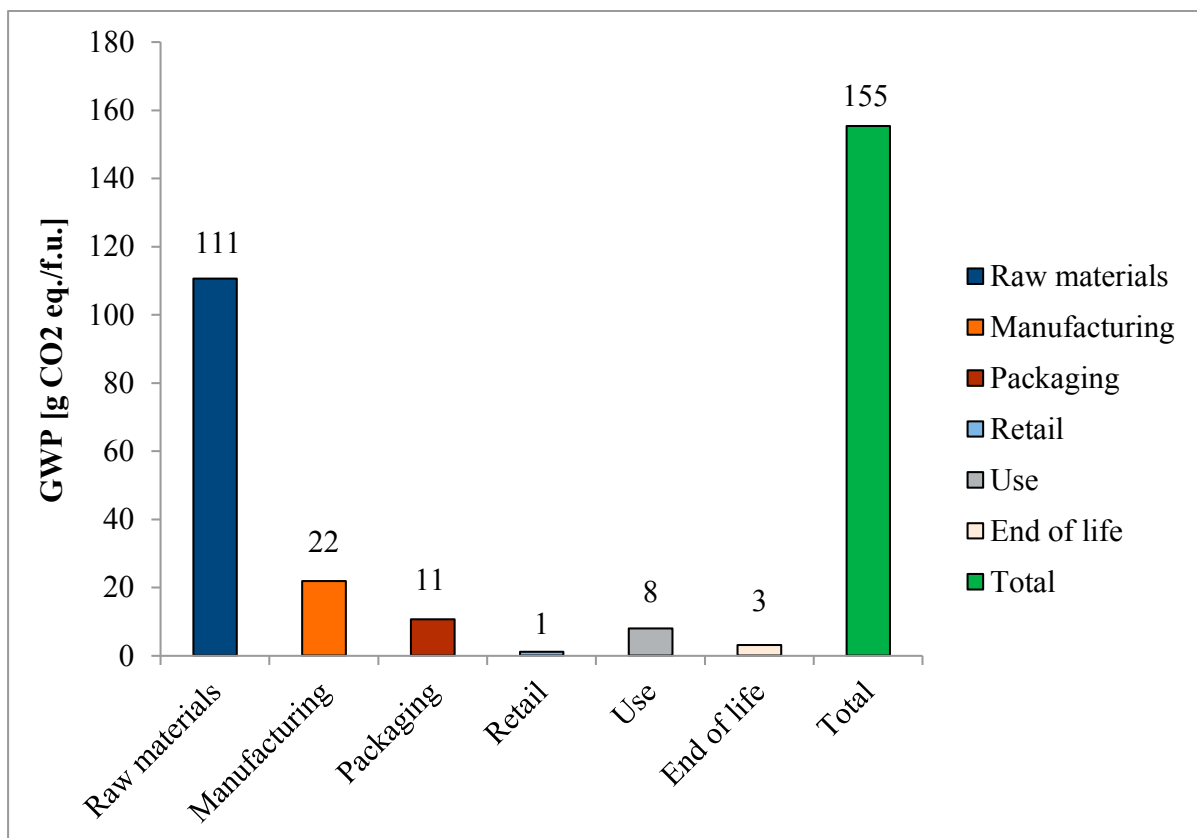


Figure 1: Global warming potential of ready-made baby oat porridge by life cycle stage (expressed per functional unit of 125 of porridge consumed at home)

Table 3: Composition of oat porridge in the base case and for different scenarios

Ingredients (%)	Base case	Product 1	Product 2	Product 3	Product 4
Oat flakes	35	39	43	46	50
Rice flour	11	12	13	15	16
Milk powder	30	25	20	15	10
Sugar	20	20	20	20	20
Palm oil	3	3	3	3	3
Malt extract	1	1	1	1	1
<i>Total</i>	<i>100</i>	<i>100</i>	<i>100</i>	<i>100</i>	<i>100</i>

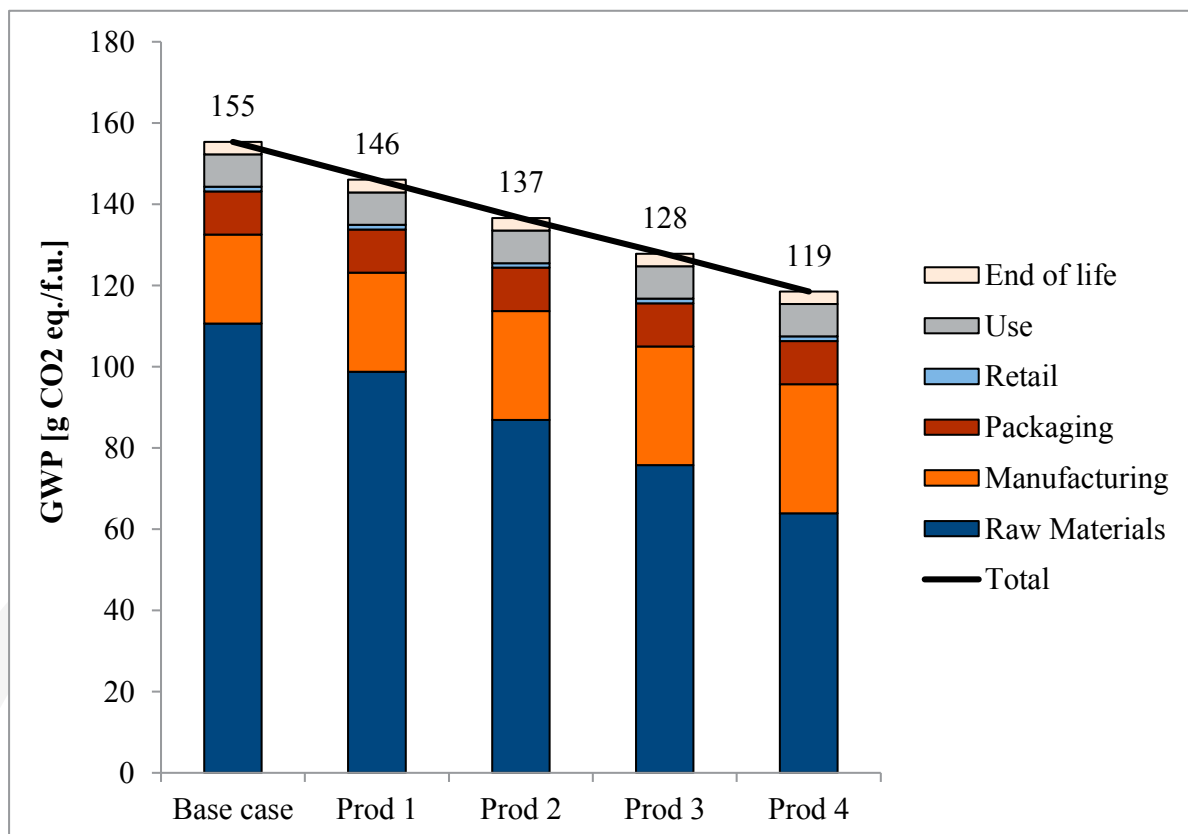


Figure 2: Opportunities for reducing global warming potential through alternative recipe formulations (expressed per functional unit of 125 of porridge consumed at home)

5. Conclusions

This study has considered global warming potential of ready-made oat porridge baby food, which is estimated at 155 g CO₂ eq./f.u. Around 70% of the impact is due to the raw materials and 15% to manufacturing. The former is largely due to the impact from milk and the latter due to drying of the product in the manufacturing process. Therefore, these results can help food companies to identify how baby food products could be re-designed to improve their environmental sustainability. However, this information should be considered alongside other factors, such as required changes in the production process and associated energy implications and production costs, as well as the nutritional aspects of the product, to ensure that one sustainability issue is not solved at the expense of another.

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246. Bioplastics: Does biodegradability have any ecological advantage per se?

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ABSTRACT

We analysed the environmental (dis)advantage of bioplastics with regard to their biodegradability in several studies. Data for composting and anaerobic digestion of bioplastics were derived from standardized composting and digestion studies. For the interpretation of the environmental impacts the ecological scarcity method was used for decision support and as a validation compared to the ILCD single score method. Biodegradability does not have an advantage per se from an environmental perspective. Composting of bioplastics leads to its disappearance in the best case. But there is no added value such as fertilisation or soil improvement (organic matter, humus) that normally occurs by composting organic waste. The same applies to digestate of bioplastics. Anaerobic digestion produces methane that can substitute natural gas. Whereas only about 60 % of the energy content can be “harvested” with anaerobic digestion, thermal exploitation in an incineration plant has higher yields. Biodegradability has no environmental advantage per se if compared to other end of life treatment options.

Keywords: bioplastics, composting, anaerobic digestion, thermal utilization, soil improvement

1. Introduction

The market of biodegradable bioplastics is growing each year. And the property biodegradable – among others such as renewable – is used for the promotion of their ecological benefits. Nowadays, many day-to-day consumer goods such as take-a-way packaging is made of biodegradable bioplastics. But how does the optimal end of life treatment look like – except for recycling which was not part of this study? Shall we compost biodegradable plastics, or put them to an anaerobic digestion plant, or is thermal utilization in an incineration plant also a advisable option? On behalf of the Amt für Umwelt und Energie, Basel (AUE) and the Amt für Abfall, Wasser, Energie und Luft, Zürich (AWEL) a study was done answering these questions.

2. Methods (or Goal and Scope)

Goal and Scope

The goal of this study was to analyse the end of life treatment options anaerobic digestion, thermal utilization or composting of biodegradable bioplastics by means of life cycle assessment. As functional unit 1 kg of biodegradable bioplastic was chosen. Only processes of the waste streams were included. The production and the use phase of the bioplastics were excluded because they are outside the system boundary and not necessary to answer the questions of the study. The different end of life treatments deliver different products: methane and digestate for anaerobic digestion, electricity and heat for thermal utilization and some sort of compost for composting. In order to make the end of life treatments comparable, credits were given for the different products assuming that they replace similar products on the market (e.g. electricity from a thermal utilization plant replaces otherwise produced electricity etc.).

Inventory data

Data for composting were derived from Pladerer et al. (2008). Data for anaerobic digestion of bioplastics are based on a preliminary study about the degradability of bioplastics in anaerobic digestion plants. In this preliminary study the actual degradability rates and methane yields were measured under standardized digestion settings (Baier, 2012).

Data for energy recovery in incineration plants were taken from the Ryttec report (2013) on energy efficiency of Swiss incinerations plants.

The data and methodology of Dinkel et al. (2011) described in Zschokke et al. (2012) was used to derive credits for organic matter.

Impact assessment methods

Different environmental impacts were analysed. But in order to guarantee sound and effective decision support aggregated single-score results were used (Kägi et al., 2016). Therefore, the ecological scarcity method (Frischknecht & Büsser Knöpfel, 2013) was used for the interpretation of the environmental impacts. This method was also used as a validation compared to the ILCD single score method (European Commission-Joint Research Centre, 2011), using the weighting scheme suggested by Huppel et al. (2011).

3. Results

Figure 1 shows the overall results of three end of life treatments of the three bioplastics polylactic acid (PLA), starch blend and cellulose acetate. It is obvious that incineration is always among the best end of life treatment options, whereas composting seems always to perform worst.

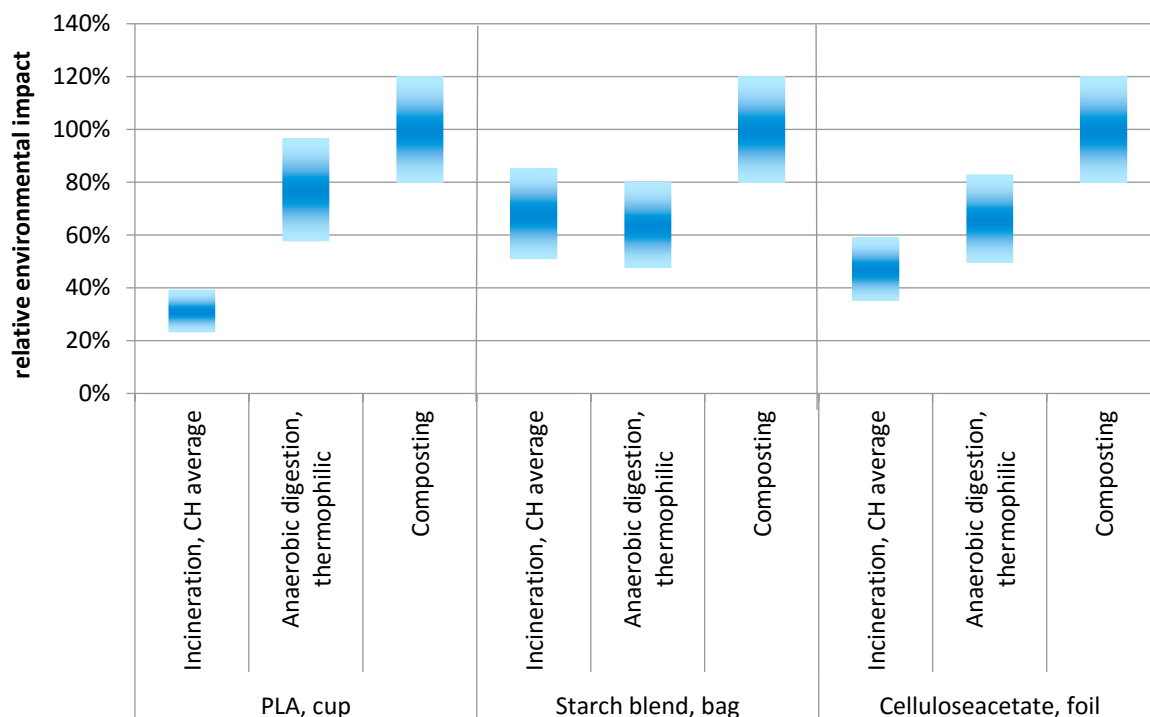


Figure 1: Relative environmental impact of different end of life treatments of bioplastics (PLA, starch blend, cellulose acetate) using the ecological scarcity method 2013 and the basket of benefits concept.

The reason can be better understood in analysing figure 2 which shows the environmental impacts and benefits of different end of life treatments of bioplastics such as cellulose acetate and biomass (example of palm leave; presented here to better understand credits for organic matter).

Inspecting the environmental impacts due to emissions only (figure 2, example of cellulose acetate), incineration shows the highest impact due to air emissions followed by anaerobic digestion and composting. For biomass (figure 2, palm leave) the results look differently: Anaerobic digestions and composting show higher impacts than incineration mainly due to the heavy metal emission to soil (digestate and compost). As there are no such heavy metals in bioplastics, the corresponding emissions do not exist at all.

Looking at credits only with the example of cellulose acetate, incineration shows the highest credits due to sold electricity and heat (replacing marginal electricity and heat), followed by anaerobic digestion with credits for sold biogas (replacing natural gas in a co-generation plant). The biogas credits are lower because – among other reasons – only a certain fraction of the embedded energy is transferred to methane (the remaining carbon is transferred to CO₂ or is not converted at all and remains in the organic residues). No credits are given for organic matter (humus) and for fertilisers. This stays in contrast to biomass, for which quite high credits are given. This is due to the fact that the

considered bioplastics do not contain any substantial nutrients such as nitrogen, phosphorus, or potassium. They are also lacking any structural molecules that could lead to humus build-up or complex top soil structures. Humus is only formed if some sort of lignin or complexing agents are included, which is the case for biomass but not for bioplastics (Dinkel & Kägi, 2013; Zschokke et al., 2012). This implies that the carbon in bioplastic digestate or compost is weakly bound and will therefore be metabolised to CO₂ sooner or later.

Over all, the credits are higher than the impacts, therefore leading to negative total results as only the end of life step was considered. In general, the anaerobic digestion and composting show equal or worse results than the incineration path for bioplastics.

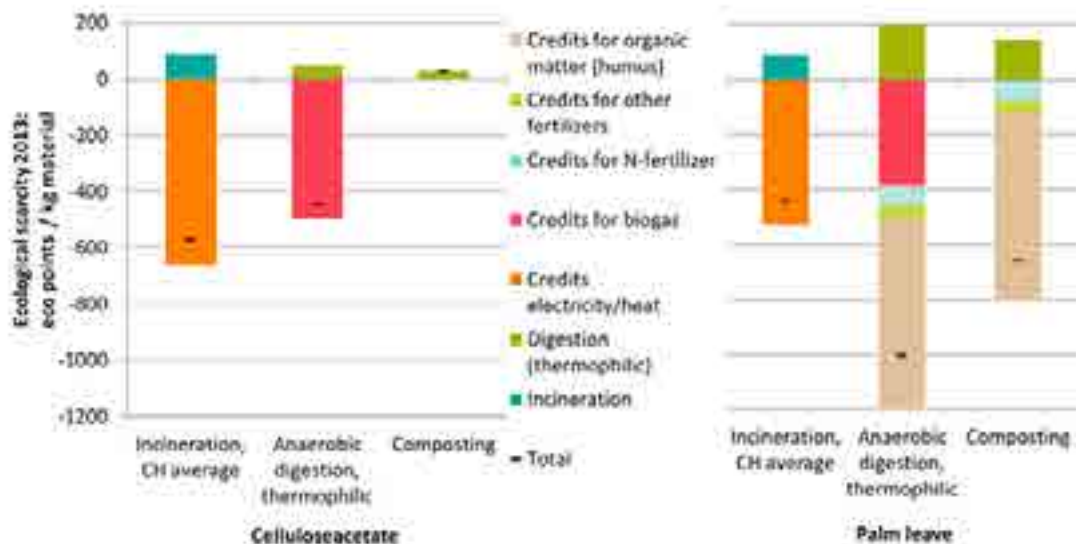


Figure 2: Environmental benefit of different end of life treatments of bioplastics (cellulose acetate) and biomass (palm leave) as an example for credits for organic matter using the ecological scarcity 2013 method and the avoided burden concept.

4. Discussion

Biodegradability does not have an advantage per se from an environmental perspective. Composting of bioplastics leads to its disappearance in the best case. But there is no added value such as fertilisation or soil improvement (organic matter, humus) which normally occurs by composting organic waste. The same is true for digestate of bioplastics. Anaerobic digestion leads to methane that can substitute natural gas. But whereas only about 60 % of the energy content can be “harvested” with anaerobic digestion, thermal exploitation in an incineration can mineralise almost all of the organic matter and shows always similar or even better results than anaerobic digestion.

5. Conclusions

Biodegradability has no environmental advantage per se if compared to other end of life treatment options. Biodegradability of bioplastics does not reduce the environmental footprint. On the contrary, the biodegradation of bioplastics often leads to higher environmental footprints compared to incinerating them. Our results are of course only valid for countries in which incineration is combined with energy recovery. In other countries where landfilling is normally employed and incineration plants are missing, the biodegradability of bioplastics may have advantages.

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241. Life Cycle Assessment of gelatin extracted from tilapia residues

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ABSTRACT

The skin and other residues that arise from the production of tilapia fillets can be used as feedstock for gelatin extraction. In order to derive maximum benefit from these residues, two gelatin extraction macroprocesses were developed and are evaluated: i) A, which uses the tilapia skin; and ii) B, which uses the residues of mechanically separated fish meat. Life cycle assessment was applied considering the unit processes of gelatin extraction and the production of inputs. Inventory data relating to the transportation and production of chemicals and generation of electricity were derived from the Ecoinvent v.3 database. The ReCiPe midpoint method and the USEtox model were applied to evaluate the categories of impact. The Monte Carlo simulation method was employed in the comparative impact assessment of the two gelatin extraction macroprocesses to analyze uncertainties in the results. Results showed that production of gelatin from macroprocess A is less environmentally intensive than that from B, among all impact categories. In macroprocess A, the extraction and milling processes were the largest contributors to the impacts assessed, while in macroprocess B, filtration and milling processes caused the highest impacts. Furthermore, the replacement of some reagents was investigated. This analysis showed that the use of sulfuric acid instead of hydrochloric and acetic acids reduced the environmental burdens of gelatin in both macroprocesses, but macroprocess A, still performed better than B. This study is original for evaluating the environmental impacts of gelatin production from tilapia residues, developing improvement options, and indicating technological routes for pilot scale developments and discussing the challenges.

Keywords: List 3 to 5 terms. Do not repeat words found in the title if possible. 9 point Times New Roman, Justified, Single Spaced.

1. Introduction

The culture of tilapia, including that of Nile tilapia (*Oreochromis niloticus*), is the largest among farmed fish worldwide and takes place in more than 135 countries (FAO 2014). Brazil is one of the seven major producers of tilapia in the world, producing around 254,000 tons of tilapia in 2011, which accounts for almost half of all fish produced in the Brazilian aquaculture industry (ACEB 2014).

Nile tilapia has high commercial value, low production costs, and excellent feed conversion efficiency, since 30% of its yield comprise the fillet. The other 70% is composed of skin, bones, cartilage, guts, fins, and head. Tilapia bones, cartilage, fins, and head are typically used in the extraction of mechanically separated meat (MSM), such as that used in the production of hamburgers and pates. However, not all of these materials are converted into MSM and 40% remains as a residue that can be transformed into collagen and gelatin (Morais et al. 2013).

Gelatin is derived from the hydrolysis of collagen, a major protein that forms the extracellular matrix. It has been used as a thickening and emulsifying agent in confectionery, dairy, and bakery products (Tavakolipour 2001). In recent years, gelatin has also been used in non-food applications, for example, as an implant or intravenous infusion matrix, or as a polymeric matrix in the composition of films for food coating (Santos et al. 2014).

Although gelatin has been historically obtained from beef and pork residues, diseases and religious restrictions have fostered the search by consumers and industry for new protein sources, such as tilapia residues. Considering the availability of raw materials from tilapia residues, two processes were recently developed to extract gelatin from tilapia skin and MSM residues. These gelatins are of sufficient quality for use as a polymer matrix in biofilms, because of their gel strength, or Bloom 189 grams for gelatin extracted from MSM residues and 747 grams for gelatin from tilapia skin. Gel

strength between 150 and 220 grams is considered medium Bloom and that above 220 grams, high Bloom (Johnston-Banks 1990).

Although the production of gelatin from tilapia residues is now technically viable, to our knowledge, the environmental burdens of the proposed extraction processes have not been studied. This knowledge is essential to identify hot spots, discuss possible alternatives, and define future research needs to improve the environmental sustainability of the proposed macro processes.

In this context, this study evaluates the environmental impact of two gelatin extraction macro processes: (i) macro process A that use Nile tilapia skin as feedstock; and (ii) macro process B that use MSM residues. This study is original for evaluating the environmental impacts of gelatin from tilapia residues, devising improvement options and indicating technological routes for pilot scale developments.

2. Methods

Life cycle assessment (LCA) was applied according to standards ISO 14040 (ISO 2006a) and 14044 (ISO 2006b).

The gelatin system under study includes the unit processes related to the two gelatin macro processes (Figure 1). Due to the high moisture content and degradability of tilapia residues, it was assumed that gelatin extraction occurs in fillet processing units. As the filleting industry still considers the skin of tilapia and the remains of MSM as wastes, without economic value, the environmental impact of tilapia aquaculture production and transportation of fish to the filleting production units were disregarded.

The functional unit under consideration is the production of one gram of gelatin.

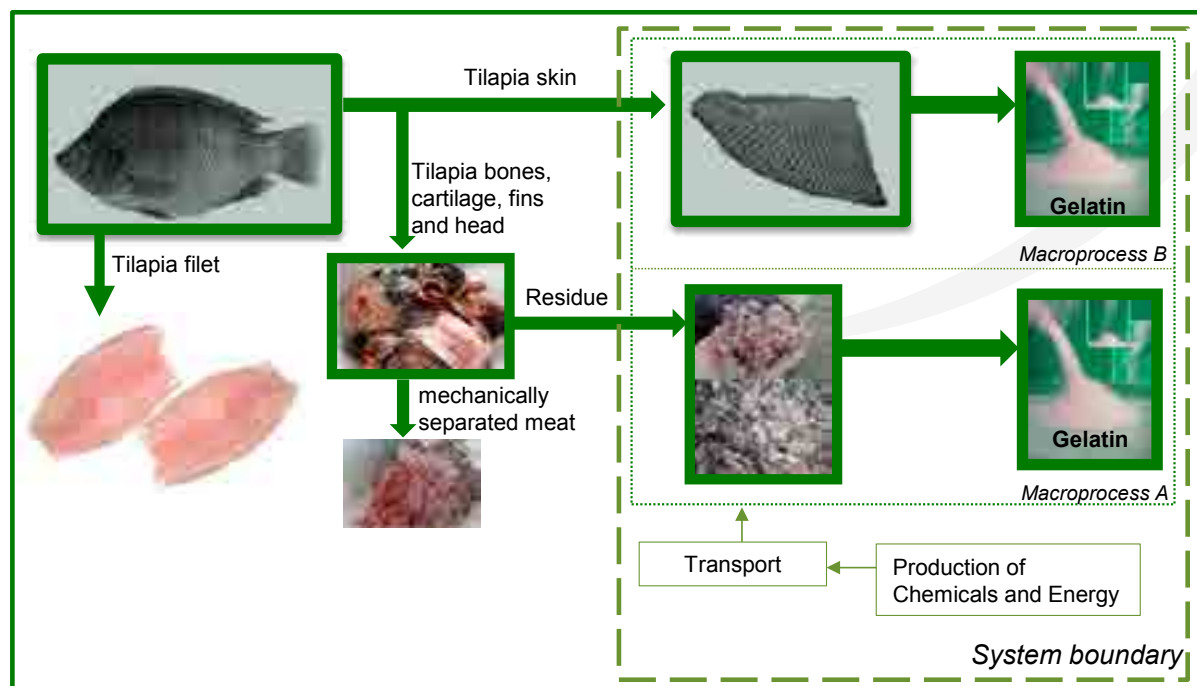


Figure 1. System Boundary

2.1 Inventory data collection

Foreground data relating to macro processes A and B were obtained in the laboratory of Biomass Technology at Embrapa Tropical Agroindustry, from August 2013 to 2014. Liquid effluents from these macro processes were analyzed, considering the following parameters and methods (APHA 2005): chemical oxygen demand (COD), closed reflux method; biochemical oxygen demand (BOD), modified Winkler method; nitrate, salicylate method; total phosphorus, digestion method with persulfate and ascorbic acid; ammonia and total nitrogen, distillation method.

Inventory data relating to the transportation and production of chemicals and generation of electricity were derived from the ecoinvent v.3 (Frischknecht et al. 2007).

Procedures regarding the foreground data are detailed below.

2.1.1 Filleting and separation of MSM

The filleting process produces the fillet, in addition to residues from skin, guts, and other remains. These remains include bones, leftover tissues attached to the bones, cartilage, and heads. The fillet was manually removed with a dorso-ventral cut, from the head towards the tail of the fish. The skin was used as a raw material to extract gelatin in macro process A, and the filleting remains were processed in a machine that removed the bones, which generated 30% MSM and 38% MSM residues. These MSM residues were used for gelatin extraction in macro process B.

2.1.2 Macro process A: gelatin extraction from tilapia skin

Macro process A consisted of nine unit processes, all performed around 24°C:

- Demineralization: the skins were immersed in 1 mol.l-1 hydrochloric acid (HCl) (at a 1/5 proportion of sample/solution) for 60 min with constant stirring.
- First washing: the solution of the previous process was filtered, and liquid effluent 1 was generated. The swollen skin was then washed with distilled water until a pH of 7 was achieved.
- Basic hydrolysis: 0.1 mol.l-1 sodium hydroxide (NaOH) was added at a proportion of 1/5 (sample/skin washing solution), and maintained under conditions of constant stirring for 60 min.
- Second washing: the resultant solution was filtered and liquid effluent 2 was thereby generated. The skin was then washed again with distilled water until achieving a pH of 7.
- Acid hydrolysis: 0.2 mol.l-1 acetic acid (CH₃COOH) was added to the washed skin at a proportion of 1/5 (sample/solution) and subjected to constant stirring for 60 min.
- Third washing: the solution of the previous process was filtered, thereby generating liquid effluent 3. The skin was once again washed with distilled water until achieving a pH of 7.
- Extraction: distilled water was then added to the skin at a temperature of 60°C, at a proportion of 1/5 (sample/solution) and subjected to constant stirring for 2 h. A magnetic stirrer with heating plate, model C-MAG HS 7, and a Frisatom mechanical stirrer, model 713D, were used throughout the procedure. The solution was filtered with the resultant generation of skin debris (effluent 4).
- Filtration: Cationic Resin Purolite C100 was added to the gelatin extract for 24 h to remove positive ions, such as calcium and magnesium. The gelatin was again filtered to remove the resin, which was subsequently recovered. This recovery was done by submerging the resin for 24 h in 1 M HCl solution and further washing with deionized water until a neutral pH was achieved.
- Milling: the gelatin extract was placed in an ultra-freezer for 24 h. It was then lyophilized using a LIOTOP lyophilizer, model LP510, for 30 h to remove excess water. It was then milled using an analytical mill, model IKA A11 basic.

2.1.3 Macro process B: gelatin extraction from MSM residues

This macro process consisted of nine unit processes, all performed around 24°C:

- First acid hydrolysis: the residues were immersed in 0.2 mol.l-1 CH₃COOH, at a proportion of 1/3 (sample/solution) for 90 min.
- First neutralization: the resultant solution was neutralized in a basic solution of 50% NaOH. The solution was then filtered to yield effluent 1.
- Basic hydrolysis: 0.2 mol.l-1 NaOH was added at a proportion of 1/3 (sample/remaining filtered solution), under constant stirring, for 60 min.
- Second neutralization: the resultant solution was neutralized in a solution of 50% (m/m) sulfuric acid (H₂SO₄). It was then filtered again, thereby generating liquid effluent 2.
- Second acid hydrolysis: 0.1 mol.l-1 H₂SO₄ was added to the remaining filtrate at a proportion of 1/3 (sample/solution), under constant agitation for 60 min.

- Third neutralization: the resultant solution was neutralized in a basic solution of 50% NaOH. The solution was then filtered again, thereby generating liquid effluent 3.
- Extraction: distilled water was added to the filtrate at a temperature of 60°C and at a proportion of 1/3 (sample/solution) for 2 h under constant stirring. A magnetic stirrer with heating plate, model C-MAG HS 7, and a Frisatom mechanical stirrer, model 713D, were used throughout the procedure. The solution was filtered and the solid residue of organic matter (effluent 4) was discarded.
- Filtration: activated charcoal was added to the gelatin sample and left for 24 h, to remove impurities and deodorize the sample. The solution was then centrifuged for 15 min at a speed of 13000 rpm, at a temperature of 28°C, and using a Hitachi centrifuge, model CR 22 GIII, to remove the activated charcoal. The gelatin extract was again filtered to remove any remnants of activated charcoal.
- Milling: the gelatin extract was placed in an ultra-freezer for 24 h. It was then lyophilized using a LIOTOP lyophilizer, model LP510, for 30 h to remove excess water. It was then milled in a basic analytical mill, model IKA A11.

2.2 Impact assessment

The ReCiPe midpoint method, hierarchical version, was applied to evaluate the categories of acidification, freshwater eutrophication, marine eutrophication, and climate change (Goedkoop et al. 2009). The categories of freshwater ecotoxicity, carcinogenic, and non-carcinogenic human toxicity were assessed using the USEtox model (Rosenbaum et al. 2008).

2.3 Uncertainty and scenarios analysis

The Monte Carlo simulation method was employed in the comparative impact assessment of the two gelatin extraction macro processes to evaluate uncertainties in the results (Goedkoop et al. 2013). Each variable was considered to have a lognormal distribution and 1000 simulations were carried out to calculate deviations in the results.

A scenarios analysis was performed with the replacement of some of the reagents used in macro processes A and B. A literature review and laboratorial tests were conducted to identify feasible production alternatives, instead of that currently employed at the macro processes. As a result, the following gelatin extraction scenarios were defined for macro process A (Grossman et al. 1989; Niu et al. 2013):

- Replacement of HCl by CH₃COOH and H₂SO₄ in the demineralization step.
- Replacement of CH₃COOH by HCl and H₂SO₄ in the acid hydrolysis step.

For macro process B, the following scenarios were defined:

- Replacement of H₂SO₄ by HCl and CH₃COOH in the second neutralization step.
- Replacement of NaOH by potassium hydroxide (KOH), sodium carbonate (Na₂CO₃) and potassium carbonate (K₂CO₃) in the third neutralization step.

3. Results and discussion

The comparison of macro processes A and B shows that the gelatin obtained from MSM residues (macro process B) presents higher environmental impacts than that from tilapia skin (macro process A), among all impact categories (Table 1). The uncertainty analysis indicates that the differences between the two macro processes are significant in all impact categories, at the 95% confidence level.

Table 1. Comparison of macroprocesses A and B

Impact Categories	Unit	Macroprocess A	Macroprocess B	A>=B
Human toxicity, non-cancer	CTUh	7.71x10 ⁻¹²	2.67x10 ⁻¹¹	0%
Human toxicity, cancer	CTUh	2.77x10 ⁻¹²	1.03x10 ⁻¹¹	0%

Freshwater ecotoxicity	CTUe	0.0003	0.0009	0%
Climate change	kg CO ₂ eq	0.03	0.1	0%
Marine eutrophication	kg N eq	6.8x10 ⁻⁶	2.57x10 ⁻⁵	0%
Freshwater eutrophication	kg P eq	5.18x10 ⁻⁶	5.56x10 ⁻⁵	0%
Acidification	kg SO ₂ eq	3.19x10 ⁻⁵	0.0002	0%

3.1 Analysis of macro process A: extraction of gelatin from skin

Regarding the environmental impacts of macro process A, the extraction and milling processes generate the highest impacts in the categories of climate change, freshwater ecotoxicity, and carcinogenic and non-carcinogenic human toxicity. This is mainly due to the considerable energy demands of these processes. According to the national energy balance, 65.2% of power in the Brazilian electricity grid is from hydropower plants (BEB 2015). The construction of these facilities can cause flooding that lead to deforestation and decomposition of plant debris, which consequently releases greenhouse gases. Although energy generation from the burning of sugarcane bagasse represents only 7.3% of power in the electricity grid, sugar cane production emits toxic substances like aldrin (a persistent organic pollutant) and 1-butanol (a combustible solvent), both of which are harmful to human health.

The demineralization process has the most significant impact on acidification and freshwater eutrophication. This process use HCl, which causes acidification by releasing H⁺ ions. The process of demineralization also releases phosphate ions, which lead to eutrophication.

For marine eutrophication, the most damaging processes are the first, second, and third washings. Washings are important to remove residues from under the skin of the tilapia. Part of this residual waste contains nitrogenous compounds, one of the main causes of marine eutrophication.

3.2 Analysis of macro process B: extraction of gelatin from MSM residues

At macro process B, filtration and milling are the most impacting processes in the categories of acidification, climate change, freshwater ecotoxicity, carcinogenic, and non-carcinogenic human toxicity. This was due to the energy demand of centrifugation, which removes the activated carbon used in the process. Furthermore, the ultra-freezer, the lyophilizer, and the analytical mill also add to energy demands.

Milling and neutralization have the most significant impact on freshwater and marine eutrophication. This is due to energy consumption during these processes and chemicals production (H₂SO₄ and NaOH) used in the neutralization process. The production of these reagents generates chemical wastes containing nitrogen and phosphate, which contributes to the eutrophication of water bodies.

3.3 Scenario analysis

The impact of the substitution of chemicals is evaluated to identify possible alternatives to reduce the environmental impact of macro processes A and B.

3.3.1 Replacement of HCl in macro process A

HCl used in the demineralization process is the main responsible for the environmental burdens in the categories of acidification and freshwater eutrophication. Niu and other researchers developed a gelatin extraction process from tilapia using different acids, such as H₂SO₄, CH₃COOH, and HCl, at varying concentrations (Niu et al. 2013). These authors concluded that regardless of the acid selected, the extracted gelatins showed no significant difference in yield or quality. Therefore, HCl can be substituted by H₂SO₄ or CH₃COOH without significant variation in the final product.

By replacing HCl with CH₃COOH, the environmental impact is lower in the category of marine eutrophication, and significantly lower in the categories of carcinogenic, and non-carcinogenic human toxicity. However, the impact is similar for all other categories under investigation. However, when

HCl is replaced by H₂SO₄, the impact is similar for the acidification category, but significantly lower for all other categories.

3.3.2 Replacement of CH₃COOH in macro process A

Because the production of CH₃COOH, used in the acid hydrolysis process, is the main source of emissions causing eutrophication in macro process A, scenario analyses is conducted based on its replacement by HCl and H₂SO₄. Kittiphattanabawon et al. (2010) carried out a comparative study of the characteristics of gelatin obtained from sharkskin using different acids and extraction conditions. That study concluded that the properties of the gelatin were affected not by the selection of various acids but by the temperature at which the extraction was conducted. These findings were consistent with other studies reporting that at a higher extraction temperature, the gel strength of gelatin was observed to be lower (Nagarajan et al. 2012; Sompie et al. 2015).

By replacing CH₃COOH with HCl, the impact is observed to be similar for marine eutrophication, carcinogenic, and non-carcinogenic human toxicity. Impacts in the other categories are slightly reduced. Furthermore, when CH₃COOH is substituted by H₂SO₄, impact decreases in all categories, with the exception of acidification, for which the impact remains similar.

3.3.3 Replacement of chemicals (H₂SO₄ and NaOH) in macro process B

In macro process B, the substitution of H₂SO₄ by CH₃COOH or HCl in the second neutralization was analyzed. Moreover, the substitution of NaOH in the third neutralization was also assessed. KOH, Na₂CO₃, and K₂CO₃ can be used to replace NaOH without any loss of gelatin quality (Grossman et al. 1989).

The results of this study demonstrated that each of these substitutions significantly reduced the environmental impact of macro process B. However, even with simultaneous replacement of H₂SO₄ by HCl and NaOH by Na₂CO₃ (substitutions that each resulted in lower environmental impact), the performance of macro process B is still worse than that of macro process A.

4. Conclusions (or Interpretation)

The main contribution of this work is the determination of an environmentally sound technological route for scaling up the production process of gelatin from tilapia residues. According to this study, the production of gelatin from tilapia residues at the pilot scale shall use skin as a raw material and replace acetic and hydrochloric acids by sulfuric acid.

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268. Milk Processing – Life cycle assessment of a detailed dairy model and recommendations for the allocation to single products

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ABSTRACT

This study analyses the environmental impacts referring to dairy products and to the operation of a dairy. The analysis is based on a detailed, product-specific model calculation. The environmental impacts are analyzed from cradle to gate including and excluding the raw milk input. The environmental impacts are assessed with the midpoint methods suggested by ILCD.

The detailed dairy model allows the assignment of inputs and outputs for each sub-process to single dairy products and thus avoids allocation to a large extent. The analysis of the model dairy shows that raw milk production has the main impact in all categories. Consumer packaging has the second biggest impact in many categories.

The analysis of inputs to the different dairy products per kilogram shows that UHT milk uses more chemicals for cleaning compared to the other products. Cream uses more electricity and heat compared to UHT milk and to yogurt. This is in contrast to the allocation suggestion of Feitz et al. (2007). The allocation of chemicals, steam and electricity can be undertaken based on the detailed dairy model developed in this study.

Keywords: dairy, milk products, carbon footprint, allocation, milk processing

1. Introduction

The inputs and outputs of dairy processing are usually only available for the whole plant. There is little information about the assignment of different inputs and outputs to the single dairy products. This assignment is important since it greatly influences the impacts assigned to each dairy product.

In the European SUSMILK project, a detailed bottom-up modelling of a theoretical generic dairy was compiled with the product portfolio given in Table 4 (Maga and Font Brucart 2016). The model of Maga et al. gives the inputs and outputs for more than 40 production sub-processes in the dairy (i.e. separation, pasteurization) and a detailed modelling of CIP (clean in place) for each machinery involved. This model was complemented with additional inputs to account for all inputs of the dairy operation from cradle to gate¹ and results in the LCA dairy model (Jungbluth, Keller et al. 2016).

Table 4: Daily amount of raw milk input and dairy products output produced in the LCA dairy model (kg/d).

	Flow name	Packaging	Amount
Raw milk input	Raw milk (4,2 % fat)	None	618'387
Dairy products	UHT ² milk (3,5 % fat)	Tetra Brik 1 l	103'125
	Stirred yogurt (10 % fat)	Polypropylene cup, 0.15 l	25'959
	Cream (30 % fat)	Tetra Brik 0.25 l	20'022
	Concentrated milk (0,2 % fat)	None	121'337
	Cream (40 % fat)	None	29'609

With the LCA dairy model, the environmental impacts of process stages of dairy processing are analyzed from cradle to gate related both to the daily dairy operation as well as to different products.

The analysis of several improvement options (heat provision, cooling) is described in a detailed life cycle assessment to be published for this project (Jungbluth, Keller et al. 2016). Improvement options that were only analyzed in lab scale were not integrated in the LCA dairy model.

¹ Additional inputs are i.e. packaging material, infrastructure and additional water and electricity inputs.

² UHT stands for Ultra-high-temperature processing. The milk is heated to 140°C.

Finally, the allocation of the inputs calculated according to the dairy model is compared to the allocation method suggested by the IDF (IDF 2010, based on Feitz, Lundie et al. 2007) and the differences in results are discussed.

2. Goal and Scope

This paper aims to show how relevant energy and water uses as well as different process stages in a model dairy are from an environmental point of view. It also aims to show the relevance of these process stages relating to the single dairy products at gate. The third aim is to present a way of allocation of dairy inputs onto different products, based on the detailed dairy model and compare these results to the recommendation of the International Dairy Foundation.

The scope of the LCA is from cradle to (dairy) gate, including the treatment of waste (i.e. waste water) up to gate plus post-consumer waste of packaging. One kilogram of processed raw milk is used as functional unit for the analysis of the dairy. This allows a comparison of dairies with different production volumes and product portfolios. The reference flow is one day of operation of the dairy model (600'000 liter raw milk). The functional unit for the analysis of the products is 1kg of dairy product. The LCA does not aim to compare different products or dairies directly.

The cumulative life cycle inventory data is assessed with impact assessment categories recommended by the ILCD at midpoint level (European Commission, Joint Research Centre et al. 2010).

3. LCI

The detailed dairy model was developed together with project partners, based on literature and estimations from dairy experts (Maga and Font Brucart 2016). All internal streams of the processing for single products (see product portfolio in Table 5) as well as of steam (heat provision), cold water (cooling) and electricity are modelled.

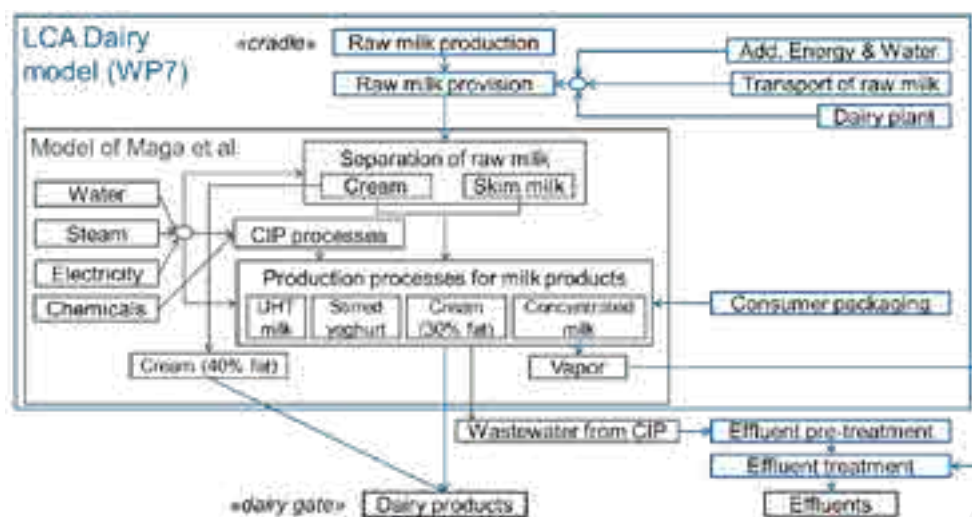


Figure 3 System boundaries and simplified model design of the LCA dairy model on milk processing. The inputs (i.e. steam, water) are specific for the respective dairy products. Circles are used to collect and redistribute the various inputs to the five products.

Table 5: Properties of the products of the model dairy, given in mass percentage

Product	Raw milk	UHT milk	Stirred yogurt	Cream (30% fat)	Concentrated milk	Cream (40% fat)	Skim milk
Water	87.10	87.73	80.56	63.45	68.25	54.55	90.87
Fat	4.20	3.50	10.00	30.00	0.20	40.00	0.05
Protein	3.30	3.33	3.58	2.42	11.97	2.07	3.44
Milk solids	12.90	12.27	19.44	36.55	31.75	45.45	9.13

The inputs of the dairy model are grouped into process stages for analysis (see Table 6), both according to aspects with high impacts (i.e. consumer packaging) and distinctions important for dairy producers (chemicals, electricity for production and for additional use).

Table 6: Name of the process stages used for analysis and the description of their main inputs.

Name of the process stage	Description
Raw milk production	Input of raw milk for processing excluding purchased products (e.g. milk powder)
Purchased products; dairy plant; additions	Purchased ingredients (e.g. milk powder), infrastructure of dairy plant, additional inputs (i.e. water and detergents; excluding additional electricity)
Transport of raw milk	Refrigerated transport of raw milk to the dairy
Effluent (pre-)treatment	Treatment of wastewater inside and outside the dairy, excluding electricity for pre-treatment as this is included in “Electricity, additional”
Consumer packaging	Product packaging (production and disposal)
Electricity, additional	Additional electricity use according to the LCA dairy model based on average literature data for electricity consumption of dairies minus “Electricity” as covered in the generic dairy model.
Electricity	Electricity use for production and the packaging process plus estimated use for lighting and compressed air according to the modelling in the generic dairy model
Steam for production /CIP ³	Heat use delivered by steam for production / for CIP
Chemicals	Chemicals used for CIP
Water use	All inputs needed for water use and cooling, including refrigerants, infrastructure, excluding electricity use

Table 7 shows important inputs and outputs of the LCA dairy model that includes packaging material, raw milk input and wastewater treatment plus additional water and electricity use. The additional inputs are added to the dataset of the raw milk provision (Jungbluth, Keller et al. 2016). The ecoinvent database and available updates, as well as ESU data-on-demand are used as a background database (ecoinvent Centre 2010; ESU 2016; Jungbluth, Meili et al. 2016). The raw milk separation step⁴ is allocated with milk solids (given in Table 5) as suggested by the IDF (IDF 2010) and Feitz et al (2007).

³ CIP means “Clean-in-Place” and is a method of cleaning the interior surfaces of machinery (e.g. pipes, vessels, process equipment) without disassembly.

⁴ Raw milk is separated into cream, 40% fat with a content of milk solids of 0.45 (weight per weight) and pasteurized skim milk, 0.05% fat with a content of milk solids of 0.09.

Table 7: Inputs per kg of product given by the LCA dairy model.

	Raw milk	Water use	Electricity	Steam use	NaOH 50 %	HNO3 70 %	Waste water
	kg	kg	MJ	MJ	G	g	l
UHT milk (3.5% fat)	1.0	1.2	0.3	0.4	6.070	1.086	1.261
Stirred yogurt (10% fat)	1.4	1.8	0.5	0.6	1.325	0.096	1.776
Cream (30% fat)	2.9	2.7	0.8	0.8	0.002	0.000	0.003
Concentrated milk (0.2% fat)	2.7	2.8	1.0	2.4	0.012	0.004	0.005
Cream (40% fat)	3.6	2.4	0.8	0.7	1.709	0.124	2.364

4. LCIA

Raw milk production has the highest share of impact in a cradle to gate analysis, varying from about half (water depletion, ozone depletion) up to almost hundred percent in the different impact categories. Raw milk production is therefore decisive for the environmental impact of the dairy products. But, this aspect lies outside the scope of the project and this LCA and it has therefore not been further investigated in detail.

The analysis of the dairy operation excluding the raw milk production⁵ shows that the crucial process stage depends on the impact category (see Figure 4). The transport of raw milk (refrigeration truck) shows the highest share for acidification, ozone formation and terrestrial eutrophication. The consumer packaging has considerable shares in land use, particulate matter, abiotic resource depletion and all toxicity categories. The effluent treatment is most important for marine and freshwater eutrophication. The chemicals used for cleaning (NaOH, HNO3) have very little effect compared to the other process stages.

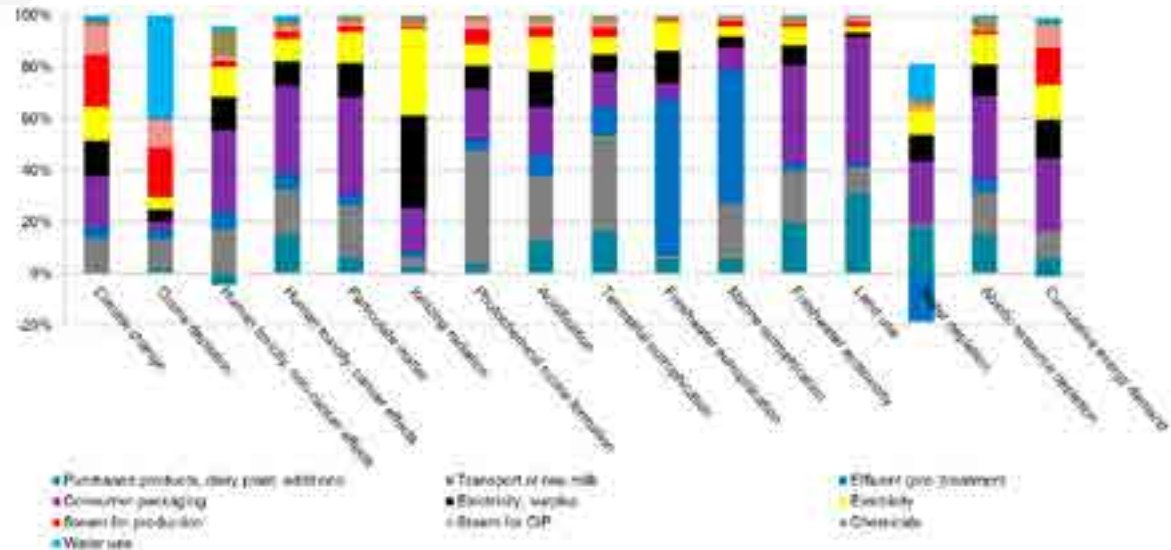


Figure 4 ILDC impact categories: Analysis of the dairy operation per day without the raw milk production and without allocation to single products. Percentage share of each process stage on the total impact in each category is depicted.

In the impact category climate change, the main impact stems from packaging of the UHT milk and cream (30% fat) which amount to 16% of the impact. When analyzing the packaging, around half stems from production and disposal of plastic parts and less than 20% each stem from the production

⁵ The model for operation includes water and waste water treatment, energy, wastes, packages incl. their disposal, infrastructure and the transport of raw milk.

of aluminum foil and cardboard. Second highest impact is the steam for production (20%), followed by steam for CIP (11%).

In the impact category water depletion, around 40% stems from packaging⁶. Almost 30% stems from additional water and electricity use that is added in the LCA dairy model. The discharge of water after the “effluent (Pre-) treatment” shows a negative percentage since for this stage as it gives back water to the environment. The water in the effluent stems from vapors from concentrated milk, tap water input and from CIP. All water input is shown in the process stage “water use” and amounts to 21% of total impact in this category. Thus, the output of water after treatment is subtracted in the water balance from all inputs of water.

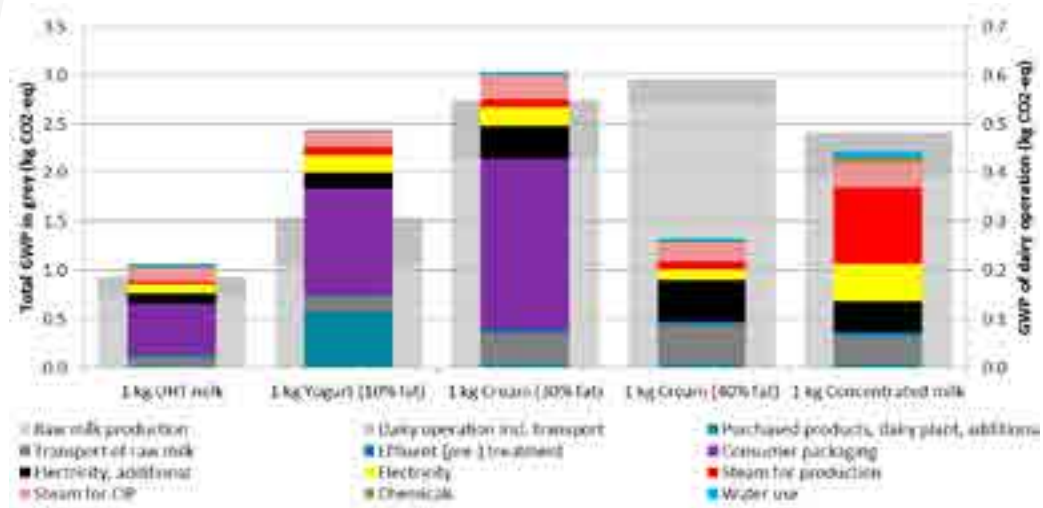
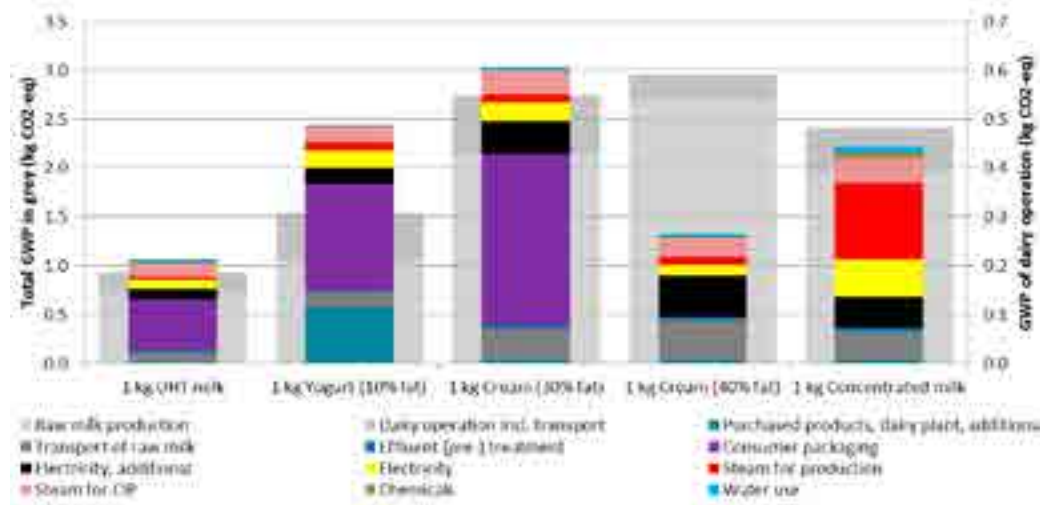


Figure 5 Comparison of impact on climate change (global warming potential, GWP) of dairy products at dairy gate. Grey columns in the background show the total GWP (cradle-to-gate), split into raw milk production and dairy operation (left axis). Coloured columns show the subdivision of the dairy operation (gate-to-gate) according to process stages (right axis).

Also when referring the impacts on climate change to the different dairy products, raw milk production has by far the biggest share of environmental impact spanning from 70% to 90% (see



⁶ For Tetra Brik, the water use stems from paper production, for the polystyrene packaging of the yogurt, the cooling water used for thermoforming has the main impact

Figure 5). The allocation of raw milk and of the separation step is conducted according to milk solids. Thus for climate change, the products with the highest milk solids content have the highest impacts. The concentrated milk has lower impacts than the cream due to this allocation choice. Steam for preheating the milk and for evaporating has the main impact for the unpacked concentrated milk, whereas for the unpacked cream (40% fat), the electricity (used for processing and electric cooling) has the main share. The share of electricity (for production plus additional uses, without waste-water treatment) varies from 14% to 40% of the climate impact, the transport of raw milk from the farm to the dairy contributes 6% to 30%.

5. Interpretation

5.1 Main results

The main impact of dairy products stems from the raw milk input. Therefore, the production systems used for the raw milk have a decisive role for the overall environmental impact of dairy products and should be given priority in environmental improvement strategies.

For the dairy operation, the amount of packaging used and an efficient transport of the raw milk to the processing plant are important, as well as an adequate waste water treatment. Energy and water uses in the dairy are of minor importance in most impact categories, but for climate change, the heat demand contributes most to the total impact.

The shares of impact of process stages are very different for the five considered dairy products. The importance of each process stage changes depending on the processing conducted. For impact on climate change of concentrated milk, the steam (i.e. heat) use should be given priority. An intelligent process design that reuses heat within the dairy and an efficient evaporation can be used to decrease heat demand. For yogurt production, the milk powder has an important share even though the respective input is less than 2% of the total yogurt weight⁷.

5.2. Allocation

Feitz et al. (2007) elaborated an allocation approach based on whole-of-plant data from 17 dairies. First, they collected total input data of dairies that only produce few products, like milk and cream. Later, they subtracted these values from the total input of dairies with a wider product portfolio. Finally, an allocation matrix for dairy products was elaborated that can be applied to whole-of-plant data of dairies with various product portfolios. This approach is part of the IDF recommendation for allocation (IDF 2010, Chapter 6.3.4).

Table 8 first shows the input per kg of market milk according to the model dairy used in the publication of Feitz et al. (Table 8a). Next, the allocation of the sum of inputs for these three products from the LCA dairy model with the method of Feitz et al. is shown (UHT milk in Table 8b and all three products in Table 9b).

The inputs per kg of market milk in the model dairy of Feitz et al. (Table 8a) are similar to the inputs of UHT milk in the LCA dairy model (Table 8b). An exception is the chemical input. There, a much higher amount is modelled in the LCA dairy model compared to Feitz et al.

⁷ This is due to the allocation behind the milk powder that is conducted based on milk solid content.

Table 8: Inputs per kg of market milk from the model of Feitz et al. and per kg of UHT milk for the LCA dairy model.

a) Input per kg of market milk according to the model dairy of Feitz et al. (2007)					
	<i>Raw milk</i>	<i>(Waste) water</i>	<i>Electricity</i>	<i>Fuel</i>	<i>Alkaline</i>
	kg	l/kg	MJ	MJ	G
Market milk	1	1.5	0.2	0.3	0.8
b) Allocation of the generic dairy inputs (3 products) according to Feitz et al. (2007)					
	<i>Raw milk</i>	<i>Water use</i>	<i>Electricity</i>	<i>Thermal energy</i>	<i>Alkaline cleaners</i>
UHT milk (3.7% fat)	1.1	1.3	0.4	0.5	4.5

Table 9 shows the allocation of Feitz et al. (Table 8a) and compares this to the allocation conducted in the LCA dairy model (Table 8b). It shows that not only the amount of chemicals used for UHT milk is higher in the LCA dairy model compared to the allocation according to Feitz et al., but also the share allocated to UHT milk is higher. In Feitz et al, the same share is suggested for these products. According to Feitz, the resolution in their study was not high enough to identify i.e. different cleaning figures for UHT milk and for fresh milk⁸. The values used in the LCA dairy model are specific to the products. They are calculated by defining cleaning programs for different operations based on literature data (assumptions are described in detail in Maga et al. 2016). The UHT unit and evaporator for the concentrated milk require longer cleaning programs and higher concentrations of chemical products. Plus, recirculation of chemicals and rinse water is not carried out. Since our model shows much higher inputs for UHT milk, there seems to be a substantial difference in chemical use between UHT and normal milk that should be taken into account. Therefore the SUSMILK model is more detailed for allocation for these inputs and could be used to further improve allocation recommendations.

Table 9: Inputs per kg of product with the allocation proposed by Feitz et al. (2007) for the 3 products yogurt, cream (40%) and UHT milk (6b) and inputs given by the LCA dairy model (6c).

a) Allocation of the generic dairy inputs (3 products) according to Feitz et al. (2007)							
	<i>Raw milk</i>	<i>Water use</i>	<i>Elec- tricity</i>	<i>Thermal energy</i>	<i>Alkaline cleaners</i>	<i>Acid cleaners</i>	<i>Waste water</i>
	kg	kg	MJ	MJ	g	g	l
Yogurt (0.2/3.4% fat)	1.2	2.5	1.0	0.8	4.5	0.745	2.535
Cream (40% fat)	3.6	1.3	0.2	0.2	4.5	0.745	1.358
UHT milk (3.7% fat)	1.1	1.3	0.4	0.5	4.5	0.745	1.358
b) Inputs according to the LCA dairy model							
	<i>Raw milk</i>	<i>Water use</i>	<i>Elec- tricity</i>	<i>Steam use</i>	<i>NaOH 50 %</i>	<i>HNO3 70 %</i>	<i>Waste water</i>
Yogurt (10% fat)	1.4	1.8	0.5	0.6	1.325	0.096	1.776
Cream (40% fat)	3.6	2.4	0.8	0.7	1.709	0.124	2.364
UHT milk (3.5% fat)	1.0	1.2	0.3	0.4	6.070	1.086	1.261

Table 10 shows the relative difference of the two allocation results. The comparison of the different allocation procedures shows the smallest difference for raw milk input. Yogurt has more raw milk input in the LCA dairy model because of the higher fat content of the yogurt in the LCA dairy model compared to the yogurt in the publication of Feitz et al. In the other process stages, the results of the two allocation types are very different, especially for cream (40% fat).

⁸ Feitz, Andrew. Personal communication via e-mail on 14.4.2016.

Table 10: Relative difference between the data of the LCA dairy model and the allocation of the LCA dairy model data as proposed by Feitz et al (2007) for the 3 products yogurt, cream (40%) and UHT milk. Formula used: (input in LCA dairy model – input Feitz)/input Feitz).

Relative change of allocation in our model compared to Feitz et al. (2007)							
	<i>Raw milk</i>	<i>Water use</i>	<i>Electricity</i>	<i>Thermal energy / Steam use</i>	<i>Alkaline cleaners / NaOH 50 %</i>	<i>Acid cleaners / HNO3 70 %</i>	<i>Waste water</i>
Yogurt	17%	-29%	-50%	-31%	-70%	-87%	-30%
Cream (40% fat)	3%	76%	357%	207%	-62%	-83%	74%
UHT milk	-7%	-8%	-12%	-15%	35%	46%	-7%

The water, steam and electricity use allocated to cream is much higher in our model than in the model of Feitz. In case of electricity, most of the electricity that is used for cream (40% fat) stems from the additional input modelled in the LCA dairy model. This input is added to the raw milk and the allocation of the milk separation step is conducted according to milk solids, a relatively high amount of this additional input is passed on to the cream (40% fat). In the case of water use and thermal energy (in the LCA dairy model: steam for CIP and for heating), most of the input stems from the separation and pasteurization step of raw milk, that is again passed on mainly to the cream. This could be an explanation why relatively more fuel is needed to produce cream (40% fat) in the LCA dairy model than expected according to the allocation of Feitz et al. Feitz⁹ states that they could not differentiate between standard cream and milk and assumed that they need the same amount of inputs. For this aspect, our model is more detailed and could be more accurate.

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ORAL PAPERS

WASTE AND BI/CO-PRODUCTS

15. Waste and bi/co-products

48. Environmental impacts and abatement costs of food waste reduction: the case of bread

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ABSTRACT

In this study, environmental impacts and abatement costs of reducing food waste in the life cycle of bread were calculated by connecting life cycle assessment with environmental life cycle costing. The life cycle includes production, processing, sale, consumption and disposal of mixed grain bread in Germany. The functional unit (FU) was set as 1 kg bread consumed. Four scenarios were modelled to examine the costs and impacts of different waste reduction measures: (1) a baseline scenario with no actions taken to reduce food waste, (2) reducing food waste at the retail stage by passing on unsold bread to food banks, (3) reducing food waste at the consumption stage by reducing the amount of bread shopped by 50% followed by a higher frequency of shopping and (4) reducing food waste at the consumption stage by freezing 50% of the bread and consume it to a later point in time. For all scenarios a strong and a weak food waste reduction effect was modelled to show the uncertainties. The life cycle inventory data was analyzed according to the impact categories global warming potential, agricultural land occupation, cumulative energy input and process costs. The calculation resulted in 2.51 kg CO₂eq greenhouse gas emissions, 18.04 MJ Energy input, 6.69 € process costs and 1.13 m²a agricultural land occupation per FU for the baseline scenario. The waste reduction measures (2) and (4) scored better than the baseline scenario in almost all impact categories with a strong and also a weak waste reduction effect, while measure (3) had higher greenhouse gas emissions, costs and also energy input (weak effect only) as compared to the baseline scenario. As a conclusion, the assessment of environmental impacts and costs of waste reduction actions should be of high priority when it comes to the choice of food waste reduction measures. Measures should be selected according to their case-specific cost-effectiveness that shows the relation between the abatement costs and resource reductions.

Keywords: food waste reduction measures, life cycle costs, mixed grain bread

1. Introduction

Worldwide, about one third of the food produced for human nutrition goes to waste (Gustavsson et al. 2011). In times of public awareness of food shortage in some parts of the world, resource scarcity and the environmental impact of food production, ambitions to reduce food waste are a prominent political and societal topic. In September 2015 the United Nations decided with the Sustainable Development Goals, target number 12.3 to “halve per capita food waste at the retail and consumer levels and reduce food losses along production and supply chains”. With this target in mind, the question about the type and consequences of food waste reduction measures arises.

Several studies have been conducted to examine the environmental impacts of food waste. Gruber et al. 2016 considered unconsumed food portions and concluded that avoiding food waste could reduce the environmental impact significantly. Eberle and Fels 2016 looked at the environmental impact of food waste in Germany along the whole supply chain, based on the average German food basket. According to their findings, losses along the product chains constitute between 13 and 20% of environmental impacts. The FAO (2013a) calculated a food wastage footprint with greenhouse gas (GHG) emissions of 3.3 Gt CO₂equivalents(eq), 30% of the world’s agricultural land occupation, and direct economic costs (based on producer prices) of 750 bill USD. Studies specifying the costs of food waste, usually refer to the monetary value consisting of the summed producer or consumer prices of the wasted food (e.g. FAO 2013a, Kranert et al. 2012). Little attention is paid to possible cost of food waste reduction either in monetary terms or also regarding other aspects such as additional time dedicated to reduce food waste. Britz et al. 2014 simulated food waste reduction scenarios and the impact on the different economy-sectors for Finland, using a regional CEG (computable general equilibrium) model. They argue that the use of waste reduction may cause severe loss of competitiveness for agriculture and food production if costs are not taken into account. Equally, Rutten and Kavallari (2013) modelled impacts of food loss reduction in agriculture in the Middle East and North Africa on economic sectors and food security. They rate reduction and thus enhanced food security as more beneficial than manufacturing and service-led growth. However, a macroeconomic view is not in the center of interest of this paper but rather a life cycle approach with the direct

environmental impacts and monetary costs of food waste reduction measures. Few studies have yet examined life cycle costs of food, including also food preparation at home (e.g. Hünecke et al. 2005 for Germany) while life-cycle costs of food waste reduction actions, have not been conducted at all (Koester 2014). Approaches to prevent food waste range from policy recommendations (especially Waarts et al. 2011, Jepsen et al. 2014) to changes in consumer behavior (Kranert et al. 2012, Göbel et al. 2012, FAO 2013b). Parry et al. (2015) assign the 15% reduction of household food waste in the UK from 2007-2012 to more attentive consumer behavior through e.g. buying appropriate amounts, storing under optimal conditions, using the freezer etc. and technical innovations such as different pack sizes, improved storage and freezing guidance, increased shelf-life, packaging innovations and clearer date labelling. According to Parfitt et al. (2010), the greatest potential to reduce food waste in industrialized countries lies within the retailer and consumer stage. In this study the environmental impacts and abatement costs of food waste reduction measures were calculated, exemplarily for the life cycle of bread. By including cost estimates in the assessment, the study offers a comprehensive evaluation of food waste reduction measures.

2. Methods

2.1 Goal and scope definition

For this study, a standard Life Cycle Assessment according to ISO 14040/44 was connected with an Environmental Life Cycle Costing (LCC), based on the sum of added values (Moreau and Weidema 2015). The product system under study is the life cycle of mixed grain bread in Germany, shown in Figure 1. It comprises the stages agricultural production, milling into flour, baking of bread, selling of bread, bread consumption and disposal during all stages of the supply chain. This operational framework is supposed to display a typical German production and consumption with involvement of medium-sized enterprises during processing and sale. The functional unit is one kilogram of bread consumed. The study is intended to serve as verification for different actions to fight food waste with the main goal to identify the most effective actions for food waste reduction. These actions concern the respective stakeholders at the different points of the supply chain but in addition also researchers and policy-makers for further discussion and application. The actions analyzed are:

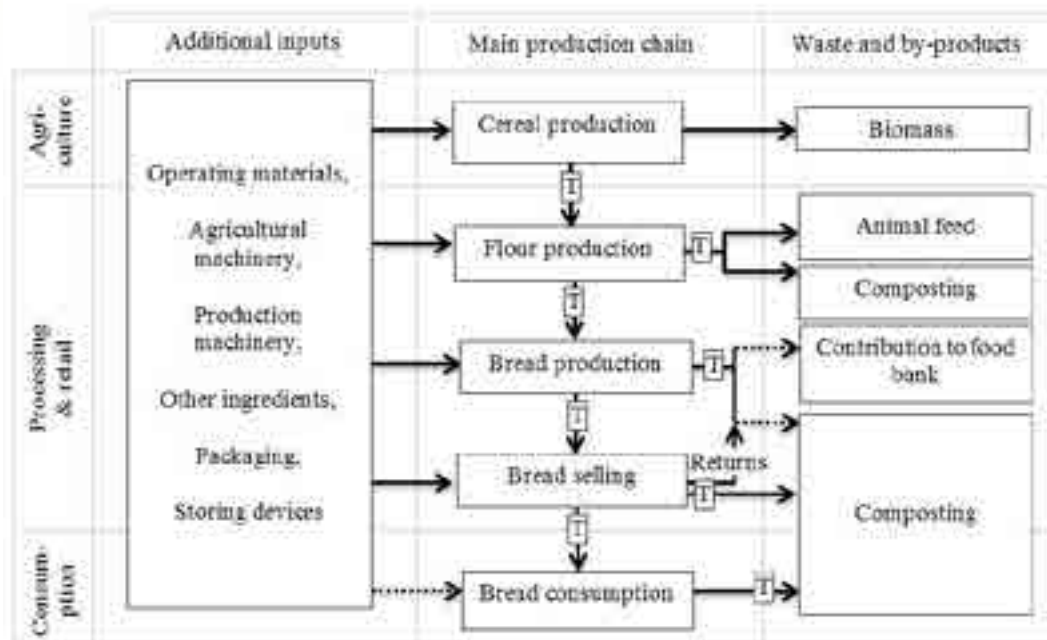
- (1) Baseline scenario with no actions taken to reduce food waste.
- (2) Reducing food waste at the retail stage by offering food that cannot be sold, but is still edible to food banks.
- (3) Reducing food waste at the consumption stage by reducing the amount of bread shopped by 50% followed by a higher frequency of shopping.
- (4) Reducing food waste at the consumption stage by freezing 50% of the bread and consume it to a later point in time.

2.2 System boundaries, assumptions and data origin

A mixed grain bread was chosen as it is with 33.7% the most frequently consumed type of bread in Germany (ZV Bäckerhandwerk 2015a). The main source for the calculation is the ecoinvent database (Weidema et al. 2013). All upstream inputs are based on ecoinvent data through the use of ecoinvent processes. Direct inputs are based on different sources of literature and on estimates for a typical German production chain. All sources of direct inputs used have been summed up in Table 1. Besides direct material and energy inputs, capital goods and tools are included in all ecoinvent processes. Capital goods such as machinery and tools of processes specifically modelled for this supply chain are also included in the calculation while this does not apply for houses and infrastructure. The LCC is conducted as a cursory calculation based on average values for bread production in Germany and exemplary values for additional costs arising within the respective scenarios. The process of agricultural cereal production refers to the production of wheat and rye in Germany within a conventional production system. It includes the inputs of seeds, mineral fertilizers and pesticides as well as the operations soil cultivation, sowing, fertilization, weed, pest and pathogen control, combine-harvest, grain drying and transport from field to farm (4km). Further, it also comprises the machine infrastructure and sheds. Losses during agricultural production refer to mature crop that is or has been edible e.g. not-harvested crops or crop loss at harvest and storage on the farm. The losses of this stage is handled as biomass contribution to the environment while the food waste at all other

stages of the supply chain is treated at a composting facility, taking into account the transport to the facility as well as direct inputs and emissions and capital goods.

Figure 1: Overview of the modelled life cycle of bread. Dashed arrows indicate different options considered in the model. “T” stands for transportation.



Source: Own illustration.

Table 1: Data sources for the direct inputs to the life cycle of bread

Processes and additional inputs	Amount	Source of amount used
Wheat and rye production (Germany)		Ecoinvent
Truck transport farm - mill	30 kgkm	Assumption
Production value wheat	0.19 €/kg wheat	BMEL 2015
Production value rye	0.13 €/kg rye	BMEL 2015
Flour production		
Electricity (German energy mix)	0.08 kWh/kg output	Nielsen et al. 2003
Organic chemicals	0.04 g/kg output	Nielsen et al. 2003
Tap water	0.1 l/kg output	Nielsen et al. 2003
Heat (gas)	0.1 kWh/kg output	Nielsen et al. 2003
Machinery	Throughput over whole lifetime 2 Mio. t of grain	Assumption
Paper sacks for transport	140 kg flour per sack	120g paper/m ²
Truck transport mill - bakery	30 kgkm	Assumption
Production value wheat flour	0.34 €/kg wheat flour	BMEL 2015
Production value other flour	0.31 €/kg other flour	BMEL 2015
Bread production		
Bread ingredients	-	Typical German recipe
Electricity (German energy mix)	0.02 kWh/kg output	Nielsen et al. 2003
Heat (gas), industrial furnace	1 MJ/kg output	Nielsen et al. 2003
Plastic baskets for transport	40 t throughput of bread	Assumption
Truck transport from bakery to shop	20 kgkm	Assumption
Production value bread	1.99 €/kg bread	BMEL 2015
Bread sale		
Paper bag as packaging, with print	4.9 g/kg bread	35 g paper/m ²
Consumer price bread	2.32 €/kg bread	ZV Bäckerhandwerk 2015b
Consumption		

Processes and additional inputs	Amount	Source of amount used
Passenger car transport bakery - home	6 km distance	Assumption, (both ways) for shopping 1 kg bread
Transport costs	0.6 €/km	ADAC, medium sized car

The waste portions for each step of the supply chain are shown in Table 2. The flour production takes place in a medium sized production facility in a distance of 20 km to the farm. The milling of the grain results in 80.5% flour, 19% animal feed and 0.5% processing waste. Animal feed is not considered as waste but as by-product. The flour is transported to the bakery in paper sacks with a capacity of 140 kg. At the bakery the mixed grain bread is produced, consisting of 35% wheat flour, 25% rye flour, 39% water and 1% salt. The bread is baked in an industrial gas-driven furnace (no baking pans used). After baking, the bread is transported to a local shop within a distance of 20 km in food transport boxes out of plastic. At the shop, the bread is sold in a paper bag and transported to the consumer's home in a passenger car. The distance between home and shop is assumed to be 3 km. It is further assumed, that shopping takes place for the bread only, therefore, the full distance of 6 km for the forward and back run is assigned to the transport of bread at the consumer stage.

Table 2: Waste portions of bread production at different stages (Jepsen et al. 2014)

	Agriculture	Postharvest	Milling	Baking	Selling	Consumption
Waste portion	2 %	4.9 %	0.5 %	10 %	2.4 %	11.1 %

The waste reduction scenarios are modelled with different degrees of effectiveness, namely, with a strong, and a weak effect (Table 3). Additional inputs considered for the different scenarios are shown in Table 4. In scenario (2), leftovers from the shop are transported on the return trip of the truck that brings the fresh produce. Therefore, no extra transport is added for this process. However, extra transport is calculated for the transfer of the products to the food bank with a large passenger car. Including forward and back run, the distance amounts to 30 km. For the organization of the donation, an estimated value of 5 seconds per kg bread is used. Additional labor due to donations to food banks is thought to be low, as only little additional logistical work occurs. The main work such as food collection and distribution is conducted by the food banks and staffed with voluntary workers. In scenario (3), only half of the usual amount of bread (0.5 kg) is purchased in connection with the activity of driving by car to the bakery and home, while less bread is wasted at the consumption stage. This also generates additional transport costs and a higher use of paper bags for packaging. Finally, in scenario (4) it is assumed, that half of the bread purchased is put into the freezer for 14 days and afterwards consumed completely (strong effect) or to 94,5% (weak effect). The de-freezing happens by exposing the bread at room temperature. In the freezing scenario, additional costs for the electricity consumed by the freezer and for the purchase of the freezer are accounted for. The freezer was selected as a small sized freezer with a capacity of 30 liters.

Table 3: Effectiveness of waste reduction scenarios at the respective life cycle stages

Scenario	Strong effect	Weak effect
(2) Contribution to food bank	-70 %	-35 %
(3) More frequent shopping	-100 %	-50 %
(4) Freezing and consuming later	-100 %	-50 %

Table 4: Additional inputs considered for the different food waste reduction measures

Scenario	Additional inputs	Amount	Source of amount used
(2) Contribution to food bank	Passenger car (large size) transport to food bank	30 km distance	Assumption (both ways) Transport capacity 500 kg, 60% utilization
	Transport costs	1 €/km	ADAC, large sized car
	Labor costs	13.78 € per hour	BMEL 2015
	Labor time	5 sec./kg bread	Assumption

Scenario	Additional inputs	Amount	Source of amount used
(3) More frequent shopping	Paper bag	4.9 g/0.5 kg bread	Size of paper bag 25*15*10 cm, paper weight 35 g/m ² , one bag per 0.5 kg bread
	Passenger car transport from bakery to home	6 km distance	Assumption
	Transport costs	0.6 €/km	(both ways) for shopping 0.5 kg bread ADAC, medium sized car
(4) Freezing and consuming later	Time in freezer	14 days	Assumption
	Electricity consumption freezer (E _p)	E _p = 1.12 kWh	E _p = (E _d /SC)*(100%/u)*V _p *t (Nielsen et al. 2003); E _d : electricity consumption of freezer = 0.6 kWh/day, SC: storage capacity of freezer = 30 liters, U: degree of utilization of refrigerator = 50%, V _p =Volume of considered product = 2 liters, t: time of storage = 14 days
	Electricity costs	0.29 €/kWh	strompreise.de
	Purchasing costs freezer	0.00244 €/l*day)	15 years lifetime, daily degree of utilization 15 liters, purchasing costs 200 €

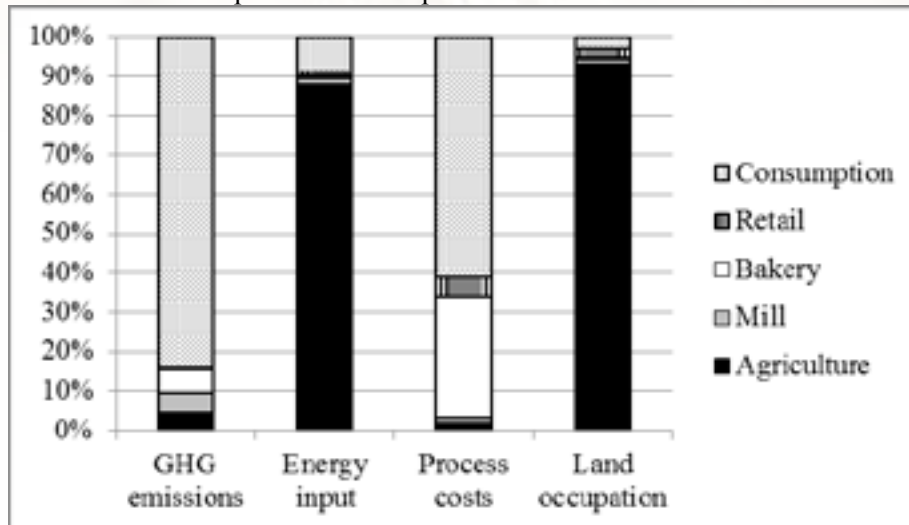
It is well-known, that costs and prices are very volatile over time. For this study, prices mainly refer to the years 2013 and 2014. Furthermore, the monetary values of by-products or waste have not been considered specifically. Apart from the additional costs from waste reduction scenarios, values are based on averages of German bread production. For the calculation, mass-allocation of environmental impacts was applied to the reference product and by-products. Production and treatment burdens of waste, including food waste, were allocated to the waste producing activity while the output from waste treatment e.g. compost from biowaste treatment becomes available burden free to the market (Ecoinvent 2016). Waste at the agricultural stage is not treated in a composting plant but directly left on the field. The biomass is seen as a by-product and no impacts for the degradation of the biomass were considered. Environmental impacts are not assigned to waste portions as those impacts are thoroughly assigned to the respective output product(s). This implies that compost from food waste composting is available burden free. The life cycle inventory data was analyzed according to the ReCiPe Midpoint impact categories regarding global warming potential and agricultural land referring to a 100 year horizon. In addition to that, the primary energy usage (including renewable and non-renewable energy) and the added monetary value of all processes are summed up, forming the additional impact categories “cumulative energy input” and “process costs”. The software openLCA was used for the calculation (Winter et al. 2015).

3. Results

The share of the different processes on the environmental impacts and costs are shown in Figure 2. Impacts from consumption dominate the GHG emissions (84%) as well as the process costs (61%). The costs of the total supply chain are also significantly influenced by the bakery (31%). Impacts from agriculture determine the land occupation and the energy input, whereby it should be noted that energy input includes the energy content (resp. calorific value) of the grain. The calculation of the whole life cycle of mixed grain bread, including waste portions at each life cycle stage, results in 2.51 kg CO₂eq per kg bread consumed. The largest portion of the emission occurs at consumption due to shopping by car (83% of overall emissions) and waste composting (1% of overall emissions). Besides consumption, 4% of emissions occur at the agricultural production, 5% at flour production, 6% during baking and 1% during selling. The cumulative energy input of the whole life cycle of bread results in a total of 18.04 MJ per kg bread consumed. Around 88% of the primary energy input occurs during agriculture due to biomass input for wheat and rye production. Milling, baking and selling each have an input of 1-2% primary energy, while the shopping trip at consumption accounts for the remaining 9% of energy input. The total life cycle costs amount to 6.69 €. Again, a large portion of the costs, namely 61%, occur during consumption as costs for fuel and the car. Also during baking the added value is comparatively large with 31% of the total costs. Finally, agricultural land amounts to 1.13 m²a per kg bread consumed. It mainly occurs during agricultural production but partly (around

7% of the total) also during other life cycle stages e.g. in the form of wood production for paper, buildings or energy generation.

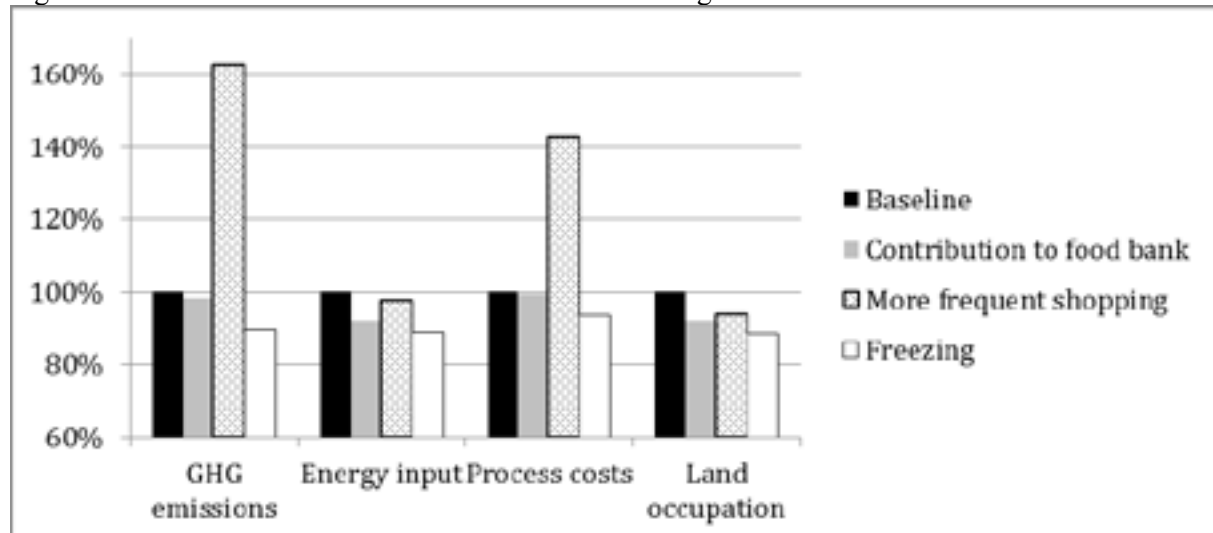
Figure 2: Share of the different processes on impacts and costs



Source: Own calculations.

When comparing the results from the baseline scenario with the three other scenarios under the assumption of a strong waste reduction effect the reduction scenarios score better than the baseline scenario in most categories as shown in Figure 3. However, the more frequent shopping scenario has outstanding more severe impacts with 63% more GHG emissions and 43% higher costs as compared to the baseline. The most effective scenario seems to be the freezing scenario with 90% of GHG emissions, 89% of primary energy use, 88% of land occupation and 94% of costs as compared to the baseline. Also the “contribution to food bank” scenario results in less impact than the baseline scenario, although the difference regarding GHG emissions and costs is marginal.

Figure 3: Results of waste reduction scenarios with a strong effect in relation to the baseline scenario



Source: Own calculations.

When assuming a weak waste reduction effect (Table 5), the impacts of the “frequent shopping scenario” become larger, as compared to the baseline, also regarding energy input while the agricultural land occupation is almost equal. The impacts of the freezing and food bank scenario remain almost all lower than the impacts in the baseline scenario but are largely approximate to the

baseline values. An exception here are the process costs of the food bank scenario, they are marginally higher than the baseline costs.

Table 5: Environmental impacts and costs of the life cycle of mixed grain bread

	Scenario	GHG in kg CO₂eq/FU	Energy input in MJ/FU	Process costs in €/FU	Land occupation in m²a/FU
	Baseline	2.51	18.04	6.69	1.13
<i>Strong effect</i>	Food bank	2.47	16.61	6.65	1.04
	Freezing	2.25	16.05	6.27	1.00
	Frequent shopping	4.08	17.62	9.55	1.06
<i>Weak effect</i>	Food bank	2.50	17.31	6.72	1.08
	Freezing	2.39	17.00	6.64	1.06
	Frequent shopping	4.33	18.66	10.11	1.12

4. Discussion

The results show, that waste reduction can also reduce environmental impacts, as it is widely promoted (e.g. Gruber et al. 2016, Eberle and Fels et al. 2016 and FAO 2013). But waste reduction actions do also have an impact on the environment and are also relevant to cost factors. In the presented scenarios, the impacts of freezing a part of the bread, as well as donating the bread to food banks are lower than the impacts of the bread production amount that is wasted in the baseline scenario. In addition, the costs of freezing and donating bread to food banks are lower than the costs of the percentage of bread that is wasted in the baseline scenario. However, the extent of the waste reduction effect is crucial. As seen in the modelling of a weak waste reduction effect in the freezing scenario, the waste reduction action should still be preferred to the baseline scenario even if it goes along with medium success. With the selected waste reduction scenarios of this model, higher life cycle costs occur together with partly higher or almost equal environmental as compared to the baseline (frequent shopping – strong/weak effect and food bank - weak effect). Although, this cannot be seen as a rule, high costs of reduction actions may be an indicator of high impact on resource use. At the consumption stage, the consumer is very likely to be unaware of the detailed costs for waste reduction as well as for the bread wasted. The food bank scenario shows that waste reduction can also be beneficial for the producer regarding costs.

It is not very productive to compare the results of the modeled life cycle with results of other studies dealing with the life cycle of bread (comp. Espinoza-Orias et al. 2011, Andersson and Ohlsson 1999) as results clearly differ due to different system borders, data used or assessment criteria. As it was not the goal of this study to find and establish new data for the life cycle assessment of bread but rather to assess the environmental impacts and costs of waste reduction measures. Therefore, comparison with existing studies is not even necessary and impacts of waste reduction actions have not been assessed, yet.

5. Conclusions

The assessment of environmental impacts and costs of waste reduction actions should be of high priority when it comes to the choice of food waste reduction measures. Measures should be selected according to their efficiency that is expressed as the relation between the abatement costs and resource reductions. It is important to give the consumer detailed information on costs of waste reduction actions as monetary savings can trigger waste reduction at consumption. Under the assumptions of the calculated example the option of freezing a part of the bread and donating the left-over bread to food banks are good approaches for food waste reduction. More frequent shopping is no option to save resources and costs when the shopping trip is done by car and for the single purpose of bread shopping. However, it can be reasonable in a coordinated action. A goal of future studies could be to assess further waste reduction actions that are finely graduated to determine tipping points of food waste reduction actions. Furthermore, the question of direct costs and externalities of resource use should be included in the calculations, as well as estimates about the effect on the national economy.

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239. The embedded resource and environmental burden of deliberately wasted lettuce

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Abstract

Consumers are judged by retailers to expect high in-stock levels of fresh produce. However, since retail forecasting and crop programming are not exact, this expectation is met through growers producing more than anticipated demand. This buffering capacity ensures a consistent flow of quality product through the supply chain. But the downside can be high levels of in-field waste of saleable product when supply and demand do not balance. This paper presents selected results from a WRAP funded project which sought to assess the levels of in-field lettuce waste in the UK. Greenhouse gas emissions, eutrophication and acidification are estimated for two waste scenarios.

Data collected from 14 growers initially suggested that typical in-field waste was between 10 and 20% although amounts varied considerably with variety and market conditions. Our assessment that 19% of mature crop was left in the field agreed with grower values but we also observed that another 19% was lost through crop trimming. In total, we suggest that approximately 38% of mature lettuce does not enter the food chain.

UK marketed production of lettuce in 2014 was 124,000 tonnes which at a 38% wastage level suggests that 200,000 tonnes were produced with 76,000 tonnes subsequently wasted. This represents a huge waste of food and also a huge waste of resources. We estimate the carbon footprint of the wasted produce to be over 14,000 tonnes CO₂e with 18 million litres of water used for no purpose. The opportunity costs are also high since the 3,800 hectares used to produce the lettuce that was subsequently wasted could have been used more productively.

The results have implications for both supply chain management and regional environmental planning.

1. Introduction

Consumers are judged by retailers to expect high in-stock levels of fresh produce. However, since retail forecasting and crop programming is not an exact science, this expectation is achieved by growers producing more than anticipated demand. Over-production provides the buffering capacity to ensure a consistent flow of quality product through the supply chain. But the downside can be high levels of in-field waste of saleable product when supply and demand do not balance.

Lettuce has a short shelf and storage life and a continuous supply is required to meet consumer demand. Unfortunately, consumer demand is highly influenced by the prevailing weather and can vary considerably from week to week. Growers build this variation into their production programme by over-planting by up to 10 or 15% since it is economically advantageous to waste excess lettuce rather than to under-deliver to a customer. The actual level of waste generated is dependent on a number of different parameters but in a good production year, when growing conditions are good and pest and disease pressure is low, it is possible that the all the over-planted area will be wasted. In addition, there will be wastage associated with plant trimmings. In a poor year, the opposite might be true and the over-planted area will be required to make up for shortages elsewhere due to poor quality produce.

In-field or on-farm waste at crop maturity falls into three categories: whole crop wastage due to over-planting or a change in demand; wastage from trimming due to cosmetic damage or to meet a specific size specification; and losses due to pests and disease. Unharvested whole crop and trimmings are almost always left in the field and will be incorporated into the soil at the next cultivation. This process allows the recovery of some organic matter and plant nutrients. Lettuce wasted through customer rejections is either returned to the farm of origin to be composted or returned to the field, or sent directly to landfill. Typically, where lettuce has been packaged it is not economically viable to unpack it and it is cheaper and more time efficient to send it to landfill.

There is only limited evidence on the amount of lettuce that is wasted. Work by WRAP in the UK found that field wastage was in the range 4 to 30% with weather being the biggest cause of variation (Terry et al., 2011). In the United States, the Natural Resources Defense Council (NRDC) found that wastage in head lettuce in the US (termed more specifically as ‘shrink’) could also be up to 14% (Milepost Consulting, 2012). Strid et al (2014) followed what they termed the ‘crop loss flow’ of Swedish iceberg lettuce from field to retail shelf and reported that at the farm level (after produce had reached harvestable stage) 14% of lettuce heads were wasted.

In all cases of wastage, the resource used to grow the crop is also wasted. This includes the area of land on which the crop is grown (this can be both an economic and opportunity loss to the grower and also a loss of overall food production), the water required in crop production, and the economic loss that occurs from wasted labour, energy (both direct and indirect), fertilizer and pesticides. The environmental impact of this wastage translates into extra greenhouse gas emissions, possibly additional water stress to the surrounding natural environment, the loss of nitrates and phosphates leading to potential eutrophication and a build-up of pesticide residues in soil and water if the crop is produced conventionally.

The environmental impact of food wastage was examined by the FAO who estimated that the global carbon footprint of wasted food was 3.3 Gtonnes CO₂ which if considered in country terms would be the third biggest emitter globally after the USA and China (FAO, 2013). They also suggested that the volume of water required to produce subsequently wasted food was the equivalent of three Lake Geneva’s and that globally 1.4 billion hectares was required (this is one-third of all agricultural land).

The objective of this paper is to present selected results from a WRAP funded project which sought to assess the levels of in-field lettuce waste in the UK and to estimate the environmental burden that occurs as a result of that wastage. The burden will be expressed in terms of the opportunity cost and wasted resources and as an estimate of the greenhouse gas emissions and eutrophication associated with wasted produce.

2. Methods

Activity data was collected by interview from 14 growers. Although this is quite a small sample size, being only 14% of all UK lettuce growers, we estimate that these growers manage 3,273 ha which represents approximately 54% of the UK lettuce area. Therefore we are confident that it provides a robust data set. Interviews took place between October 2015 and January 2016. The boundary of the investigation was the farm gate which included field production and associated pack house activities. Very few growers collect data on wastage so the levels of waste reported in this study are derived from direct grower and researcher observation, the area planted, the quantity sold and any retailer returns.

Life cycle assessment was not undertaken on the individual data sets. Instead, data on crop waste was combined with secondary data on the environmental impacts associated with lettuce production (Lillywhite et al., 2007; Lillywhite et al., 2009). This approach aligns with methods used in other WRAP reports to quantify the sector-level impacts of waste and provides a reasonable estimate of associated impacts (WRAP, 2013). Activity data was converted into an assessment of impact using conversion factors taken from either the academic literature or research reports. The environmental

impact of lettuce production is sparsely represented in the academic literature, however, there is enough evidence to develop factors to convert the amount of lettuce wastage into both resource use and environmental impact (Table 1).

Table 1. Conversion factors for the estimation of resource use and environmental impact

Parameter	Conversion factor	Source
Land use	20 t/ha	Defra Basic Horticultural Statistics 2014
Water footprint	237,000 litres/t	WaterStat (Water Footprint Network)
Energy	6 GJ/t	Fisher et al. (2013)
Carbon footprint	189 kg CO ₂ e/t	Lillywhite (2009)
Eutrophication potential	0.71 PO ₄ e/t	Lillywhite (2009)

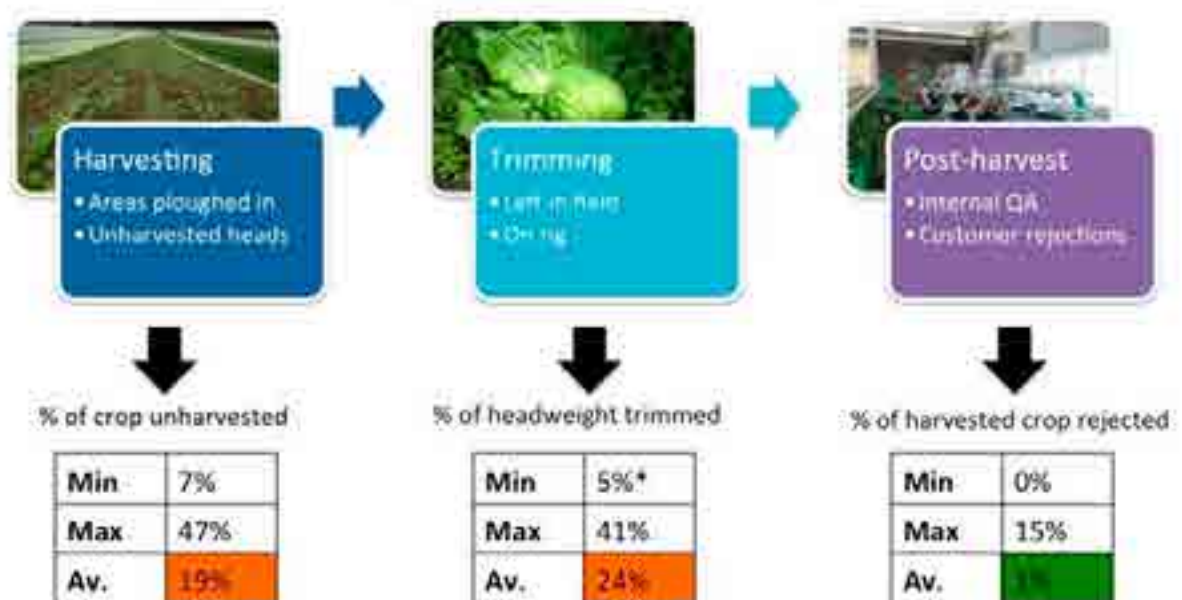
3. Results

The results show that on average 38% of the biomass of the mature lettuce crop is wasted. Approximately half of the wastage is due to mature heads not being harvested and the other half is due to trimming of the heads in the field.

Harvesting losses can be divided into two categories: whole areas of crop being ploughed back into the soil because they are surplus to requirements, and selective harvesting. The first category includes grower over-planting and crop gluts when weather conditions lead to sequentially planted crops maturing at the same time and demand is low. Since there is no secondary market for lettuce, any crop that cannot be picked during its maturity window will be destroyed by returning it to the soil. Selective harvesting occurs during gluts or when demand is less than the programmed volume and pickers only harvest the best crop to meet customer specifications. As the cost of harvesting is the major cost involved in production, pickers do not typically return to the same area that has been selectively picked and any crop remaining will be destroyed.

Trimming losses occur in the field for a number of reasons. The removal of outer leaves for cosmetic reasons is normal as head lettuce is packed in the field and they are typically not graded again before dispatch to the customer. Cosmetic reasons include water splash marks, bacterial discolouration and pest damage. Trimming for cosmetic damage does not generate large volumes of waste. The second trimming loss is trimming to a customer's specified size which at different stages of maturity can easily lead to half the plant being left in the field. Often pickers will only harvest the top half of the plant and leave the rest growing. Depending on variety, this can generate very large trimming losses.

On-farm pack house and customer rejections are less common and only result in small amounts of



*One outlier was removed

Averages were weighted by production size of grower.

wastage. Waste generated on-farm will be returned to the soil. Waste generated off-farm is typically sent straight to landfill because it is not economic to return it to the grower. The results are shown diagrammatically in Figure 1.

Domestic marketed production of lettuce in the UK in 2014 was 124,000 tonnes (Defra 2014). This value takes into account the crop already wasted so it is estimated that at an average wastage rate of 38%, field production was originally 200,000 tonnes and therefore 76,000 tonnes was wasted. Even at a minimum wastage rate of 12%, field production would be 140,100 with 16,100 being wasted. This is a large volume of material and constitutes a considerable loss of resources and potentially a sizeable environmental impact. The minimum and average scenarios are summarised in Table 2 which estimates the loss of resource and potential impact for wastage at the minimum wastage (12% of production) and average wastage (38% of production). No data is presented at the maximum wastage (97% of production) as that is an unreasonable scenario.

Table 2. Loss of resources and potential environmental impact at different levels of wastage

Waste level	Min (12%)	Average (38%)
Wastage (t)	16,100	76,000
Land use (ha)	805	3,800
Water footprint (million litres)	3,816	18,012
Energy (GJ)	96,600	456,000
Carbon footprint (t CO ₂ e)	3,043	14,364
Eutrophication potential (PO ₄ e)	11,431	53,960

4. Discussion

It is difficult to be precise on the amount of lettuce being wasted as it varies by grower, by type and by year. In 2014, Defra estimated that approximately 6,000 ha were planted to lettuce and that 124,000 tonnes per marketed from that area (Defra, 2014). At a simple level, this suggests that each hectare yielded 21 t/ha of marketable crop, but this does not take into account the fact that some of this land will be doubled cropped which suggests that the yield per hectare might be lower than 21 t/ha. However, research evidence has shown that the field yield of lettuce is between 35 and 45 t/ha depending on variety, suggesting that the gap between field and marketed yield is probably in the region of 19 t/ha, or 48%. This is higher than the 38% reported in this study but suggests that the results reported here are comparable (Lillywhite, 2009; Defra, 2010). This collaboration suggests the results from this study are probably robust and representative of the industry as a whole.

Wastage of edible food

Although the lettuce sector is relatively small and might not generate high levels of waste in comparison to other crop sectors, it can act to highlight supply chain inefficiencies and the real resource and environmental burden issues associated with crop wastage. The major concern must be the overall level of waste generated at the production end of the supply chain. Any agricultural sector which on average wastes more than a third of its output cannot be considered efficient and inefficiencies of this scale almost always imposes a burden somewhere in the supply chain. In this case, ensuring high in-stock levels on retailer's shelves by buffering through over-production is an economic and opportunity loss to growers.

To reduce wastage in the lettuce sector, and the same argument applies to some other fresh produce sectors, therefore requires a new strategy from the supply chain. This could focus on three areas: firstly, improved forecasting and programming which has the potential to mitigate some of the losses;

secondly, a relaxation of size and quality specifications, in specific circumstances, which might reduce in-field trimmings; and thirdly, a new approach to high in-stock levels which might reduce the current level of grower over-planting.

The finding that on average 38% of lettuce is wasted at farm level is probably symptomatic of much of the fresh produce industry. Zero wastage is unrealistic as pest and disease damage cannot be completely eliminated and cosmetic trimming will always be required to obtain a saleable product but reducing waste levels to the practical minimum would benefit all supply chain stakeholders. Food campaigners have already initiated this process in the UK but this study illustrates that there is still a lot to do.

Growers undoubtedly incur a financial burden in the current situation but the real issue must be that edible food is either being returned to the soil or sent to landfill at a time when food security has become a political issue.

Opportunity cost

The opportunity cost at the average wastage rate is 3,800 ha and 18 million litres of water. In a time where food production and supply are high on the political agenda, the loss of this capacity is concerning. This is especially true of lettuce production since it is typically high grade land with installed irrigation capacity with the ability to grow other high value and nutritious crops.

The resources used to produce food that is subsequently wasted could either be put to an alternative use or not used in the first place. Wasted food is a good example of a double impact. In the case of average wastage, the result suggest that not only were 76,000 tonnes of lettuce not available for human consumption but that producing crop that was subsequently wasted prevented something else being grown. The area occupied by the wasted crop could have produced more than 30,000 tonnes of wheat which would have made a positive contribution to UK growers, consumers and the economy.

Resource use and environmental burden

The same argument that applies to opportunity cost also applies to the resources. The energy, fertilizers and pesticides applied to wasted crop cannot be recovered although there is some fertilizing value and soil quality improvements to be obtained from returning it to the soil.

In many ways, the environmental burden, although of concern, is possibly the least important of the three aspects. It is irrational to generate environmental burden in crop production and then to fail to eat the food. In this sense, wasted food imposes a further burden on society through greater energy use, more greenhouse gas emissions and potential environmental damage to the natural environment.

5. Conclusions

This study reveals that wastage in the lettuce sector is chronic and that its generation has become an accepted part of doing business. This is not because lettuce growers are less professional or less skilled than other growers but because lettuce is a short-shelf life product within a hugely competitive market sector. A significant proportion of wastage is due to a combination of high supply/demand variability and concerns about losing business in a mature and competitive market. Put simply: the potential benefits from reducing wastage are intuitively understood by growers to be outweighed by the risk of losing a customer and their continuing business. A supply chain revolution is required to change the basic relationship between grower, retailer and consumer to allow as much as possible of the food that is grown to be sold and consumed because it is difficult to see how society can continue to tolerate such high levels of food waste in a world with both a growing population and growing environmental problem.

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173. Environmental Impact of Food Losses from Agriculture to Consumption in Switzerland

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ABSTRACT

Food losses and waste (FW) cause environmental impacts, which can be reduced with measures to avoid FW and to optimize the methods of treatment. In order to prioritize these measures, we analysed the environmental impacts of FW with life cycle assessment (LCA) at the different stages of the food value chain (FVC) and for different products in Switzerland, considering agricultural production, transportation, storage, processing, cooking, and different methods of FW treatment. Environmental impacts were quantified in terms of climate change impacts, biodiversity impacts due to land use, and with the Swiss method of ecological scarcity. The results show that reducing FW is more effective than optimizing the methods of treatment and that FW at the end of the FVC is both quantitatively and environmentally more relevant than at the beginning of the FVC. Most of the impacts on biodiversity take place in the countries where the food exported to Switzerland is produced. As a conclusion, measures to reduce FW should primarily focus on final consumption, especially regarding fresh products with short shelf live as well as products imported from other countries.

Keywords: food waste, life cycle assessment (LCA), mass flow analysis (MFA), food value chain (FVC), food supply chain, food waste treatment

1. Introduction

Twenty to thirty percent of the environmental impacts of consumption are caused by food consumption (Tukker et al., 2006). A key element to make our food system more efficient and sustainable is the reduction of food losses and waste (FW) across the entire food value chain (FVC). However, for the implementation of measures against FW it is important to know which losses are environmentally most relevant. In recent years several countries have done first quantifications of FW at different levels of the FVC (Quested and Johnson, 2009, Kranert et al., 2012, Schneider et al., 2012, Hanssen and Møller, 2013, Hamilton et al., 2015). Nevertheless, since the definitions of avoidable FW, the methodologies, and the system boundaries are varying substantially, it is difficult to compare countries, regions, and changes over time.

In this paper FW refers to food which is originally produced for human consumption but then directed to a non-food use or waste disposal (e.g. feed for animals, biomass input to a digestion plant, disposal in a municipal solid waste incinerator, composting). FW is defined “avoidable” if it can be avoided by best practice methods of efficient supply chains, by a reduction of cosmetic standards for products such as fruits and vegetables (e.g. using all forms and sizes of potatoes for human consumption), and by applying appropriate methods of preparation to use all potentially edible parts of the products (e.g. stem of broccoli and skin of apples). This definition is consistent with Norwegian food waste studies (Hamilton et al., 2015).

Scherhauser et al. (2015) estimate FW related emissions at EU level at 16% to 22% of the total emissions of consumed food, production and consumption being the most important contributors of the stages in the FVC. They conclude that environmental data on a product category level and data on the recovery and disposal of FW, especially for the valorisation as animal feed, represents a relevant data gap for further research.

An analysis of the environmental impacts of FW over the whole FVC, distinguishing by food categories, stages of the FVC, and including FW treatment is still lacking in literature. Therefore, the present paper aims to update the mass flow analysis by Beretta et al. (2013) and to complement it with an environmental assessment. This helps to identify the most relevant FW flows, as a basis to develop effective measures to reduce the future impact of FW.

2. Methods

In order to quantify the environmental impacts of FW and to compare them with the impacts of food consumption, a model of the whole Swiss FVC was created, covering agricultural production, trade, processing, retail, the food service sector, and private households. The first part of the model consists of a mass and energy flow analysis of all the food that is produced in Switzerland, excluding exports, and all the net imports of food (Fig. 1). Thus, the model covers all the food and FW flows that are related to Swiss food consumption. This approach covers FW in foreign agricultural production of imported products and excludes Swiss FW related to exports. The mass flow analysis (MFA) of Beretta et al. (2013) was updated, especially integrating recent literature relating to FW in agricultural production of exotic fruits, in the FVC of potatoes, in the retail and the food service sector, and relating to the use of FW for treatment.

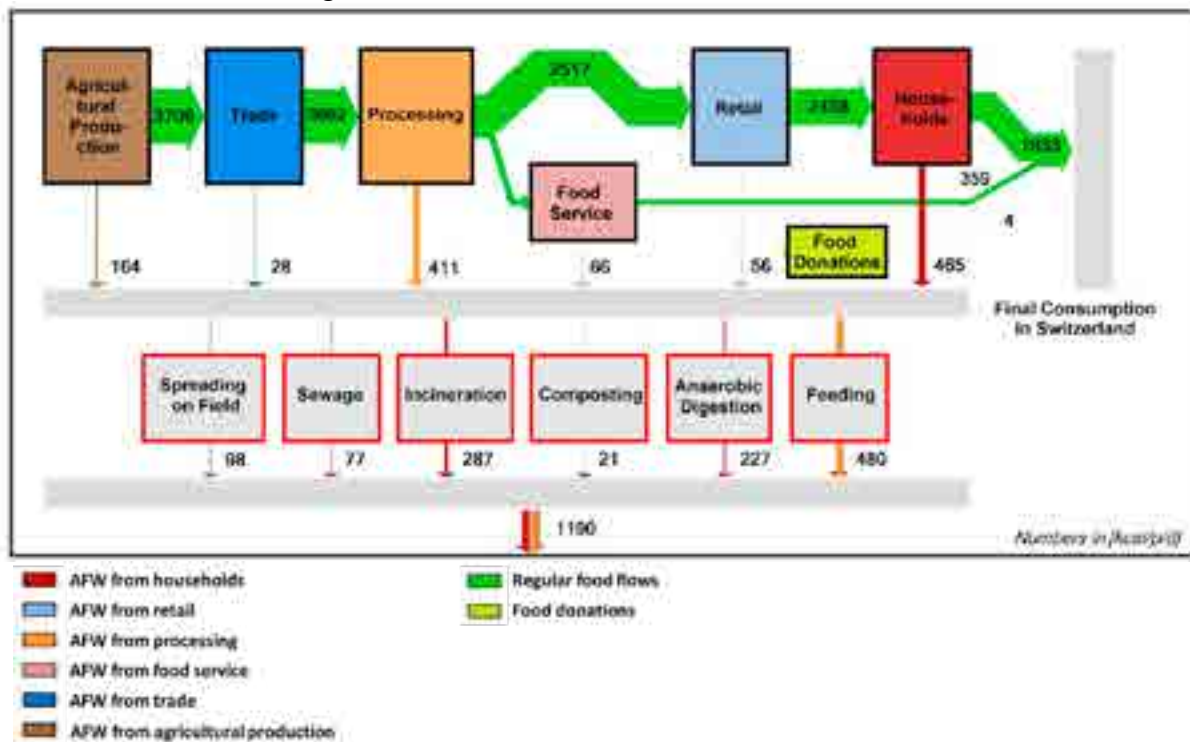


Figure 1: Energy flow analysis of Swiss food consumption and losses. The green arrows show the regular food flows from agricultural production through the stages of the food value chain to final consumption in households and food service institutions. The vertical arrows show the avoidable food waste flows (AFW) from the individual stages of the FVC to different treatment methods (spreading on field, sewage, incineration, composting, anaerobic digestion, feeding). The numbers refer to metabolisable energy, expressed in kilocalories per person per day.

The second part of the model links the mass flows of food and FW with the environmental impacts associated with these flows, i.e. with the processes of agricultural production, transport, processing, preparation and cooking, and with the treatment of FW. The life cycle assessment (LCA) is done with the avoided burden approach and for the impact category

climate change 100a (IPCC, 2013), for a sum of environmental impacts with the aggregated method ecological scarcity 2013 (Jungbluth et al., 2011), and for biodiversity. The regionalised biodiversity assessment of land impacts is based on Chaudhary et al. (2015), and Chaudhary et al. (2016). Global biodiversity loss considers endemic species richness. The corresponding characterisation factors are available for 124 crops and 158 countries (Chaudhary et al., 2016, Scherer and Pfister, 2016). They are multiplied with net production and imports from each country to Switzerland (SBV, 2015, Scherer and Pfister, 2016) and aggregated to the 33 food categories modelled in this study. We assume that the wasted products originate from the same mix of countries as the consumed products.

We used the software Simapro 8.23 (Pré, 2016) and mainly based our inventories on the LCA databases ecoinvent 3.2 “allocation recycled content” (ecoinvent, 2016), World Food LCA Database 3.0 (Bengoa et al., 2015), Agri Footprint 2015 (www.agri-footprint.com), AGRIBALYSE v1.2 (www.ademe.fr), a food inventory collaboration with ZHAW and Eaternity (Eymann et al., 2014, Kreuzer et al., 2014), and data from the SFOE (Dinkel et al., 2012). The World Food LCA Database is linked to the ecoinvent 2 database, which may provide some inconsistency. However, the differences between ecoinvent v2 and v3.2 “recycled content” are irrelevant for most agricultural products (Steubing et al., 2016). Some datasets are used in their original version, some are modified if more appropriate data related to Swiss consumption was available. Generally, if datasets from different countries are available for specific food categories, they are weighted based on import shares from FAOSTAT (2015). If datasets for different products within a food category are available, they are weighted according to Swiss consumption of the corresponding products according to SBV (2013). For canned fruits, the composition of the consumption mix is based on the sales from a Swiss supermarket chain.

3. Results and discussion

Overview of the life cycle impacts of Swiss food consumption and losses

The yearly, agricultural production of an average Swiss consumer’s food basket is illustrated in Fig. 2 and causes about 4 million ecopoints/cap (20% of total Swiss consumption). The second largest impact fraction is caused by households with 340’000 ecopoints/cap or between 8 and 9% relative to agricultural production. Trade, processing, and food services cause less than 5% each, and retail about 2% compared to agriculture. The net environmental benefits from the treatment of avoidable and unavoidable losses are between 60’000 and 70’000 ecopoints/cap, each. The main benefits of FW treatment are due to credits from feed substitution, which are higher than the credits for fertilizer, electricity, and heat substitution from FW treatment with anaerobic digestion, composting, or incineration. The total impacts of consumed food, including consumption in households and food services and food donations, amount to 3.6 million ecopoints/cap, the impacts of FW to 1.3 million ecopoints/cap. All in all, Swiss food consumption causes about 25% of the impacts of total Swiss consumption, avoidable FW about 6-7%. Food donation institutions save food with supply chain impacts of 5’400 ecopoints/cap. For the distribution they emit 200 ecopoints/cap, which is about 4% of the saved emissions. (Fig. 2).

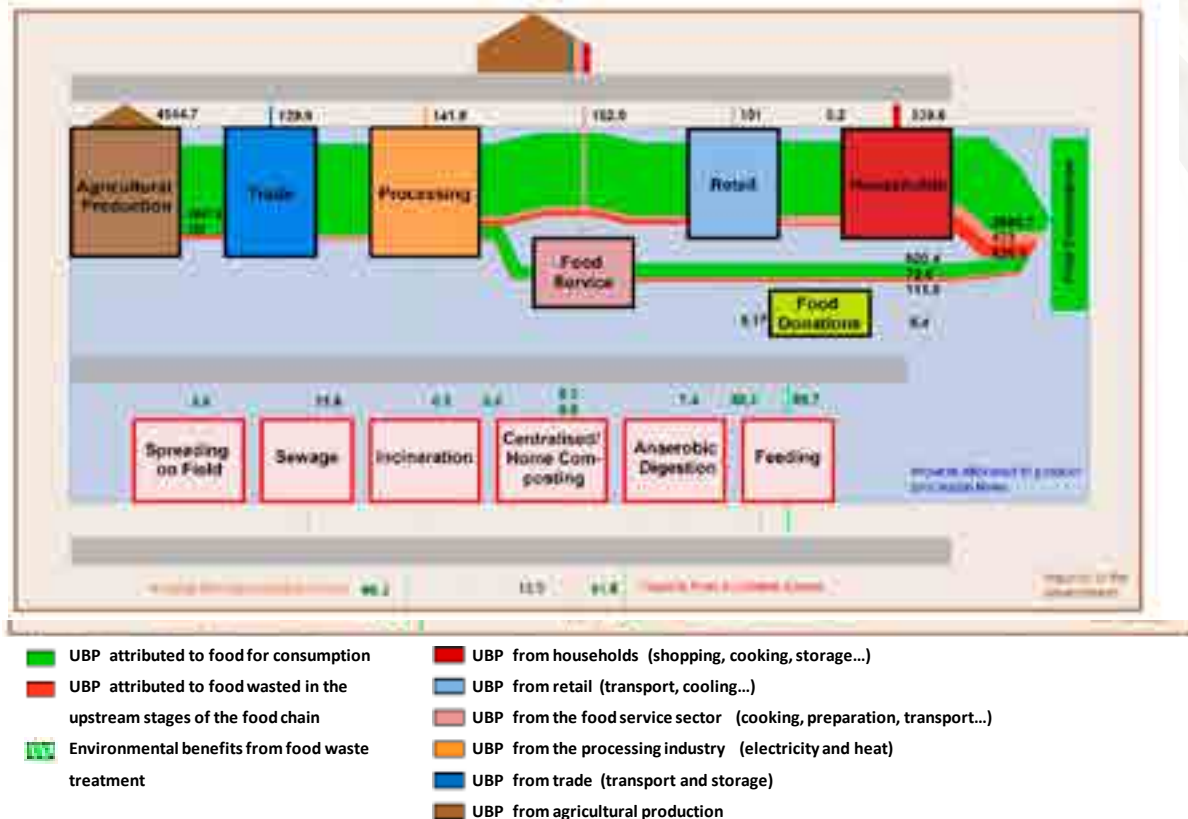


Figure 2: Environmental impacts of Swiss food consumption, including the production and treatment of FW, with the Swiss method *ecological scarcity 2013* (in 1'000 ecopoints/cap) for the year 2012. The flows on the top show the impacts arising at the various stages of the FVC, the flows at the bottom the net environmental benefits of FW treatment, including the substitution of resources and energy (forage, fertilizer, peat, electricity, heat). The horizontal flows show the cumulated impacts of the upstream processes of the FVC. The attribution to consumption (green) and waste (red) is based on the metabolisable energy content of the food. The numbers of the flows are rounded to one decimal digit, but the uncertainty can be higher.

The total environmental impacts of the FVC and of FW treatment that are allocated to avoidable FW (red flows in Fig. 2) are displayed in the pie chart in figure 3, distinguishing the stages of the FVC where the food is lost. Nearly half of the impacts are caused by households, not only because most of the food is lost at this stage of the FVC quantitatively (Fig. 1), but also because of the accumulation of environmental impacts across the FVC. The losses in food service institutions are less relevant because only a relatively small share of food consumption takes place out of home (about 15% estimated). Agricultural production and processing are also relevant with 14 and 20% of the impacts. For the interpretation, however, it is important to note that these numbers refer to the stages of the FVC, where the food is wasted, but not necessarily to the stages where the FW is caused. For example, losses in agriculture are mainly caused by standards defined by the food industry, based on consumers' preferences. Thus, in order to reduce the losses in agriculture, other actors of the FVC than farmers may be most relevant.

Differences between food categories

Figure 3 shows the net environmental impacts in terms of ecopoints per person allocated to the top ten food categories and differentiating the stages of the FVC where the losses arise. Breads and pastry losses cause the highest impacts, followed by fresh vegetables, cheese, pasta, and beef. Household FW is relevant in all food categories, whereas processing losses dominate in the food chain of cereals (mainly declassified cereals and bran used for feeding) and dairy products (mainly whey and buttermilk). In the case of fresh vegetables, losses in agriculture are similarly relevant to household losses.

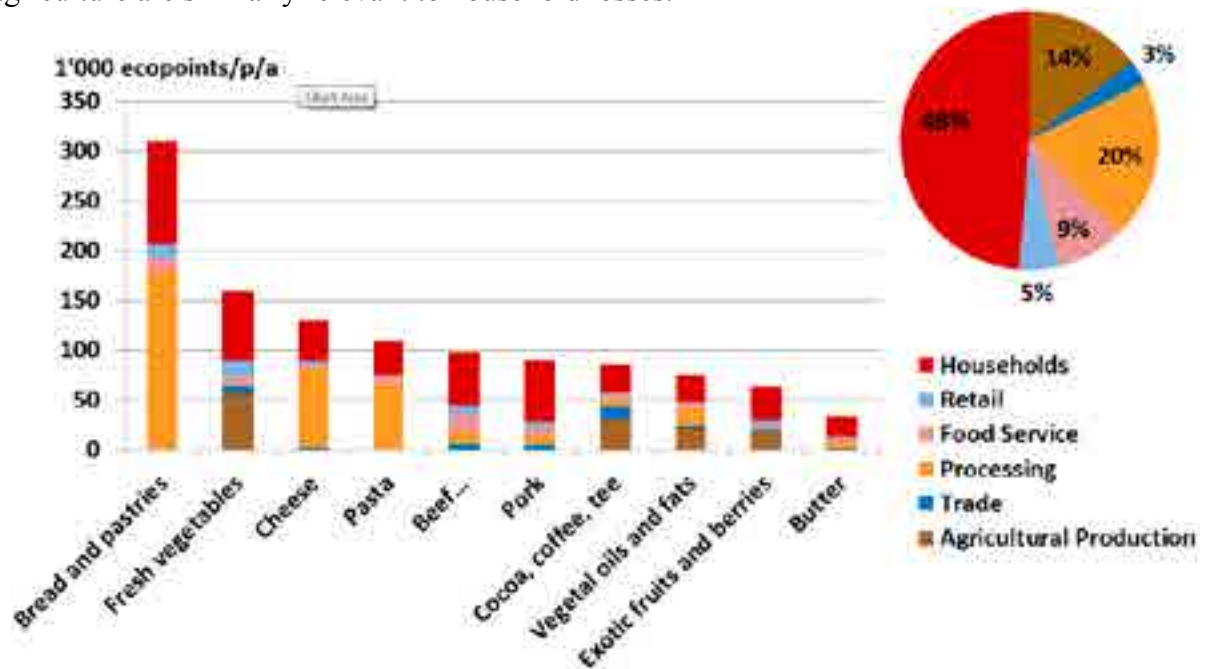


Figure 3: Net environmental impacts (*ecological scarcity 2013*) caused by the production, supply, and treatment of food that is wasted at the different stages of the FVC, considering the benefits from FW treatment (electricity, heat, fertilizer, feed substitution etc.). The results are shown for the top ten food categories... The pie chart shows the relative contribution of each stage of the FVC for total FW.

In order to prioritize specific FW reduction strategies it is not only important to know which FW flows are environmentally most relevant in total for Switzerland, but also which food categories are most relevant per kg of FW. This is shown in Fig. 4 for the top ten food categories and three stages of the FVC. The food categories are ranked by the environmental impacts of FW arising in households. In most cases the impacts grow along the FVC because of the accumulation of impacts; however, in some cases the benefits from FW treatment (mainly feed substitution) are larger for FW from processing than for agriculture, overcompensating the impacts of the additional transport, storage, etc. For processed products with higher calorific content of the final product the impacts per kg can increase substantially after processing because of the energy based allocation (e.g. vegetal oils and fats). Furthermore, figure 4 shows that in terms of ecopoints the type of food is more important than the stage of the FVC where it is lost.

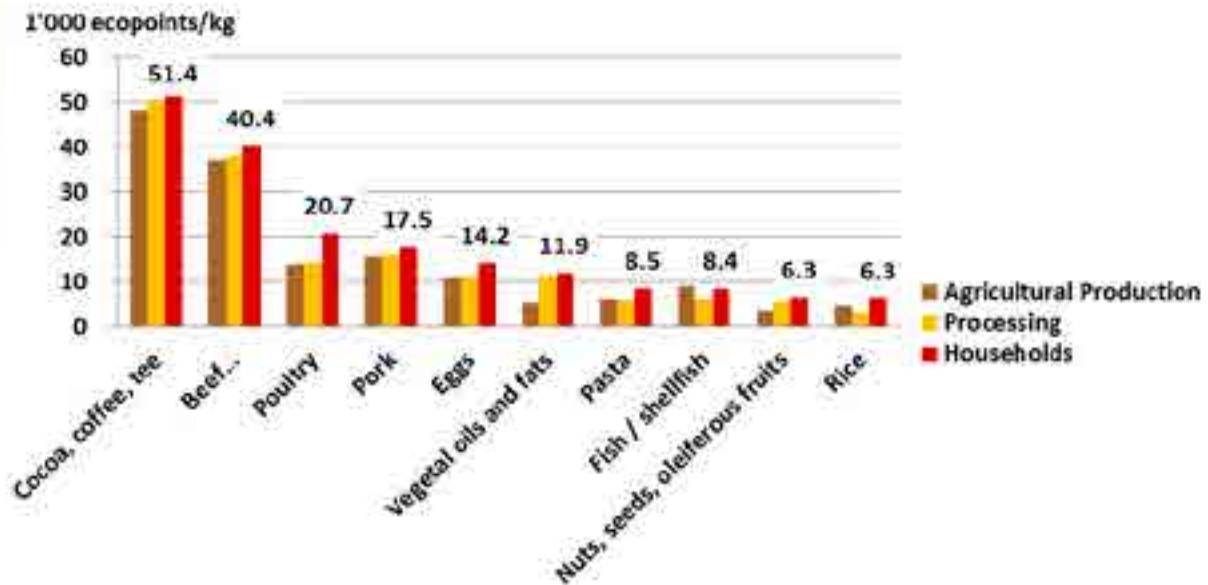


Figure 4: Net environmental impacts (*ecological scarcity 2013*) caused by the production, supply, and treatment of food that is wasted at three different stages of the FVC (agriculture, processing, households), per kg of FW.

The food category which causes the highest impacts of FW of one average Swiss person on global biodiversity is cocoa and coffee, even considering the relatively low amounts of FW. Beef is in the second place, followed by breads and pastries, fresh vegetables, vegetal oils and fats, nuts, seeds, and exotic fruits. A reason may be that many of these products (including imported feed for beef production) are produced in areas with high occurrence of endemic species; however, the impacts on regional biodiversity may be highest for other food categories.

Similarly to the environmental impacts in terms of ecopoints (Fig. 3), household FW contributes most to biodiversity impacts (Fig. 5). However, the share of the consumption stages (households and food services) is lower in terms of biodiversity (50% of total impacts) than in terms of ecopoints (58%), because biodiversity impacts from land use are mainly arising in agricultural production, whereas other environmental impacts (e.g. GHG emissions, use of mineral resources etc.) are also caused by activities along the FVC.

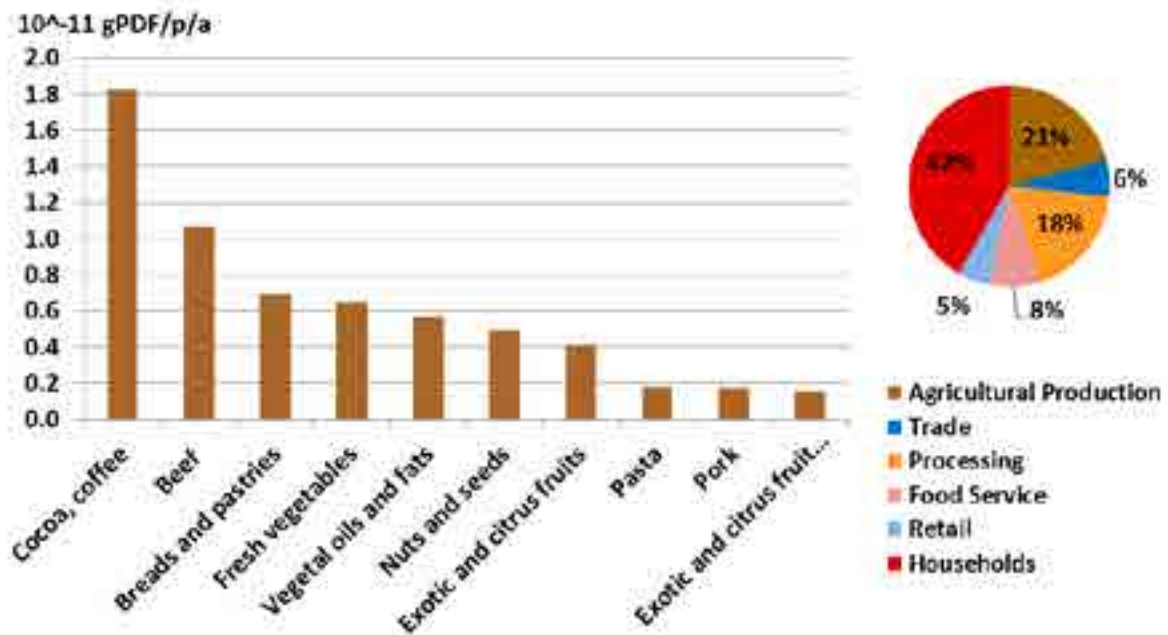


Figure 5: Global biodiversity impacts from land use allocated to FW per person and year for the ten most relevant food categories. The pie chart shows the contribution of each stage of the FVC for the whole food basket.

If we compare the total greenhouse gases (GHG) caused by FW in Switzerland (0.5 t CO₂-eq/cap/a) with other sectors, they are in a similar range as the direct CO₂-emissions caused by leisure mobility (0.7 t CO₂-eq/cap/a, 54% of total private mobility by car, BFS (2016)). So, the theoretical potential GHG savings of FW prevention, taking into consideration the present benefits of alternative uses of FW, are comparable to replacing all car rides in leisure time with means of transport that are climate neutral in their use phase. The potential GHG savings from FW prevention in households and food services alone (0.3 t CO₂-eq/cap/a) are higher than half of the total impacts from FW.

4. Interpretation

The food that is wasted across the entire supply chain of Swiss food consumption causes major environmental impacts. The main impacts are generated in agricultural production (about 80% of the total emissions of 5 million ecopoints/cap). The impacts of the FVC are also relevant; for some products with relatively low impacts from production, e.g. potatoes, they are even dominant. The environmental relevance of FW increases along the FVC, with the implication that FW at the end of the FVC is environmentally more relevant than at the beginning. This effect, in combination with the high quantities of FW at household level, makes household FW an environmental hotspot. Cocoa, coffee, and meat are the most important food categories in terms of ecopoints per kg of FW, bread and pastry losses, fresh vegetable waste, and whey from cheese production in terms of ecopoints per average consumer. The largest impacts on global biodiversity are caused by cocoa and coffee production in tropical regions because of the high endemic species richness in the original habitat of these areas. However, other food categories may be relevant if regional biodiversity and other impact categories are considered.

The treatment of FW leads to net environmental benefits, but compared to the impacts of the previous FVC they are low (5% of UBP). The environmentally best treatment option for products with high nutritional value (e.g. bread, cereals, dairy...) is feeding to livestock.

Finally, this study provides a scientific basis to develop measures for FW prevention in Switzerland. For detailed questions and the practical implementation of measures more research is needed; nevertheless, this study shows that there is a major potential to create

environmental benefits and identifies hotspots to prioritize interventions for FW reduction, for example with consumer sensibilisation, cooking classes in schools, and the valorisation of whey, buttermilk, sub-standard vegetables and other food losses into new, marketable food products.

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109. Mapping food waste in Norway – Methodology and results from the ForMat-project

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Abstract

Through the ForMat-project, edible food waste separated into nine main categories of food, split in twenty-one different product categories and four steps of the food-value-chain (food industry, wholesale, retail and households) in Norway has been documented over a six-year period (2010-2015). The main aim of the project has been to contribute to food waste prevention in Norway with a goal to reduce edible food waste with 25 % at the end of 2015, with reference to 2010, as well as gaining knowledge about where and why edible food waste is generated.

This paper seeks to present the results of the ForMat-project, as well as the methodology and reporting systems used to obtain the data, which in turn can be used in LCA studies to assess environmental impacts of food waste prevention and treatment. Data has been gathered through voluntary reporting and waste-composition analysis, and national statistics have been compiled by upscaling the data, using share of turn-over/production and national waste statistics. The project has retrieved insight about the generation, hot-spots, composition and amounts of edible food waste in Norway, as well as knowledge about root-causes and challenges in how to document real reductions in edible food waste. The project shows that a total of 355 thousand tons of edible food waste was generated in 2015 within the four steps of the food chain. This equals a reduction of 7 % measured in tons since 2010. Although edible food waste is reduced by 7 %, the environmental impacts has only been reduced by 3 %. The data obtained through the ForMat-project can help understand the environmental impacts of food waste, and connecting the food waste statistics to LCA-data shows that a detailed material flow analysis is crucial knowledge for food waste prevention, assuming that the ultimate goal is to reduce environmental impact. There are some limitations and uncertainties related to the data and methodology, however, the ForMat-project is a unique compilation of food waste statistics, that provides valuable information on aspects such as what, why and where related to food waste generation in Norway.

Keywords:

- Food value chains and LCA
- Extent and reasons
- Status and trends

Introduction

During the past decade, food waste has become an increasingly hot topic, most recently in light of EUs circular economy package. The main goal of food waste prevention is to reduce environmental impacts, such as green-house-gas (GHG)-emissions and resource/land/water use. In order to assess these environmental effects (e.g. through LCA studies), the first step is to conduct a material flow analysis. Many national projects have been carried out in Europe to determine both the amounts and the composition of edible food waste, as well as where in the food value chain edible food waste is generated, e.g. in UK by WRAP (Ventour 2008) and in Norway (Hanssen and Schakenda 2010,

Stensgård & Hanssen 2015). International projects have been established by the EU (FUSIONS project (Östergren et al. 2014)) as well as globally by FAO (Gustavsson et al. 2011), OECD and UNEP. Additionally, several national food waste surveys have been carried out over the last years, with quite different results regarding kg of edible food waste per capita, varying between 23 and 55 kg per capita from households (Hanssen et al. 2016). An important challenge with comparisons between national food waste statistics is that the methodological basis for the studies is not necessarily the same (Hanssen et al. 2013a, Møller et al. 2014), as definitions of edible food waste might differ between studies as well as methodologies for data gathering (Møller et al 2014). There is also considerable variation within the environmental impact assessments of food waste prevention. Schot & Cánova (2015) showed that avoided emissions can vary between 0.8 to 4.4kg CO₂-equivalents (eq.) / kg prevented food waste, depending on differences in system boundaries, food waste composition, and assumptions related to the avoided food supply system. Essentially, in order to know the environmental impacts of food waste and food waste prevention, the food waste must be thoroughly mapped.

Methodology and data gathering

Food waste in the ForMat-project is defined as “*all food that could or should have been eaten by humans, but which for some reason is not*” (Stensgård & Hanssen, 2015). This definition is in compliance with the definition used in the Interim Sector Agreement with the Norwegian government for the four steps analysed. As opposed to the FUSIONS-definition of food waste, food waste documented through the ForMat-project includes food waste utilized as animal feed or bio-based material/chemistry processing, and does not include inedible parts of food (Møller et al. 2013). This definition is used to ensure focus on opportunities for prevention instead of handling and treatment.

Edible food waste has been documented for the following four steps of the food chain: the industry, wholesale, retail and households. This means that food waste from other steps/sectors such as primary production, HORECA, public sector (schools, university, hospitals) and offices are not documented through the ForMat-project. Additionally, food waste from the fish industry (Norway’s largest industry after the oil- industry), has not been possible to document due to reduced data quality. All of the ForMat project's results must be considered in light of these limitations.

Edible food waste from industry, wholesale and retailers has been assessed through voluntary reporting of economic turn-over and amounts (tons) produced or sold, together with economic losses related to edible food waste or amounts (tons). 13 companies has contributed to data for the production step. These companies represent a wide range of production facilities and covers about 25 % of total sales in the Norwegian food industry. Data from the wholesale step is based on food waste records at a number of wholesale warehouses in Norway covering close to 50 % of the whole sector.

Food waste from the wholesale and industry has been documented for the nine main categories of food.

Food waste from retailers is based on data from 29 stores in 2010 and 2011, 58 stores in 2012 and 89 stores in 2013 to 2015. Both retail stores and wholesale warehouses are a representative sample of the whole retail and grocery outlets in Norway, with regard to geographical and demographic distribution, and stores with- or without DELI-sections. Food waste from the retail step has been documented for twenty-one different product categories.

Edible food waste from households has been assessed through waste-composition analysis and surveys. The waste-composition analysis was conducted in 2011 and 2015, and included one week of household waste from ca. 200 different households both years (Hanssen et al 2013b; 2016). This means that the project has only been able to document food waste discarded through the municipal waste system, excluding any food discarded through the sewage system. Food waste from the households has been documented for seven different product categories.

Data has been upscaled to national statistics by using the reporting companies' share of turn-over for the respective food-chain-steps. For the wholesale and retail steps, NOK/kg-ratios has been used for the different food categories to convert food waste in economic value into tons. For the households, the waste composition analysis has been upscaled by using national household waste statistics, and the results for 2010 and 2012-2014 has been calculated by extrapolating the data from 2011 and 2015.

An environmental impact analysis of GHG-emissions related to production, transportation, manufacturing and packaging of the edible food waste has been conducted for each year, using life cycle assessment (LCA)-methods in accordance with ISO 14040/44, European Commission JRC (2010) and European Commission JRC (2011). The GHG-emissions are estimated based on the amounts and composition of food waste in the various steps of the value chain, and is calculated by multiplying the amount of food waste (tonnes) for the relevant product group with the corresponding emission factor. The emission factors include all greenhouse gas emissions from cradle to retail, thus emissions from consumption (transportation home and cooking), disposal (waste collection and waste treatment) is not included. All emissions are converted to CO₂-equivalents by using standard characterization factors from IPCC.

In addition to gathering data on food waste statistics, the ForMat-project has conducted several surveys to gather information concerning root-causes, challenges and measurements on food waste. The retail sector was first surveyed in 2011 on root-causes, and later again in 2015 on both root causes and food waste prevention efforts, whereas a survey among 250 food manufacturing

companies was carried out in 2014 on root causes and food waste prevention efforts. Consumers has been surveyed by web panels on behaviour, attitudes and extent of food discarded every year – between 2010-2015.

Results

The project has retrieved insight of the development, hot-spots, composition and amounts of edible food waste, as well as root-causes, challenges and measurements to reduce food waste. The study shows that, in 2015, edible food waste from the four steps amounted to a total of 355 thousand tonnes per year, which equals 68,7 kg per capita over the total food chain.

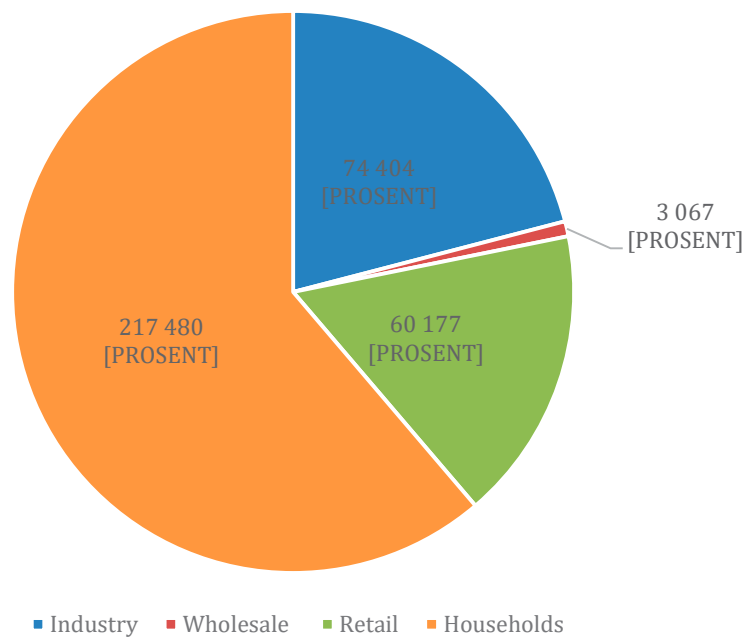


Figure 4 Tons and share of food waste generated within the four different steps of the food-value-chain, Norway 2015.

In 2014, edible food waste from households contributed to 61 % of total edible food waste, while industry and retailers contributed to respectively 21 and 17% of total edible food waste in Norway (Figure 1).

The composition of edible food waste varies between the different steps in the food chain; bread, fruit/vegetables, and leftovers from dinner meals are the main product groups that constitutes edible food waste from households. Fruit and vegetables is by far the largest category at the wholesale step, while bread, dairy products and fruit/vegetables contribute the most in retail. Liquid dairy, frozen ready-made food and dry goods are the main food waste categories in the industry.

By using LCA-data for primary production, manufacturing, packing and distribution of food, it is estimated that the edible food waste represents about 978 thousand tonnes of CO₂-eq. each year in Norway (note that this is only emissions from cradle to retail, meaning that emissions from usage (transport home and cooking) and waste collection/treatment are not included). It is important to note that prevention food waste could have a positive effect on many other environmental indicators as well (eg. Acidification, eutrophication, photochemical oxidation, NO_x and particulates mm.) and resource use (eg. Use of water primary energy and phosphorus).

Through the six year period, the overall mass of edible food waste has decreased by 7 %, or 25,4 thousand tonnes (Figure 2). The reduction has been most prominent in the retail step where food waste has been reduced by 10 %. However, in total, the households have contributed the most, reducing food waste from 230 to 217 thousand tons, equivalent to 5 %, or 10 thousand tons. The reduction at the household and retail step is mainly due to reduced wastage of bread, one of the largest food waste categories in Norway. For the industry sector food waste has been reduced by 8 %, or 6,7 thousand tons. The wholesale sector is the only sector where food waste has increased, and the increase is strong - equivalent to 16 %. The increased amounts of food waste at the wholesale step is solemnly due to increased wastage of fruit and vegetables. Despite the drastic increase in food waste, the wholesale step contributes to less than 1 % of total food waste, and thus the total increase in food waste is relatively small (ca. 0,4 thousand tons).

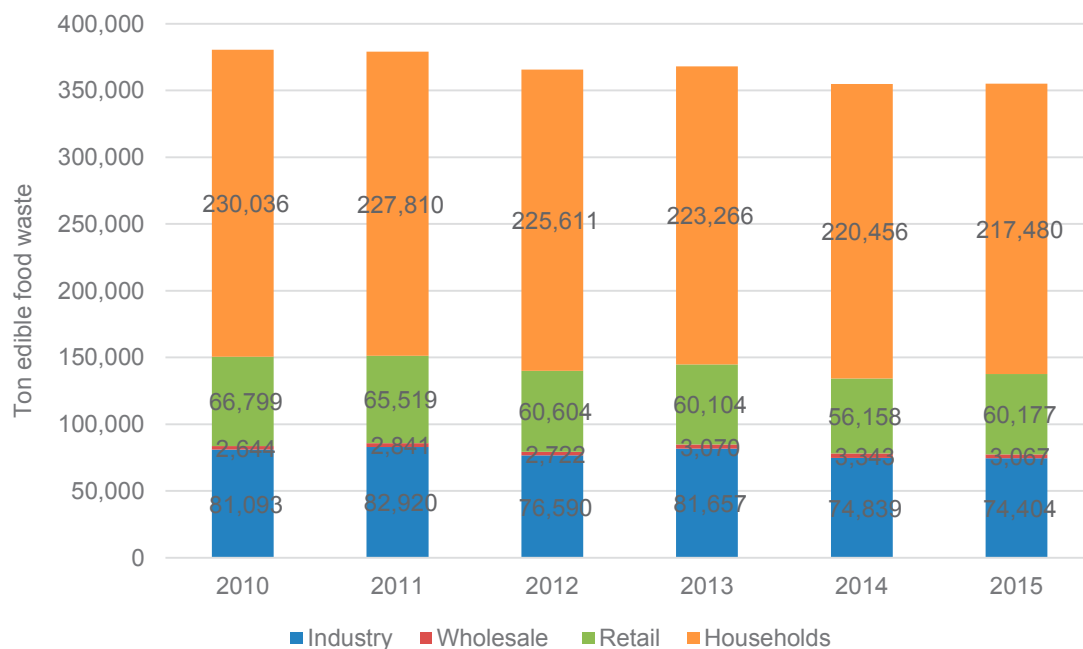


Figure 5 Tons and of edible food waste generated within the four different steps of the food-value-chain, Norway 2010 - 2015.

For the whole food value chain, edible food waste per capita in Norway has been reduced by 12 % from 2010 to 2015 (78,3 kg /capita to 69,7 kg/capita). This shows that in parallel with a steady growth in turnover in the Norwegian industry and retail-sector and an increasing population, the food-chain has increased its efficiency with regard to food waste prevention. In other words, this streamlined food chain is now producing, selling and consuming more food, yet the total mass of food waste has decreased.

The GHG-emissions related to the food waste has remained relatively stable throughout the period with a slight reduction of 3 % in 2015. The emissions have not been reduced as much as the amount of food waste measured in tons. This is mainly because food waste has increased for relatively climate-intensive food categories (meat and fresh ready-made food), and reduced for categories with low climate impact (bread). For example, the GHG-emissions related to food waste at the household step has increased by approximately 28 thousand tons of CO₂-eq., while the amount of food waste is reduced by approximately 10 thousand tons. This can be explained by the fact that the food waste from households has increased for meat-based products, like cookware and dish leftovers, while it is reduced for bread and bakery products.

The deviation between the development of GHG-emissions and ton food waste shows the importance of detailed material flow analysis. The main purpose of food waste reduction is to reduce GHG-emissions as well as other environmental burdens. Detailed food waste statistics used as input to LCA-studies, can make it easier for the food sector and/or government to make the right priorities, regarding both which step in the food chain, and which type of food that prevention measures should be focused on.

In order to reduce food waste, it is important to know the amounts and composition of food waste at each step, combined with knowledge about root-causes of why food is discarded. The surveys carried out in the industry and retail shows that the main reasons for food waste generation in the interface between food producers and retailers are:

- Difficulties in estimating how much food will be sold, and thus how much should be ordered. This is particularly a problem with seasonal products and products sensitive to weather conditions, such as barbecue items in summer.
- Too many consumer units in the retail pack, i.e. the cartons/trays delivered to the shop contain far too many consumer units than the shop can sell.
- A wide range of products varieties. Analysis of waste as a percentage of sales shows a sharp increase for many groups when sales volumes are low. This applies especially to products with a short shelf life. From a total food waste volume, it might however be more important

to focus on food products with medium sales volumes and medium waste percentages, which might give higher total food waste volumes.

Other causes were also identified as important, such as pressure on products and promotions (Hanssen & Getz 2011). At the consumer step the main reasons for discarding food is that the product has passed its expiry date, followed by reduced quality of product.

Conclusion and interpretation

The ForMat-project shows that generation of edible food waste in the four steps of the food-value-chain has been reduced by 7 % measured in total tons, and 12 % measured in kg per capita from, 2010 to 2015. Although food waste has been reduced, GHG-emissions from food waste has not be reduced as much (only 3 %). This shows that knowledge about food waste composition is crucial to achieve the desired effects of food waste prevention (reduced environmental impact).

Through the project, better methodologies for quantification and assessment of edible food waste has been developed and data quality has been improved significantly. Through yearly reporting of results back to the food sector (a sort of feedback-system), misunderstandings and errors has been corrected. Additionally, this feedback-system has increased the awareness of food waste in the sector. However, there are still uncertainties and limitation related to the data and associated results; the most important limitation is probably the lack of data from the fish industry, food waste discarded through the sewage system and lack of data for the other steps of the food value chain.

Despite the limitations, the detailed food waste statistics gathered through the ForMat project is a unique compilation of data, and gives insight to the development of food waste, hot-spots – both in terms of food chain steps and in terms of food categories, as well as insight to root-causes.

The data obtained through the ForMat-project can also help understand the consequences of food waste, both environmental, social and economic. National food waste statistics is often limited in terms of its application, especially when it comes to LCA, because data rarely are accessible on a product-specific level as well as allocated between the different value chain steps. The ForMat-project has succeeded in retrieving this detailed mass balance, ensuring a better understanding of the development and environmental effects of edible food waste in Norway.

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272. Wasted food minimisation versus utilisation within the circular economy – a case study of Ireland

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ABSTRACT

The potential environmental impact of food waste minimisation versus its utilisation in a circular bioeconomy was investigated using a case study of Ireland. The amount of wasted food and food residue produced in 2010 was used for business-as-usual (option a) and four alternate management options were assessed, option b: minimisation; option c: composting; option d: anaerobic digestion and option e: incineration. The environmental impacts global warming (GWP), acidification (AP) and eutrophication (EP) were considered. The results from the study found that a minimisation strategy of wasted food would result in the greatest reduction of all three impacts, -4.5 Mt CO₂-e (GWP), -11.4 kt PO₄³⁻-e (EP) and -43.9 kt SO₂-e (AP). For use in the circular bioeconomy anaerobic digestion resulted in the lowest environmental impact. From an environmental perspective this study showed that a wasted food prevention strategy yields far greater benefits for GWP, AP and EP compared to composting, anaerobic digestion and incineration.

Keywords: **Food waste, life cycle assessment, composting, anaerobic digestion, incineration**

1. Introduction

The principles of the circular economy are being promoted as a path to reduce the effects of climate change and the depletion of finite resources. At the 2016 World Economic Forum held in Davos, such narratives were being presented, endorsing a reorientation from the traditional linear path through a life cycle to a state of circularity. Whether such an approach is economically and environmentally beneficial for all ‘waste’ materials is questionable.

The Food and Agriculture Organization of the United Nations (FAO) (2015) estimated that approximately one third of global food production is wasted. In Ireland, approximately 1,267,749 t of wasted food and food residue (WFFR) was produced in 2010 (CSO, 2012), Oldfield and Holden, (2014) estimated that this WFFR contained approximately 4,204 t available N, 1,996 t available P and 2,313 t available K, which could be theoretically recovered and utilised through circulation rather than raw material consumption. Such cycling of nutrients from WFFR would divert mass from landfill, transforming “waste” materials into a value-added product (Mirabella et al. 2014).

The environmental impact of numerous management options for WFFR needs to be known in order to define the best strategy in a given situation (Ekvall et al. 2007) and the recovery of nutrients must contend with technologies that handle the material efficiently and effectively such as incineration, which might be of greater social importance. Life cycle assessment (LCA) has been used to evaluate waste handling for many years (Bernstad and la Cour Jansen, 2012, Laurent et al. 2014a/b). Recent attention has concentrated on how to advance life cycle assessments applied to the traditional waste hierarchy (Ekvall et al. 2007), where the function is to handle the waste, towards approaches to integrate waste prevention (Bernstad Saraiva Schott and Andersson, 2015; Cleary, 2010; Nessi et al. 2013).

The objective of this study was to calculate the potential environmental impacts of four WFFR management options (reduction, composting, anaerobic digestion (AD) and incineration) compared to business-as-usual (in 2010), considering the necessity to recover nutrients for primary production and the generation of energy as well as the primary function of handling waste.

2. Methods

An attributional LCA was conducted adhering to ISO 14040/44 (2006a; 2006b), implemented in GaBi v 6 software (thinkstep, 2015) with foreground data from Irish sources and peer reviewed journals, and background data from ecoinvent (ecoinvent, 2015) and GaBi 6 (thinkstep, 2015).

The goal of this study was to assess the potential environmental impact of wasted food reduction vs. WFFR utilisation in a circular economy in Ireland from a life cycle perspective. This was done in order to better understand the impact of decisions made with regards to WFFR management at a national level.

The LCA was carried out following the four stage LCA methodology. The CML midpoint methodology was used, and included the environmental impacts global warming (GWP), Eutrophication (EP) and Acidification (AP).

As the primary function of all options is to handle waste, the functional unit was the annual amount of WFFR managed in Ireland, using data for 2010, which was the most recent complete data available at the start of the study. The system included WFFR collection, transport, treatment and use. The impact of food production was excluded as it was common to all options and therefore had no impact on the analysis. Technologies included in the study were composting, anaerobic digestion and incineration.

A baseline of food WFFR management in Ireland along with four WFFR management options were assessed (Table 1): (a) baseline was business-as-usual (BaU), landfill and composting (2012 figure); (b) WF reduction with FR and minimal wasted food being composted (full capacity) and landfill; (c) composting of all WFFR; (d) AD and composting (to existing capacity); and (e) incineration and composting (to existing capacity).

Avoided mineral NPK production was credited to compost and AD digestate, and carbon sequestration was credited for compost. A carbon stability factor of 8% was used and was converted to CO₂-e using the method set out by Brandão et al. (2013). Electricity produced from incineration and AD was assumed to displace Irish average grid electricity supply and was taken from GaBi 6 database (thinkstep, 2015).

Table 1: Flows of food and WFFR (Tonnes) for Baseline (Option a) and four management options.

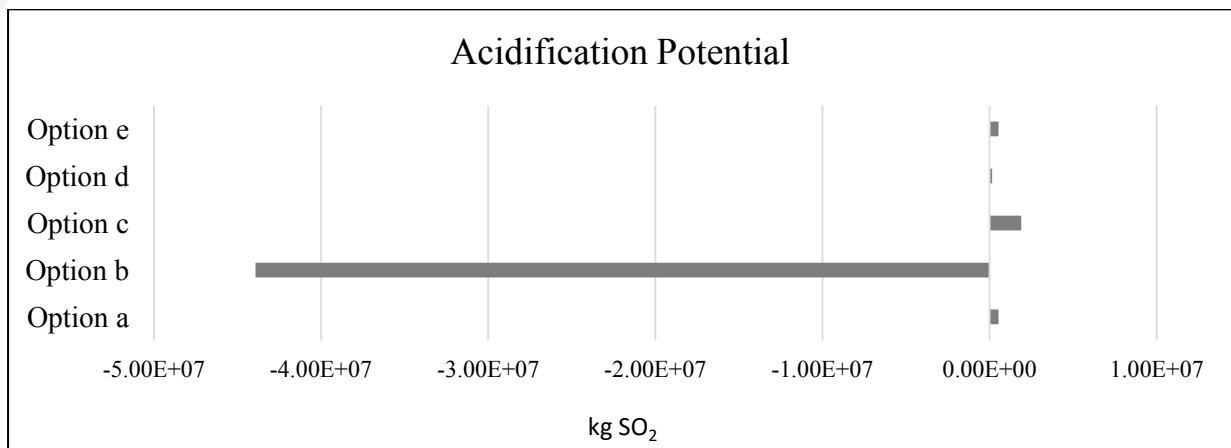
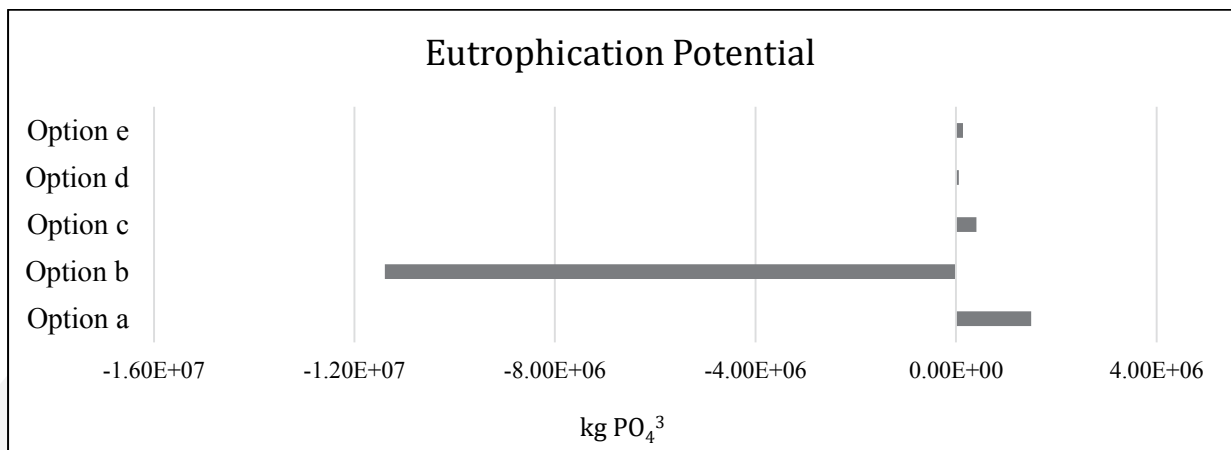
	FP	FC	FR	WF	WFFR	FP-C	L	C	AD	I
Option a	4,225,830	2,958,081	253,550	1,014,199	1,267,749	-	1,140,075	127,674	-	-
Option b	4,225,830	2,958,081	253,550	169,032	422,582	845,167	246,582	176,000	-	-
Option c	4,225,830	2,958,081	253,550	1,014,199	1,267,749	-	-	1,267,749	-	-
Option d	4,225,830	2,958,081	253,550	1,014,199	1,267,749	-	-	176,000	1,091,749	-
Option e	4,225,830	2,958,081	253,550	1,014,199	1,267,749	-	-	176,000	-	1,091,749

FP = Food produced, FC = Food consumed, FR = Food residue, WF = Wasted food, WFFR = wasted food and food residue, FP-C = Food produced credit, L = Landfill, C = Composting, AD = Anaerobic digestion, I = Incineration

3. Results

The results presented were consistent with the system defined in the scope of the study and included a contribution analysis as per ISO 14040 (2006a). It was found that the option with the lowest impact was Option b, the prevention of wasted food and composting of food residue, for global warming (-4.5 Mt CO₂-e), eutrophication (-11.4 kt PO₄³-e) and acidification (-43.9 kt SO₂-e)(Figure 1). There were significant differences in magnitudes for all three environmental impacts compared to the other options. The second lowest option for all three impacts was Option d, the management of WFFR through AD. This was approximately, 4.4 Mt of CO₂-e greater, 43.8 kt of SO₂-eq greater and 11.4 kt of PO₄³ greater than option b.

The contribution analysis for the Baseline and four management options was by stage: feedstock collection, technology processing, downstream benefit, distribution and application for AD and composting, including, electricity generated (AD and incineration) and carbon abated (compost), NPK avoided (compost and AD), and avoided food production (waste minimisation). For the baseline Option a, the most significant process was landfilling of WF+FR for all impacts. A small reduction in global warming and eutrophication was credited to the system for the 34,000 t of compost that was produced and the avoided mineral fertiliser (NPK), but this was insignificant when compared to the environmental impact of landfilling WFFR. For Option b, wasted food prevention with the management of residue via composting, the most significant process was the avoidance of food production for all three impacts. For Option c, the composting of all WFFR, the most substantial impact was the composting process for all environmental impacts. There was an offset from increased soil carbon due to compost application. For Option d, WFFR management by AD (86% of total WFFR) and composting (14% of WFFR) of WFFR, and Option e, the management of WFFR by incineration (86% of total WFFR) and composting (14% of WFFR), the process that had the largest contribution in both cases was the avoidance of grid electricity production for all environmental impacts.



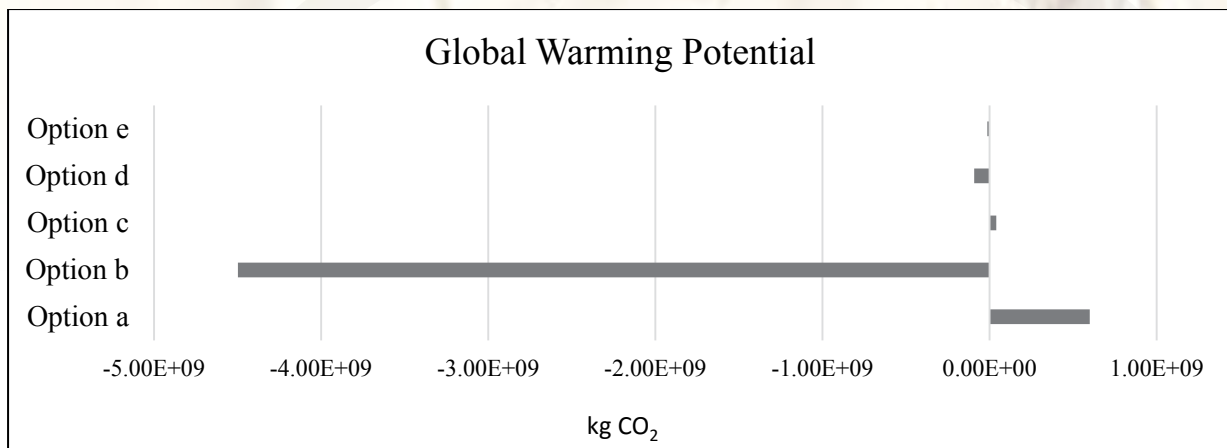


Figure 1 Global warming potential, eutrophication potential and acidification potential per functional unit of baseline and four management options.

4. Discussion

The results (and approach) of this study are similar to Bernstad Saraiva Schott and Andersson (2015). The wasted food minimisation option was found to lead to the greatest decrease in global warming impact compared to baseline. The technical alternatives for WFFR treatment/valorisation were also calculated to reduce global warming impact through nutrient and energy recovery, but wasted food prevention yielded far greater benefits compared to composting, AD and incineration. A similar case, but through different mechanisms was found for eutrophication and acidification. To put the impact savings into a wider context, the wasted food minimisation (option b) was calculated to have a global warming impact equivalent to a 31% reduction of vehicle emissions (circa 14 Mt CO₂) based on 2010 data (EPA, 2010).

There will always be a small amount of unavoidable wasted food and a slightly larger amount of food residue. A circular economy approach of creating high value products from WFFR can reduce the 2010 baseline environmental impacts using composting, AD, and incineration. From a systems perspective, only marginal differences were observed between competing downstream technological solutions (compost, AD, and incineration), which suggested the focus of efforts in this part of the food system should be matching feedstock to the best technology and maximising return on investment for processing food residues. Creating a bioeconomy built on wasted ‘fossil’ foods should be avoided because over capacity of technologies will have significant negative effects by reducing any incentives to minimise wasted food. For instance, a recent study of AD found the UK market was saturated due to the lack of feedstock (Eunomia, 2014). This demand does not have to be met by WFFR, but the results of this study (Figures 1) indicate that meeting such a demand should not undermine the incentive to reduce WFFR for maximum reduction on the environmental impact of WFFR.

The role of WFFR within the Irish bioeconomy must be carefully considered as the embedded impact of WFFR represents a significant environmental impact (GWP, AP and EP). The prevention of wasted food (Option b) has the potential to significantly reduce Ireland’s food related environmental footprint. The idea of diverting WFFR from landfill to a more productive use in a “circular bioeconomy” might be misleading when wasted food minimisation seems to have a much greater beneficial impact.

5. Conclusions

Of the four WFFR management options considered, wasted food minimisation was calculated to offer much greater environmental benefits than downstream processing options that would encourage a circular bioeconomy in Ireland. For unavoidable food residues and minimum wasted food, there was

little difference between anaerobic digestion and composting, but both were better than incineration. A clear understanding and distinction between what is food residue and what is wasted food is required to develop key performance indicators for the end of the food chain.

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157. Production of Stabilized Lactic Acid Bacteria viewed from a Life Cycle Assessment perspective

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ABSTRACT

Among food ingredients, concentrates of lactic acid bacteria are widely used as starters for producing cheeses, fermented milks, meats, vegetables and health benefit products (probiotics). This requires the stabilization of microorganisms allowing long term conservation and the best recovery of functional properties at the moment of product's use. Lactic acid bacteria production involves many unit operations: fermentation, cooling, concentration, formulation, freezing or freeze-drying, storage, transport between producers and users and reactivation of the stabilized micro-organisms for end-using (thawing, rehydration). Freeze-drying is known to be an energy-intensive process whereas freezing appears as a more eco-friendly alternative. However, environmental impact of storage and transport at very low temperatures (-40°C and below) may counterbalance such hypothesis. This work aims at analyzing such production system by using environmental Life Cycle Assessment (LCA) to identify hotspots and compare different existing scenarios.

LCA has been performed on the basis of collected data at laboratory scale, completed by databases. SimaPro (V8.0.5 PRé consultant) has been used for the LCA modeling with ILCD 2011 method. Since freezing, freeze-drying and storage damage the lactic acid bacteria, the impact scores have been weighted by the final bacteria quality, i.e. their physiological state, to obtain meaningful comparisons.

This work revealed that freeze-drying, frozen storage and fermentation were the hotspots to master the environmental impact of the system. Since freeze-dried bacteria can theoretically be stored at ambient temperature, the relevance of such a scenario was also evaluated as an option to decrease the environmental impact. However, due to bacteria quality losses, it was not found to be relevant. Finally, by comparing scenarios, it was found that the choice of the stabilization process to be used (freezing or freeze-drying) was depending on the storage duration that would be required for the product.

Such a study highlighted the relevance of a Life Cycle Assessment approach to provide valuable improvements in the management of the system.

Keywords: Environmental management, Micro-organisms, Preservation, Freezing, Freeze-drying, Cold chain.

1. Introduction

Micro-organisms are widely used to produce food (starter cultures for fermented products such as yoghurt, cheese, wine, beer, bread...) and bio-products (e.g. probiotics or production of chemical molecules, biofuels...). The need of ready-to-use micro-organisms is increasing. This requires the transformation of these unsteady structures into stable forms which can preserve their target functionalities obtained during their production. Stabilization must allow the transition between alive and dormancy states for a long term conservation and an efficient reactivation with the best recovery of functional properties at the moment of product's use. Micro-organisms can be stabilized by freezing or drying, and are then stored (most of time at cold temperatures) until end-using. Among processes, heating, cooling and refrigeration have been reported to consume approximately 45% of the total energy used in the food, drink and milk sector (European Commission, 2006). In addition, refrigerants may produce air pollutants and noise (European Commission, 2006) and cold chain has been shown to be closely related to global warming (James and James, 2010). Hence, processes involved in the production of stabilized micro-organisms clearly need environmental improvement.

The improvement of the environmental performances of such a production system must take into account the high sensitivity of the micro-organisms to the different steps of the whole process, from their production by fermentation to their final use. The production of microorganisms is thus a multi-stages system, with strong connections between steps, requiring a holistic analysis.

Life Cycle Assessment (LCA) is a holistic method which allows the quantification of the environmental impacts of a product, a process or a service. It is a standardized methodology (ISO 14040, 2006), and it is the most successful method in terms of global evaluation and multi-criteria. Concerning the food systems, this approach is widely applied to primary production. Some studies also consist in evaluating hotspots on the entire food chain (Roy et al., 2009). Few LCA studies focus on scenario comparison based on alternatives in the transformation part of the system. Food preservation scenarios have successfully been compared: autoclave pasteurization vs microwaves vs

high hydrostatic pressure vs modified atmosphere packaging (Pardo and Zufia, 2012). Environmental comparison of two different chicken meals (homemade and semi-prepared) has also been done, with two scenarios for each meal: the present conditions of the food chain vs a number of improvement actions in the stages after the farm (Davis and Sonesson, 2008). Such studies are very scarce and this is a pity because there is a real lack of informed evidences to choose process scenarios in a food chain.

In this paper, the production of concentrates of *Lactobacillus delbrueckii* subsp. *bulgaricus* CFL1, a lactic acid bacterium, has been investigated. It is mainly used for its capability of acidifying milk to produce yoghurt. As for all micro-organisms, its production involves many unit operations: fermentation, concentration, addition of protective molecule, stabilization, storage, transport between producers and users and reactivation of the stabilized micro-organisms for end-using, (Béal et al., 2008). This strain is very sensitive to cellular damages and has to be stabilized by using low temperatures. Freezing can be used to efficiently preserve the bacteria functionality, but very low temperatures are required (about -50°C) for storage and transport, which can be an issue from an environmental point of view. Freeze-drying is also a mild process with regards to the biological activity and allows the storage and transport of bacteria at higher temperatures (from -20°C to ambient temperature), but it is known to be energy-intensive.

The first aim of this study was to establish an environmental profile of the production system of such stabilized lactic acid bacteria in order to identify hotspots. The second objective was to compare the impact of the two scenarios of stabilization (freezing vs freeze-drying).

2. Methods

2.1. System under study

Lactobacillus delbrueckii subsp. *bulgaricus* CFL1 was the lactic acid bacteria strain studied in this work. The production process under study is composed of the following operations: growth medium preparation, fermentation (including sterilization of medium culture and fermentor, culture growth and cleaning), concentration by centrifugation, addition of cryoprotectants, stabilization by freezing or freeze-drying, storage and transport between producers and users.

2.2. Goal and scope of the study

The goal of this work aims is to analyze such production system by using environmental Life Cycle Assessment (LCA) to identify hotspots and compare different existing scenarios.

As described above, this study is a cradle-to-gate analysis. The functional unit used is stabilization and storage of 3kg of protected bacteria produced by three independent fermentations. This amount of bacteria completely fills one pilot scale freeze-dryer.

2.3. Impact characterization

SimaPro (V8.0.5 PRé consultant) has been used for the Life Cycle Assessment modeling with ILCD 2011 method (European Commission, 2011), which involves the midpoint impact assessment method.

The freezing, the freeze-drying and the storage damage the lactic acid bacteria. In order to compare the use of bacteria on the same basis, it was necessary to weight the impact scores by the final bacteria quality, *i.e.* their physiological state. The specific acidifying activity (t_{spe}) is a meaningful measurement of the physiological state of the bacteria (Gautier et al., 2013; Streit et al., 2007): the lower t_{spe} , the better physiological state is. Consequently, the environmental impact scores were multiplied by specific acidifying activity measured after storage.

2.4. Data collection

Most of raw material inputs and outputs were manually measured during pilot scale experiments performed at the Laboratoire de Génie et Microbiologie des Procédés Alimentaire (INRA, Thiverval-Grignon, France).

The energy consumptions of the storage freezers were quantified by using a tool provided by Intelligence Energy Europ (ICE-E). The annual leakage rates of refrigerants were assumed to be 15 % (ADEME, 2010). Volume allocation was applied to the storage freezers's total consumption and

leakage (0.08 % and 0.01 % of the total walk-in freezers' volume for the frozen and the freeze-dried products, respectively).

Transport to the customer was considered to be performed by a 20 t truck over a distance of 500 km. The truck was fully-loaded for going trip, and the return trip was considered with an empty truck (volume allocation of 0.16 % and 0.02 % of the total trailer for the frozen and the freeze-dried products, respectively). The transport data came from Ecoinvent v3 (Ecoinvent, 2013) with an increase of 21 % of fuel consumption and a leakage of 10 % per year of refrigerant charge (DEFRA, 2008) to take into account the refrigeration of the truck.

Generic data concerning production of energies, water and raw material have been recovered from Ecoinvent v3 (Ecoinvent, 2013). For grid electricity, average France data for electricity production were used.

3. Results

3.1. LCA of frozen bacteria

The environmental profile (GWP and ODP indicators) related to the production of bacteria stabilized by freezing is presented on Fig. 1. The contribution of each stage is reported as a percentage of the total impact.

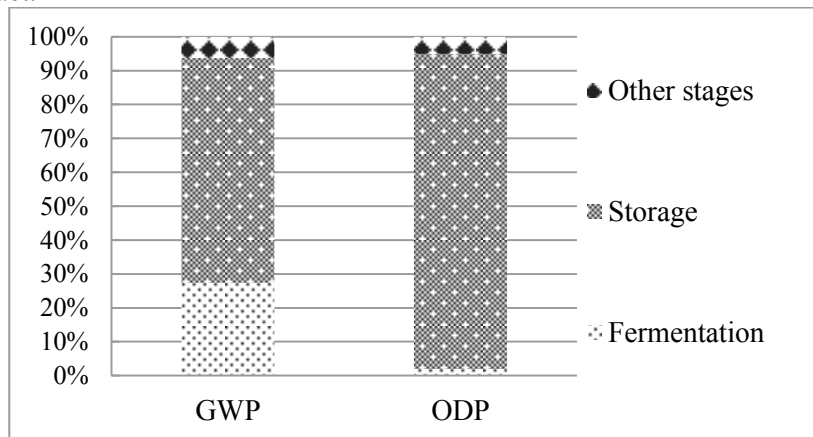


Fig. 1. Global Warming Potential (GWP) and Ozone Depletion (ODP) indicators of the contribution analysis for the global life cycle of frozen bacteria with -50 °C storage during 1 year - ILCD midpoint method.

When bacteria were stabilized by freezing, a considerable contribution was given by the storage stage (66 % for GWP and 91 % for ODP), mainly due to the huge electricity consumption of the walk-in freezer for GWP and to the leakage of refrigerant gas for ODP.

The fermentation stage was the second hotspot for GWP (27 %), due to the use of natural gas (steam production for the sterilization and the cleaning of the fermentor) and tap water (circulating in the double envelope for the control of the fermentor's temperature).

3.2. LCA of freeze-dried bacteria

The environmental profile (GWP and ODP indicators) related to the production of bacteria stabilized by freeze-drying is presented on Fig. 2. The contribution of each stage is reported as a percentage of the total impact.

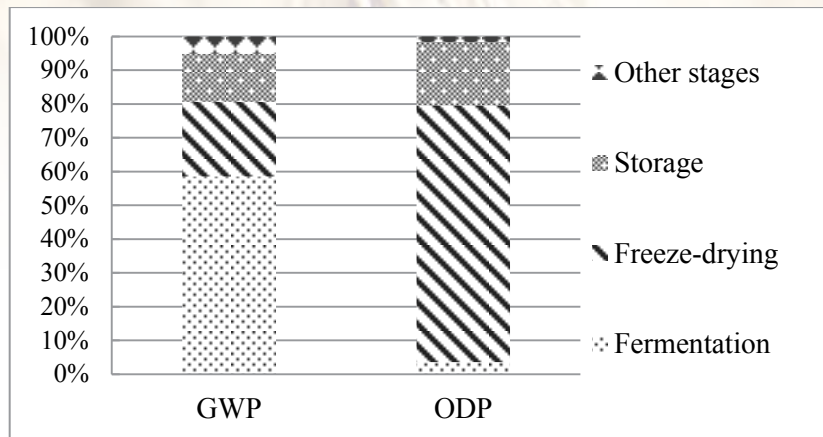


Fig. 2. Global Warming Potential (GWP) and Ozone Depletion (ODP) indicators of the contribution analysis for the global life cycle of freeze-fried bacteria with -20 °C storage during 1 year - ILCD midpoint method.

Freeze-drying itself was the main contributor to ODP (76 %), and also significant to GWP (20 %). The ODP contribution was mainly explained the leakage of refrigerant gas whilst electricity consumption explained GWP contribution.

Fermentation was the main contributor to GWP (58 %). As for the frozen bacteria, it was due to the use of natural gas (steam) and tap water (to maintain fermentor's temperature), but also from liquid wastes, which are a mix of organic residues (coming from culture medium), water, acids and bases used to clean the fermentor.

The storage was the third hotspot, contributing to GWP (15 %), mainly due to the huge electricity consumption of the walk-in freezer, and to ODP (20 %), mainly because of the leakage of refrigerant gas.

In the scenario presented above, the freeze-dried bacteria were stored and transported at -20 °C, but they can theoretically be stored at positive temperature, from 4 °C to ambient temperature (Béal et al., 2008). An idea to improve the freeze-drying scenario could thus be to increase the temperature of the freeze-dried bacteria's storage and transport. Fig. 3 shows the environmental impact (GWP and ODP indicators) of freeze-dried bacteria stored at ambient temperature during three months in comparison with the same bacteria stored at -20 °C (scenario above with storage duration of three months).

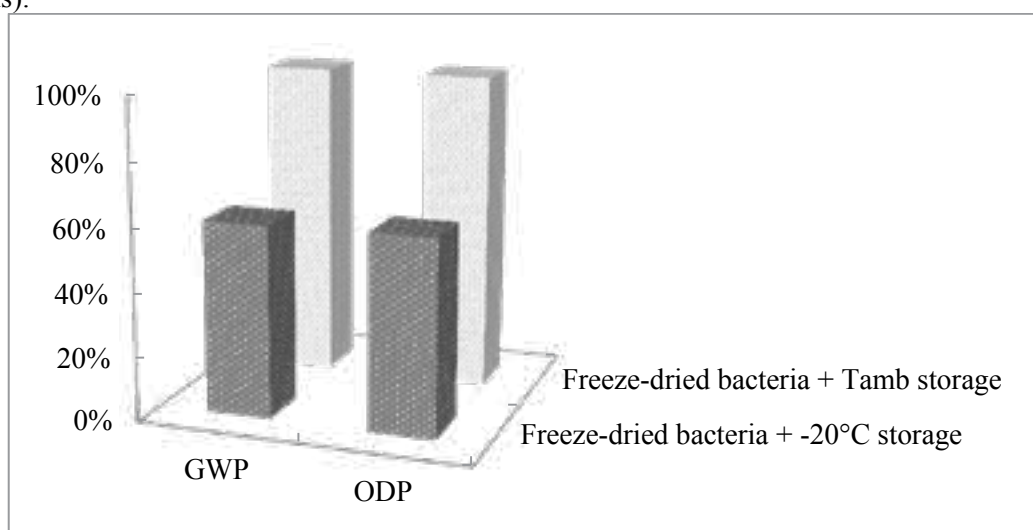


Fig. 3. Weighted impacts (Global Warming Potential (GWP) and Ozone Depletion (ODP) indicators - ILCD midpoint method) of freeze-dried bacteria stored at -20 °C (■) and freeze-drying bacteria stored at ambient temperature (□). Bacteria cells are supposed to be stored during 3 months.

Surprisingly, bacteria stored at ambient temperature led to a very much higher environmental impact than bacteria stored at $-20\text{ }^{\circ}\text{C}$ (about 40 % more, on both GWP and ODP). The drastic quality loss endured by the bacteria during storage at ambient temperature explained this surprising result. Such quality loss leads indeed to the use of more bacteria for a same usage function, and consequently more production is necessary. It was calculated that if bacteria quality remained constant, raising the storage temperature would reduce environmental impacts of about 10 % (data not shown).

3.3. Comparison between frozen and freeze-dried bacteria

The comparison between frozen bacteria stored at $-50\text{ }^{\circ}\text{C}$ and freeze-dried bacteria stored at $-20\text{ }^{\circ}\text{C}$ was performed. Environmental impacts have been calculated for different storage durations, from 1 month to 2 years (representative of standard conditions for industrial storage). Results from 1 to 8 months of storage are reported on Fig. 4 for GWP indicator.

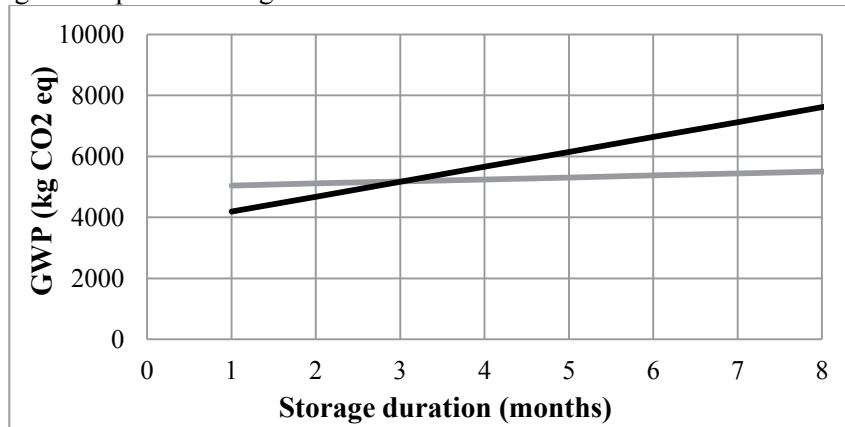


Fig. 4. Weighted Global Warming Potential (kg CO₂ eq) of frozen bacteria stored at $-50\text{ }^{\circ}\text{C}$ (—) and freeze-dried bacteria stored at $-20\text{ }^{\circ}\text{C}$ (—) as a function of storage duration (months) - ILCD midpoint method.

The GWP increased linearly with the storage duration. This increase was much faster for frozen than freeze-dried bacteria. For storage durations lower than 3 months, frozen bacteria had lower environmental impact than freeze-dried bacteria. Then, at 3 months of storage, GWP was the same for both frozen and freeze-dried bacteria. Above 3 months of storage, the freeze-dried bacteria's impact thus became lower than frozen bacteria's impact.

4. Discussion

This study evidenced improvement options on processes themselves which have been identified as hotspots in scenarios analysis.

Storage appeared to be critical whatever the chosen stabilization process. It has also been reported that cold storage was a key-point in the food cold chain and that energy savings could be estimated to 20-40 % by improving storage conditions (James and James, 2010). General recommendations are difficult to provide because each store room has its own specificities. In this work, bacteria quality loss during storage was dramatic, thus excluding the possibility of increasing storage temperature to reduce its impact. An environmental benefit could clearly be obtained if bacteria quality could be more efficiently preserved during storage.

The use of steam during fermentation has also been evidenced as an environmental issue. Its impact could be reduced by optimizing production planning, advanced control of steam-boilers, pipe and equipment insulation, and/or gas replacement by renewable sources (biogas) (Pardo and Zufia, 2012).

Another improvement option with regards to fermentation stage would be to reduce tap water consumption by using a recirculation loop, as it is industrially the case, and/or by using rainwater instead of tap water to maintain fermentor's temperature. Moreover, optimization of the acids and bases volumes used during the cleaning would be a way to reduce the environmental impact of the liquid wastes generated at this stage.

Finally, optimization of freeze-drying to reduce its electricity consumption would be of utmost interest to reduce the environmental impact when this process is used. As for storage, maintaining bacteria quality during this stage would also be of major importance from an environmental point of view.

5. Conclusions

Production of stabilized bacteria was investigated from an environmental point of view.

For freezing scenario, the main hotspot was storage and the second hotspot was fermentation. For freeze-drying scenario, freeze-drying itself was the first hotspot of the system, followed by fermentation and storage.

Comparison of both scenarios with regards to GWP showed that for storage shorter than 3 months freezing should be the preferred stabilization method whereas for storage longer than 3 months freeze-drying should be the choice to preserve bacteria.

Finally, a crucial aspect was the ability of the stabilization process to maintain bacteria quality over time. Quality loss means more production needed to fulfill the same usage function by the bacteria, and induces increased environmental impact.

6. References

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ORAL PAPERS

CASE STUDIES: DIET

16. Case Studies: Diet

254. LCA results for Austrian dairy farms related to socio-economic functional units

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ABSTRACT

Objective: In order to promote their sustainable development, food production systems (PS) should be analysed not only concerning environmental sustainability, but also in terms of their socio-economic performance and optimisation potential. Therefore, LCA results for Austrian dairy PS were related to the alternative functional units (FUs) labour income and satisfying working hours (SWh), thereby addressing both ecological and socio-economic aspects.

Material and methods: We used data from 30 dairy PS from Hörtenhuber et al. (2013), who analysed environmental impacts, economic performance and social aspects of Austrian dairy farms. Three exemplary LCA-indicators were selected: MJ primary energy demand for inputs, eutrophication and global warming potential. The FU SWh represents recorded working hours per kg of energy-corrected milk (ECM) which were reduced proportionally if the degree of job satisfaction was below average.

Main Results: Overall, the best results for the selected LCA indicators related to socio-economic FUs were found for pasture-based upland PS, upland PS with arable land and lowland PS keeping several livestock categories. While relatively poor results are caused by low milk yields and low income for the alpine PS, specialised PS perform poorly due to lower SWh per kg milk and due to bad LCA results. Despite correlations between the FU area and SWh on the one hand and product and labour income on the other, the performance of a PS will differ for different FUs.

Conclusions: The multifunctionality of agriculture can be reflected by relating impact category values which result from agricultural LCAs to different FUs (e.g. area, quantity of product, energy or protein). Considering important socio-economic services, the commonly used FUs should be extended to include FUs such as satisfying work time and labour income. Thereby ecological and socio-economical perspectives may be linked and thereby integrate different sustainability perspectives.

Keywords: milk production, sustainability, labour income, quality of work

1. Introduction

Generally, milk markets are in a changing situation due to internationalised markets, a reduction in trade barriers as well as an elimination of production quotas. For Austrian dairy farms, the rapid change is additionally driven by high production costs (about 0.50 to 1.00 € per kg milk; Kirner 2015), especially in the alpine regions, resulting in low and further diminishing margins. Many Austrian farmers try to compensate for this by increasing production to reduce costs. This, together with stagnant milk demand and despite the continuing process of small Austrian dairy farms going out of business, is the reason for rather declining milk prices. The number of dairy farms declined by 3.5 percent in the last decade (Kirner et al. 2015), simultaneously the milk yield per cow increased by 800 kg (ZAR 2015). The diminishing availability of agricultural land in the favoured areas exacerbates intensification. While growth is the strategy for many dairy farmers, especially in favoured areas, to survive in the competition (Schönhart et al. 2012), policy is focussing on support of less favoured areas, promoting product and process quality of Austrian milk and development of marketing and distribution channels for national and international markets (BMLFUW 2015). However, this support does not seem to be able to stop the financial stress for a proportion of dairy farms.

In order to promote their sustainable development, food production systems (PS) should be analysed not only concerning environmental sustainability, but also in terms of their socio-economic performance and optimisation potential. For this contribution, we focussed on dairy farm's most important key indicators from the different dimensions of sustainable development.

The study of Hörtenhuber et al. (2013) on sustainability aspects for Austrian dairy farms found – as other studies did (see Lebacqz et al. 2013) – that less intensive farms showed good environmental impacts per hectare, but unfavourable results per product unit. When addressing socioeconomic issues Life Cycle Costing and Social LCA-approaches generally use the same functional units (FU) as

environmental LCA does (see UNEP 2009). In contrast, the aim of this contribution is to connect environmental LCA results with alternative functional units (FUs) to gain a more integrated perspective for the interpretation of results on sustainable development. The following research questions are discussed and answered for the case study of Austrian raw milk production: What is the effect of the farm size, productivity and/or intensity on aspects of sustainable development? Which relevant synergies, trade-offs and the background driving forces can be found for the sustainability performance of Austrian raw milk production?

2. Methods

We used data from 30 dairy farms from Hörtenhuber et al. (2013), who analysed environmental impacts, economic performance and social aspects of Austrian dairy production. The 30 farms (23 conventional, 7 organic; representing the distribution in Austria) are allocated to six PS, mainly based on the level of milk delivered to dairies, the region and proportions of arable land and pastures. These six PS were defined as "Alpine" (AL), "Alpine-intensive" (AI), "Upland-pasture" (UP), "Upland-arable" (UA), "Lowland-mixed" (LM) and "Lowland-specialised" (LS) dairy farms. In the following figures the 30 farms are sorted according to an increasing number of cows within the PS. Three different sources were used to generate the data base for the analysis in Hörtenhuber et al. (2013): (1) Comprehensive farmer interviews with standardised questionnaires (on production-specific parameters and working processes, on costs and revenues, etc.) with at least the two main managing/working persons interviewed. (2) Farm data for the reporting to the IACS database, which is the basis for public payments. Data from sources (1) and (2), together with statistical data and results from scientific studies, were combined and put into farm models to obtain missing data for farm-specific parameters, using e.g. material flow models.

For the LCA data, system boundaries include all inputs and environmental impacts during the life cycle up to the farm gate. This covers on-farm processes during feed and milk production and also the production and transport of external inputs, i.e. concentrates, fertilisers, pesticides, fuels or electricity. Direct land use change and soil carbon sequestration effects were considered. Economic allocation was used to separate the demand for production inputs and environmental impacts to milk and beef. Further information on the LCA methodology can be found in Hörtenhuber et al. (2013).

For this contribution, we selected three exemplary environmental LCA-indicators from Hörtenhuber et al. (2013) to be related to physical and socio-economic functional units: MJ primary energy demand for inputs (PED), global warming potential (GWP) and eutrophication potential (EP). We tried to select important indicators for social and economic aspects for the FUs. Additionally, we wanted to address complementary aspects and decided to address (a) *labour income (LI)* and (b) *satisfying working hours (SW_h)*.

Although some of the 30 studied farms are farmed part-time and do not provide the main income for the farm workers, the *LI* from agricultural activities contributes an important proportion of the household income. *LI* is a net income (per working hour or per unit of milk) in the dairy farm, which was adjusted for costs of the own capital and owned land. It was calculated based on the method TIPI-CAL (Technology Impact and Policy Impact Calculation Model) which was developed by the International Farm Comparison Network IFCN to analyse the economic performance of farms and to compare them internationally. A low working time per unit of produced milk, i.e. a high labour productivity, is an important factor for high *LI*-results.

Contrarily, a high number of *SW_h* was identified to contribute to increased social sustainability; Timmermann and Félix (2015) argue that a higher working time in sustainable farming should be welcomed, because it allows for a fairer distribution of tedious and meaningful tasks. Particularly complex activities such as the observation of agro-ecosystems and how they respond to agricultural practices, e.g. maintaining and promoting biodiversity or farming small-structured units, require a lot of time and restrict efficient farming. These activities are particularly "meaningful" because they promote autonomy and self-determination, local knowledge, site-specific problem solving and improve the community among farmers (see Timmermann and Félix 2015; Weis 2010; Nordström Källström and Ljung 2005). Likewise, many Austrian dairy farmers understand their work as "meaningful" (Hörtenhuber et al. 2013) and they seem to consciously accept longer working hours to

carry out more complex, non-standard work processes. *SWh* represent recorded working hours per unit of ECM which were reduced proportionally if the degree of job satisfaction was below average. The working hours were calculated based on time (minutes) needed for specific processes. This also covered the involved people and their specific performance, i.e. their individual working time needed for specific processes; individual performances were assessed by the interviewed farmers and combined with default values calculated for the specific machinery and housing systems present on the farms. The estimated working hours were compared to the calculated ones and a very good agreement was found for daily work processes, which are mainly in or around the livestock housing systems. For other work processes, for instance in feed production, farmers had problems to assess the actual working hours, and hence detailed model calculations were used which considered e.g. the type of soil, the size of fields and used machinery. Furthermore, the degree of work satisfaction was surveyed according to a Likert-scale by using a standardised questionnaire with 23 themes that were differentiated according to work processes (e.g. satisfaction with work safety, learning opportunities, recognition, responsibility, etc.). If a single farm's degree of work satisfaction was below average, e.g. within one standard deviation, we proportionally reduced the number of working hours by one standard deviation as compared to the average of all 30 farms to obtain the *SWh*. The reduction of recorded working hours by the degree of job satisfaction to *SWhs* diminishes the working time per kg product for 15 out of the 30 farms, on average by 12%; the *SWh* of the other 15 farms with an above-average job satisfaction are exactly the same as the unweighted working hours. The job satisfaction is highly variable, reflecting farm-specific influences and shows no difference between the PS. While within the less intensively producing farms in AL and UP the job satisfaction seems to increase with the number of cows, it diminishes for farms within AI, LM and LS, which generally produce at a higher level of intensity. Contrarily to the economic sustainability indicator *LI*, a higher number of *SWh* was interpreted as contributing to increased social sustainability.

3. Results

For all PS except for AL, the productivity was found to increase with the numbers of cows per farm (see trends given in Fig. 1). These numbers for the productivity seem to be correlated to the milk yield per average cow and year and to PED per ha. Both the milk yield per cow and the PED per ha as indicators for production intensity also increase with the number of cows per farm and over the PS from AL to LS. Consequently, we used the term of increased “production intensity” to describe this combined effect.

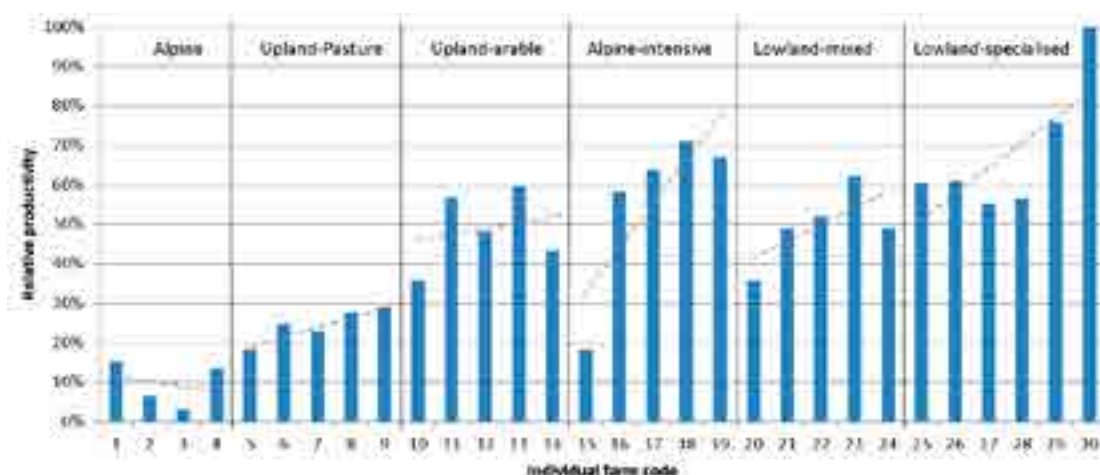


Figure 1: Relative values for productivity (indicated as mean between milk produced per area and per working hour); results for the individual farms are expressed in relative terms, with the lowest individual farm result as 0% and the highest as 100%; dashed lines represent the trends for productivity with increasing herd size within each production system.

The trends for the labour income (*LI*) per working hour increase from AL to LS and within the PS with an increasing number of cows and production intensity (Fig. 2). From an economic perspective alone, one may conclude that an increasing number of cows and production intensity leads to an increased economic sustainability. However, the relative values from the worst relative result of an individual farm (0%) to the best result (100%) for environmental impacts (LCA-results for PED, GWP and EP) per *LI* are similar for the different PS except for the alpine PS, which have substantially lower values (see Fig. 3). Results presented in Fig. 3 (and Fig. 5) are based on average estimates for the three indicators (PED, GWP and EP) related to alternative functional units and provide information on the relevant proportions of the three indicators for the overall result. For farms with a (very) low *LI* per working hour (e.g. for alpine farms), the low income determines the result for environmental impact per *LI*. For a moderate *LI* both moderate performances of the LCA results and the *LI* lead to the mostly moderate results. For the highest *LI* per working hour, moderate to good LCA impacts lead to good relative values (see farms 6, 7, 21 or 23) whereas comparably bad final results are defined by high environmental impacts (e.g. farm 30). For this relation of LCA results to the FU *LI*, the highly variable economic FU (with a factor of 29 between the lowest and the highest *LI*) strongly dominates over the LCA-impacts (with substantially smaller relative differences between the lowest and the highest farm results).

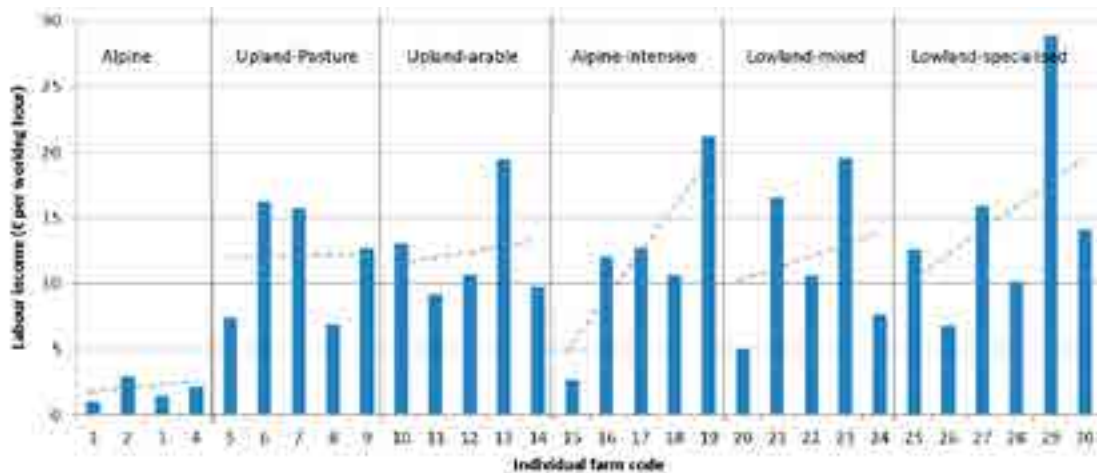


Figure 2: Labour income (*LI*) in € per working hour for 30 dairy farms (dashed lines represent the trends of labour income with increasing herd size within each production system).

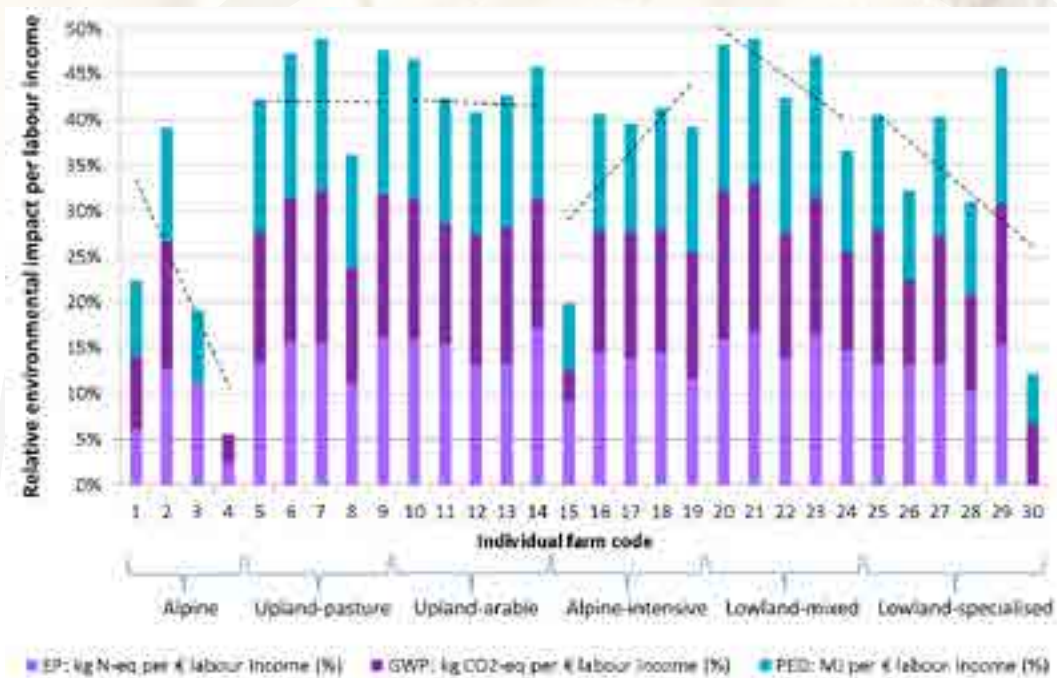


Figure 3: Relative values for environmental impacts (indicated as the sum of impacts from EP (kg N-eq), GWP (kg CO₂-eq) and PED (MJ) relative to the poorest, i.e. 0 and the best, i.e. 100 individual farm result for the single impacts) per € labour income (LI). Dashed lines represent the trends for environmental impact per labour income with increasing herd size within each production system.

For an increasing production intensity of the dairy farm (mainly influenced by its location), we found a strongly decreasing number of working hours and consequently a strongly decreasing number of *SWh* per kg product (energy corrected milk, ECM). Within the six PS, the *SWh* also tend to decrease with an increasing number of dairy cows per farm and increasing production intensity (Fig. 4). For the LCA-results related to the social FU *SWh*, decreasing relative values were found from PS AL to PS LS with an increase of production intensity and the number of cows (Fig. 5).

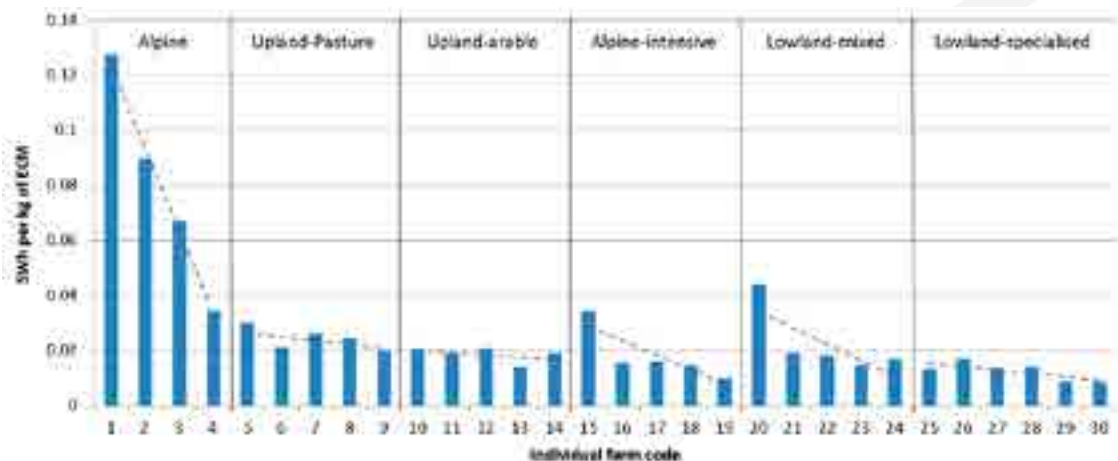


Figure 4 Satisfying working hours (*SWh*) per kg of energy corrected milk (ECM) for 30 dairy farms (dashed lines represent the trends for working hours with increasing herd size within each production system).

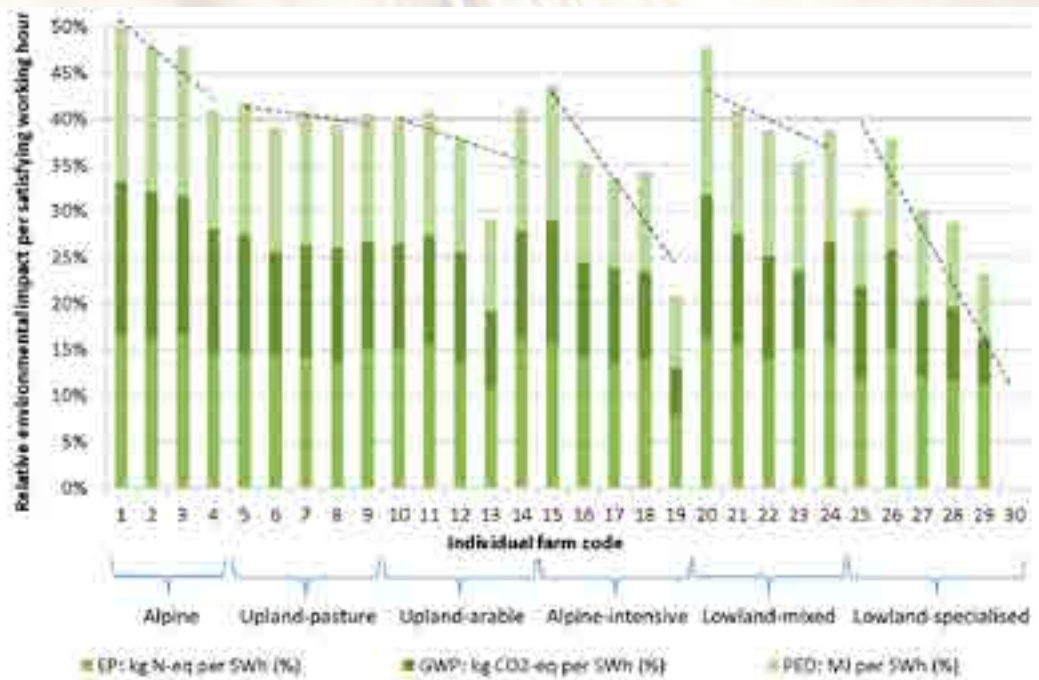


Figure 5: Relative values for environmental impacts (indicated as the sum of impacts from EP, GWP and GWP relative to the poorest, i.e. 0 and the best, i.e. 100 individual farm result for the single impacts) per *satisfying working hour* (*SWh*). Dashed lines represent the trends for environmental impact per satisfying working hour with increasing herd size within each production system.

The overall sum of all environmental impacts (PED, GWP, EP) related to both the social FU *labour income* (*LI*) and the economic FU *satisfying working hours* (*SWh*) represents best results for the PS UP and LM, followed by UA. The worst overall results were found for LS, followed by AL and AI. The reason for the poor overall performance of AL is mainly the poor economic sustainability, for AI the high production intensity, mainly connected to the low *SWh* and the only moderate *LI*. For LS not only the environmental impacts are bad, but also the *SWh* per kg product are low; however, they are compensated for by relative high *LIs*. The moderate intensity of UP, UA and LM provides moderate to good environmental impacts, moderate results for the analysed FUs and in conclusion good overall results. For LM this is due to the fact that the dairy branch of the mixed farm, which is only assessed herein, is operated less intensively than the other branches (e.g. fattening pigs or beef production).

4. Discussion

A substantial trade-off was found between results related to either *kg product* (*ECM*) or *hectare* by Hörtenhuber et al. (2013); this is also discussed in the literature (see e.g. Lebacqz et al. 2013). Hörtenhuber et al. (2013) did not conclude whether the FUs *hectare* or *kg product* are better for specific comparisons. The results per product unit are good measures for eco-efficiency. However, the highest relative eco-efficiency has no advantage if the limits for absolute environmental impact are exceeded; the latter may be characterised by relating environmental impacts to the FU *hectare*. Contrarily, Pierrick et al. (2012) found a close match between economic and global environmental performance within their investigation on eco-efficiency of Swiss dairy farms. Similar conclusions were drawn by Mouron et al. (2006), who state that ecotoxicity, eutrophication and non-renewable energy use do not necessarily increase with increased farm income on 12 specialised Swiss fruit farms. Thomassen's et al. (2009) relation of LCA indicators to labour productivity of Dutch dairy farms showed a more differentiated picture: High labour productivity on dairy farms was associated with low on-farm energy use, total and on-farm land use, total and on-farm global warming potential, and total and off-farm acidification potential per kg fat-and-protein-corrected-milk. High labour productivity, however, was associated also with high on-farm eutrophication and acidification

potential per hectare. Thus, our study benefits from a more detailed view on eco-efficiency of different PSs.

Hörtenhuber et al (2013) identified “ecologically successful” farms as a consequence of moderate to good environmental impacts for PED, GWP and EP, both per *hectare* and *kg product*. Interestingly, this approach leads to the identification of the same PS (UP, UA and LM) as above for overall environmental impacts related to the socio-economic FUs. For the case of Austrian dairy production (some even strong) correlations between results for the FUs *hectare* and *SWh* on the one hand and *kg product* and *LI* on the other were found, but results for the performance of single farms will differ for different FUs. As for the FUs *kg product* and *hectare*, trade-offs for the two socio-economic indicators are visible: an increase in *LI* tends to be related to a decrease in *SWh* and vice versa. This trade-off between complementary FUs was consciously chosen in order to adequately consider sustainable development from multiple perspectives.

From a practical perspective and taking results from the literature into consideration, the findings appear logical: a moderately-intensive, forage-based dairy PS (e.g. UP and UA) which is well adjusted to the respective site conditions should allow for moderate to good results for the FUs *hectare* and *kg product*. Furthermore, those PS are also economically successful and moderately labour-intensive. This is to some extent also confirmed by results from the literature on the comparison of the dairy industry regarding LCAs in different countries: the relatively poor economic performance of Austrian milk production due to locational disadvantages and less intensive systems (leading to higher production costs as compared to e.g. Danish PS; (IFCN 2015) – are contrarily accompanied by best GWP and PED values per *kg product* and per *hectare* (see e.g. Leip et al. 2010). If Austrian farms intensify with an increasing dependency on external inputs, e.g. concentrates which are partly connected with (direct) land use change-related emissions, a majority of indicators would show disadvantages, as visible in countries with intensive milk production (Leip et al. 2010).

For the case study of Austrian dairy production, the large deficit of economic sustainability for the PS AL (measured in terms of the *LI*) leads to reduced overall sustainability and relatively poor LCA results when they were related to the socio-economic FUs. This indicates that the public sector would have to support these alpine farms constantly and additionally, if they should continue to provide their services (food production and maintenance of the cultural landscape) because these farms will otherwise not survive economically in the internationalised markets. An intensification of alpine farms could also lead to benefits, if the *LI* increases without growing negative environmental impacts. The risk, however, exists that marginal areas with a high value for biodiversity and the landscape could be given up in the course of an intensification process.

The impact of a general increase of the farm size, i.e. the numbers of dairy cows and the cultivated area, cannot be easily estimated. For some effects, e.g. for *LI* and job satisfaction in PS AL or UP, a positive trend was found; other aspects may react in an opposite way. According to our results, the specialisation of the LS-farms does not seem to be optimal: As compared to the PS LM, milk in PS LS is produced at a very high level of intensity and environmental limits seem to be overstretched, especially per hectare (see Hörtenhuber et al. 2013) and per *SWh*. For LM-farms other livestock than dairy cattle is managed intensively, including the use of comparatively high proportion of concentrate feed; this reduces availability for dairy cattle. Furthermore, results related to the FUs *LI* and *SWh* are in opposite to each other for the PS LS.

The combination of unfavourable results for environmental impacts and unfavourable results for one of the two socio-economic FUs can be seen as a knock-out criterion. Based on our results, we conclude that an advantage exists of constant, moderate to good performance in all analysed aspects of sustainable development (as for UP, UA and LM); a different scenario exists for LS, AL and AI, which rank very good for some indicators, while others show only moderate to poor results. A similar interpretation can also be assumed for the results of a comprehensive sustainability assessment with a multi-criteria analysis (see e.g. Schader et al. 2016).

Generally, if manageable concerning data collection, it was found to be very important to additionally consider some socio-economic indicators besides LCA-traits for a detailed analysis from a life cycle perspective. For specific research questions and indicators (e.g. energy use, global warming potential and profitability), the detailed and quantitative approach described herein may be particularly suitable. For other specific purposes, such as benchmarking or farm advice, a comprehensive multi-criteria analysis probably fits better and the two different approaches should be

seen as complementary (Schader et al. 2016). To summarise an important methodological issue: the use of the socio-economic FUs seems appropriate in order to broaden the perspective in the interpretation of results and to consider the multifunctionality of agriculture by directly combining results from different sustainability dimensions, i.e. environmental impacts related to socio-economic services. However, combining results from different dimensions via alternative FUs provides clear statements only in connection with identification of the driving force(s) for the results.

5. Summary and conclusions

Dairy farming is the main agricultural branch in Austria, where (alpine) grassland dominates the landscape in many regions. It provides many products and services, but also shows a substantial environmental impact. Dairy farming produces food (milk and beef) from mostly non-edible plants for humans, it contributes to the maintenance of the cultural landscapes (with a high value for Austrian tourism) and biodiversity. Additionally, it provides job and income opportunities for many people. Four functional units (FUs) were identified which represent the multi-functionality of dairy production: *kg product*, *hectare agricultural area* (for maintenance of landscape and its biodiversity), *labour income* and *satisfying working hours*. Behind the background of the reviewed literature and from a methodical point of view we found it useful to differentiate our study according to several production systems. Relating the results for important impact indicators to these FUs and especially to the alternative socio-economic FUs shows the best overall results for pasture-based upland (UP) PS, upland PS with arable land (UA) and lowland PS keeping several livestock categories (LM). The findings reported herein seem to indicate that the intensive alpine PS (AI) and the more intensive specialised lowland PS (LS) operate at a too high level of production intensity, leading to relatively poor LCA-results. Alpine dairy farms show poor overall results related to the alternative socio-economic FUs because their income is too low. To sustain the multifunctionality of agriculture for the future, agricultural policy and public payments should specifically address the respective weaknesses of PS. Hence, the integration of socio-economical perspectives needs to be included in the analysis of sustainability aspects of (dairy) farming.

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273. Environmental assessment of the impact of school meals in the United Kingdom

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ABSTRACT

The aim of this research work is the creation of a robust methodology and a related tool (**E**nvironmental **A**ssessment **T**ool for **S**chool meals - EATS) that can facilitate those involved with providing / deriving school meals menus to assess the environmental performance of their school meals. The EATS tool utilizes secondary data to calculate values of carbon footprint and water footprint for a school meal from *cradle to plate*. This includes four phases: (1) food production, (2) transport of each ingredient from the country of origin to the UK, (3) storage at regional distribution centre and (4) meal preparation in a generic school kitchen.

EATS was tested against a set of nutritionally compliant meals; this paper presents the results from which it can be seen that there is a predominance of the production phase in the overall carbon footprint. In addition there is a decrease in carbon and water intensiveness when shifting from meat based recipes to non-meat ones. The main outcome of this work is the creation of a tool that can potentially be used by any school and catering provider in the UK to assess the performance of its menus and which, thanks to its simple user interface, has a great potential for engaging non-scientific audiences on the topic of sustainable food choices.

Keywords: Public food procurement, carbon footprint, water footprint, environmental impact, sustainable diets.

1. Introduction

There is increasing awareness of the role schools play in both promoting healthy eating habits and providing education for sustainable development (Jones et al., 2012, Morgan and Sonnino, 2007, Weitkamp et al., 2013). In the last decade a number of programs have appeared in the literature, with a shared aim of reconnecting school pupils with the natural component of food. The underlying ethos of each was to form empowered consumers (i.e. children and parents), aware of the consequences of their food choices on their health and the environment. In Italy for instance a program called *Cultura che Nutre (Culture that Feeds)* was set up in order to teach schoolchildren about the links between products and places with the sole purpose of making them aware of the value of locally produced high quality food (Morgan and Sonnino, 2007). Likewise, in the UK the *Food for Life Partnership*, a coalition of charities that promote food-based environmental learning in schools, collaborated with over 3600 schools between 2007 and 2011 (Weitkamp et al., 2013).

Within this context, we propose a methodology and a related tool (**E**nvironmental **A**ssessment **T**ool for **S**chool meals - EATS) that can facilitate those involved with providing / deriving school meals menus (e.g. catering providers and schools) to assess the environmental performance of their school meals. The aim is to develop a robust methodology, based on quantitative assessment of the environmental impacts of food (in terms of carbon footprint and water footprint) that can be used to assess and compare existing menus and help suggest improvements therein. Additionally this tool can be used for educational purposes to teach pupils / parents / stakeholders about the impact of food - engendering more sustainable choices. This paper explains how EATS was created (Section 2) and how it takes into account the different phases of the life cycle of a meal, whilst in the results section a number of meals are analyzed and compared using EATS (Section 3). A discussion of the findings is provided in Section 4 and conclusions are subsequently drawn in Section 5.

2. Methods

The ethos behind EATS is that it should provide the users with a simple-to-use interface (Figure 1) that allows them to input information on an individual recipe and be provided with respective outputs on the impact of each portion served.

As such the following *inputs* are required from the user:

- Name, weight and country of production of each ingredient;
- Number of portions required;
- Cooking appliances used and for how long.

The respective *outputs* are given:

- Carbon Footprint (CF)
- Water Footprint (WF)

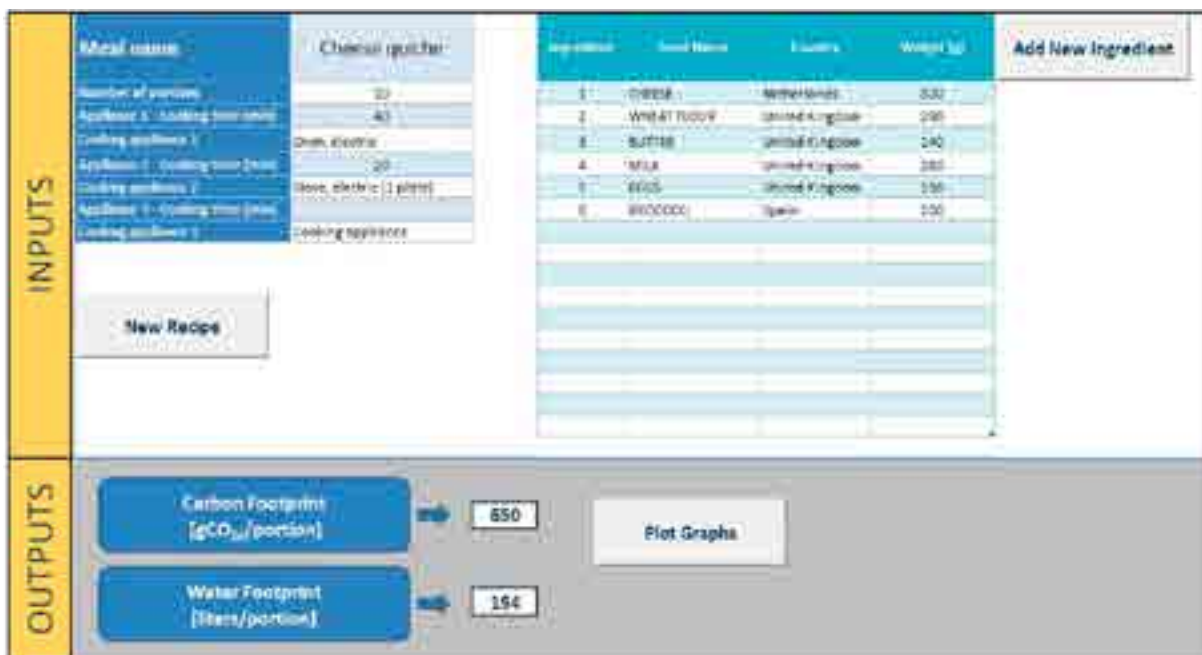


Figure 1: Interface for the EATS tool

The study is from *Cradle-to-Plate* and therefore system boundaries and assumptions are required within the following phases of the life cycle (Figure 2):

1. Production (Section 2.1);
2. Transport (Section 2.2);
3. Storage at regional distribution centre (RDC) (Section 2.3);
4. Meal Preparation (Section 2.4).

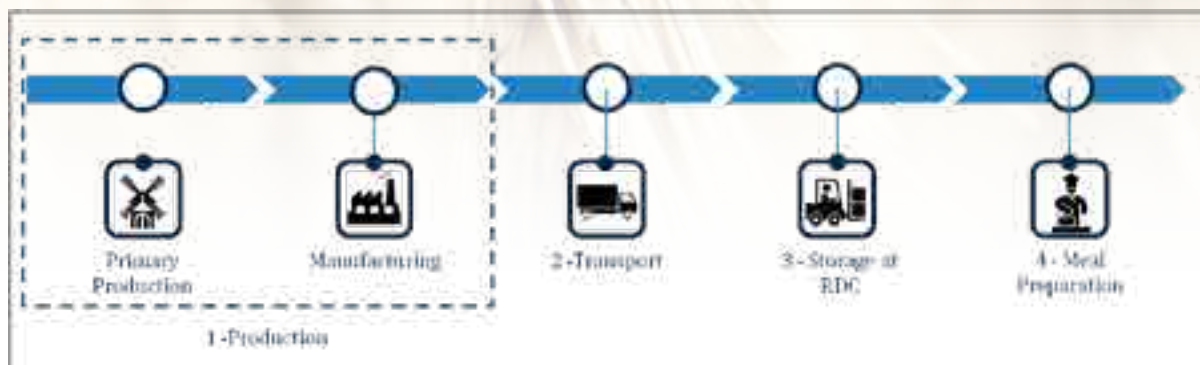


Figure 2: System boundaries and life cycle phases

For example, waste is included along Phases 1, 3 and 4 (waste during the transport phase is assumed to be zero) but waste generated at the consumption stage (i.e. plate waste) is not taken into account.

2.1 Phase 1: Production (*cradle to gate*)

In this phase EATS provides values of CF and WF for the list of food items within the menu relative to the production phase. The corresponding values were obtained through the collection of secondary data, which led to the creation of a database. To ensure completeness of the database, the food items to be included were obtained from the analysis of the results of the Primary School Food Survey, a national survey conducted in 2009 by the School Food Trust (Haroun et al., 2009) to collect information on school dinners across the UK.

For each food item, a search was performed through peer reviewed articles, conference papers, existing databases and Environment Product Declarations (EPDs) of food items in order to collect existing values of:

- Carbon Footprint (i.e. Global Warming Potential - GWP): these are calculated following the life cycle assessment (LCA) methodology, as specified in the ISO 14040/14044 (ISO, 2006a, b).
- Water Footprint: this includes green, blue and grey-water calculated according to the methodology set out in Hoekstra et al., (2009) [The authors appreciate that ISO 14046 (2014) is directly applicable to WF. However, due to data availability, when collecting values of WF of food products it is preferable to use the methodology developed by Hoekstra et al., (2009)].

When collecting values of GWP, some articles included a range of system boundaries, therefore only values relative to the '*production*' phase were extracted.

Emissions related to packaging production were typically included in the system boundaries of the studies consulted. However, there appeared to be a lack of a systematic approach adopted by authors in order to verify this.

The functional unit considered in the EATS database is 1 kg of each product. Specifically, for meat and fish products, the functional unit considered is 1 kg of retail weight. When in the studies consulted the functional unit was different (e.g. 1 kg of carcass weight or 1 kg of live weight), the conversion methodology proposed by Nijdam et al. (2012) was adopted.

2.2 Phase 2: Transport

In this phase the EATS tool calculates emissions related to transport for each ingredient inserted by the user, based on the country of origin selected (these are accessed via a dropdown menu). Average transport routes from every European capital to the UK and respective values for transportation distances were taken from the following websites:

- Transport routes from <http://www.cargorouter.com/>;
- Sea distances from <http://www.sea-distances.org/>;

- Road distances from <http://maps.google.co.uk/>.

The corresponding emissions were calculated using coefficients for road freight (1 Km = 0.1625 gCO₂/Kg) and sea freight (1 Km = 0.0156 gCO₂/Kg) suggested by DEFRA (2015).

2.3 Phase 3: Storage at regional distribution centre (RDC)

Emissions related to storage of food at regional distribution centers (Phase 3) were not included as they are considered to be negligible (Brunel University, 2008). However, this phase was taken into account when assessing waste levels through the life cycle of a meal (see Section 2.5).

Similarly, emissions related to refrigerated storage in school kitchens were not included as the purpose of the tool is to enable a comparison between different meals. [Any changes in the menus offered would unlikely affect the number and size of refrigerators utilized and their consequent energy use - at least in the short term (Garnett, 2008)].

2.4 Phase 4: Meal Preparation

The contribution to CF of the preparation phase is calculated according to values of energy consumption (kWh/minute) for a range of cooking appliances - see Carlsson-Kanyama and Faist (2001) - and the cooking time. These values were converted into corresponding emissions (gCO₂) using coefficients for the UK electricity grid (1 kWh = 0.5311 KgCO₂) and natural gas consumption (1 kWh = 0.2093 KgCO₂) provided by DEFRA (2015).

Based on the inputs provided by the user (i.e. type of cooking appliance used, the cooking time and the number of portions), the tool calculates the relative CF for preparation of each portion(s) of the meal analyzed. Water use during the preparation phase is assumed negligible when compared to production phase (Strasburg and Jahno, 2015).

2.5 Waste

The values of CF and WF relative to the production phase are recorded in the EATS database according to the functional unit of 1 kg at farm/factory gate. When the user inputs the weight of each ingredient in the tool, these values are scaled accordingly (see example below). However, some additional considerations are required to account for waste through the remaining parts of the supply chain (*gate to plate*). In other words if a recipe requires, for example, 100 g of broccoli, 110 g will need to leave the farm (Figure 3). These include a rate of 2% waste at regional distribution centers (RDC), akin to our Phase 3 (Brunel University, 2008) and a rate of 7.4% waste at Phase 4, meal preparation (Quested et al., 2012)

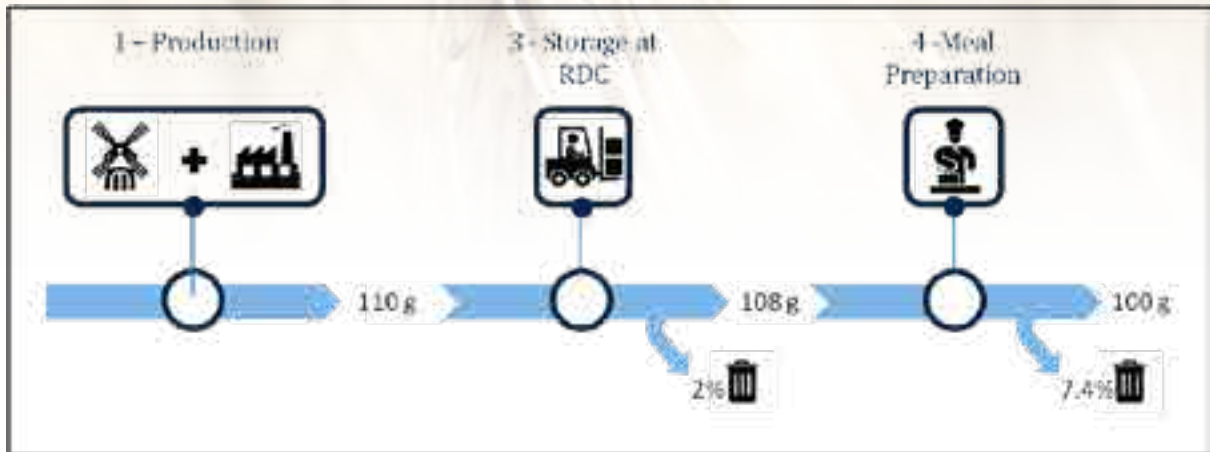


Figure 3: Food flows through the life cycle of a meal, waste at Phase 2 (transport) is equal to zero

Hence, in this case the user will input 100 g of broccoli into EATS. The tool will extract the corresponding values of CF from the database (i.e. 1 kg of broccoli = 377 gCO_{2e}) and scale them to 110 g, which equals to 41.5 gCO_{2e}. The same applies to the WF values.

An example application of EATS is provided in the following section.

3. Results

3.1 Application of EATS to an individual recipe.

For this paper the example of a fish-based recipe “Salmon and Broccoli Pasta”, suggested by the *School Food Plan Alliance* (<http://www.schoolfoodplan.com/>) was inserted into the EATS tool in order to assess its environmental performance in terms of CF and WF.

The list of ingredients required as input parameters for this recipe are shown in Table 1. The respective weighted quantities necessary to produce 13 portions (i.e. a typical primary school serving) are shown in the right column. The cooking procedure according to the recipe comprises 30 minutes on an electric stove. The total CF and WF per portion are 382 gCO_{2e} and 113 liters respectively. Breakdowns of the CF and WF according to each ingredient are represented in Figures 4 and 5 respectively. The figures show that Salmon contributes most significantly to CF (62%) whilst Pasta contributes most significantly to WF (64%). In fact both Pasta and Milk feature strongly in both the CF and WF (Pasta – CF = 20% and WF = 64%, Milk – CF = 13% and WF = 22%).

Table 1: Ingredients for Salmon and Broccoli Pasta

Ingredient	Food Name	Country of Production	Weight [g]
1	PASTA	Italy	650
2	SALMON	United Kingdom	800
3	BROCCOLI	United Kingdom	200
4	MARGARINE	United Kingdom	35
5	ONIONS	United Kingdom	100
6	WHEAT FLOUR	United Kingdom	35
7	MILK	United Kingdom	500
8	SPICES	World	5

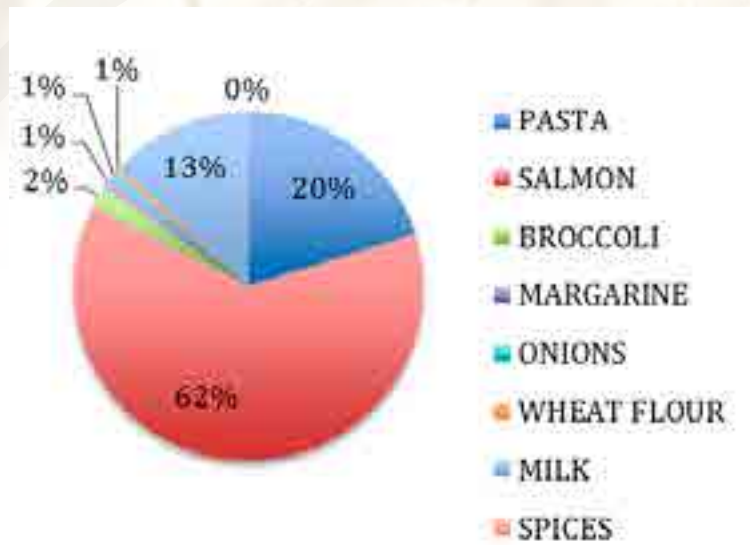


Figure 4: Contribution of each ingredient to the Carbon Footprint (CF)

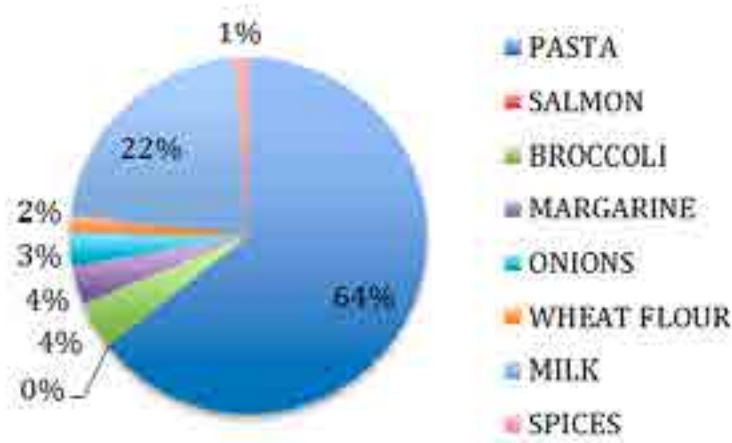


Figure 5: Contribution of each ingredient to the Water Footprint (WF)

Figure 6 shows the predominance of the production phase of CF, this is in line with the findings from existing literature (Heller et al., 2013). In addition and as found in similar studies conducted (Carlsson-Kanyama, 1998, Davis and Sonesson, 2008, González-García et al., 2012, Saarinen et al., 2012, Sonesson et al., 2005), the transport phase tends to have a minor weight if no product is transported through air freight (Carlsson-Kanyama and Gonzalez, 2009). The preparation phase may have a larger weight in recipes that involve a more extensive use of cooking appliances. However in this case the contribution from transport amounted to only 30.6 gCO_{2e} (3% of total) per portion.

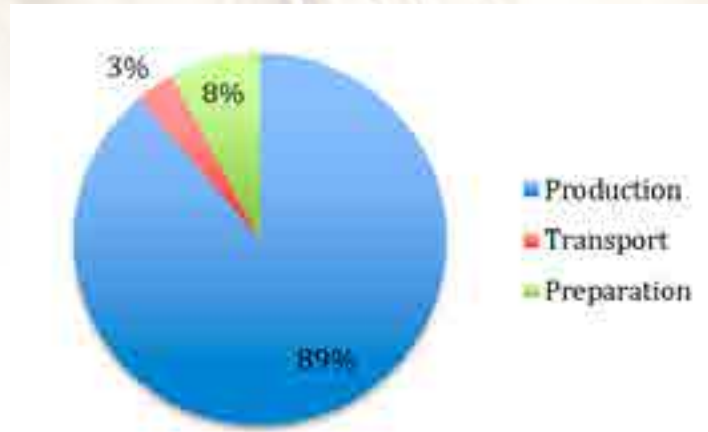


Figure 6: Contribution of each phase to the Carbon Footprint (CF)

3.2 Comparison between a set of meal recipes

The same analysis was performed for seven additional meals suggested by the *School Food Plan Alliance* (<http://www.schoolfoodplan.com/>). All of them comply with the nutritional requirements of the British government (Department for Education, 2015). One half of these recipes are vegetarian (V1 to V4) and the other half contain either meat (M1 to M3) or fish products (F1). The overall numerical results of CF and WF for all eight recipes are presented in Table 2 and Figures 7 and 8.

Table 2: CF and WF of the eight meals analyzed

Meal Name	Code	Carbon Footprint [gCO _{2e} /portion]	Water Footprint [liters/portion]
Mumbai meatballs	M1	1865	501
Sticky chicken and vegetable rice	M2	957	432
Macaroni and cheese with pork	M3	764	720
Salmon and broccoli pasta	F1	382	113
Cheese and broccoli quiche	V1	650	194
Quorn™ curry	V2	458	159
Pizza with vegetable sauce	V3	464	151
Vegetable curry	V4	569	185

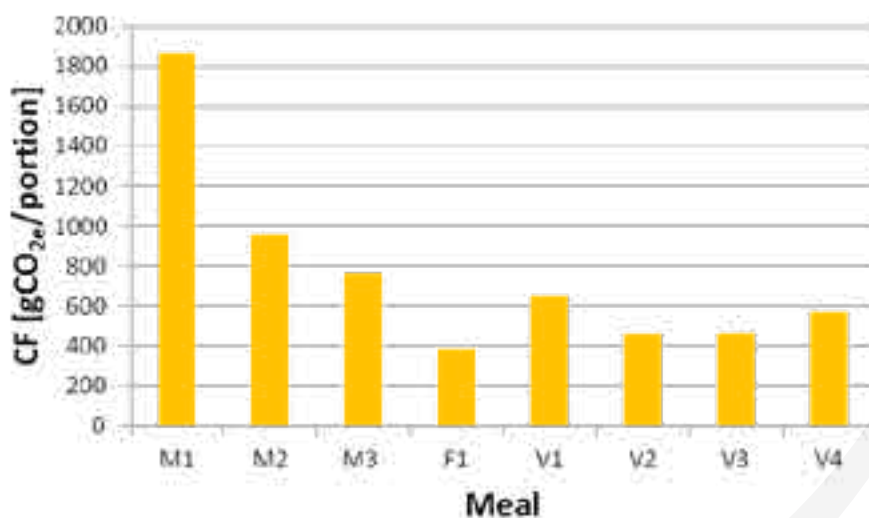


Figure 7: CF of meal recipes analyzed

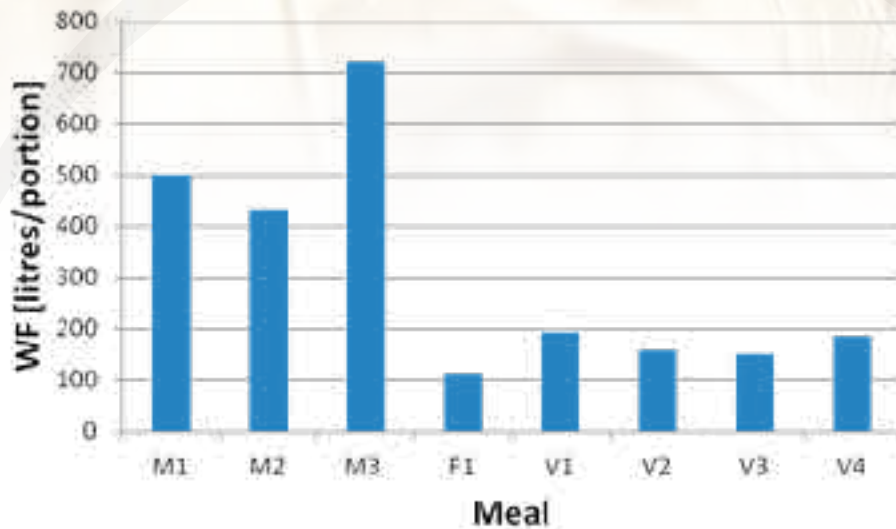


Figure 8: WF of meal recipes analyzed

This analysis shows clearly how the meat-based recipes (M1, M2 and M3) have a higher carbon and water footprint than the non-meat ones. This is in line with a large body of existing research (Audsley et al., 2010, Baroni et al., 2007, Carlsson-Kanyama and Gonzalez, 2009, Davis et al., 2010, Heller et al., 2013, Saxe et al., 2012). In addition the recipe that performs best in terms of CF and WF (F1) is fish-based. From a carbon perspective this is due to the low quantity of carbon intensive dairy products required in the recipe (Pulkkinen et al., 2015). From a water perspective this is due to the fact that the main ingredient, fish, has zero WF, as the freshwater inputs to marine aquaculture and marine capture are considered to be negligible (Hoekstra, 2003, Verdegem et al., 2006).

4. Discussion

4.1. Why is EATS limited to two categories of CF and WF?

The EATS tool considers only two impact categories, CF and WF, and this is due to a number of reasons.

Firstly, the aim of the tool was to create results that are easy for non-LCA experts to interpret. This includes catering staff operators, school staff in charge of choosing the menus and non-scientific audiences. As such EATS allows the concept of carbon and water footprint to be easily explained to students, and provides results that can be used not only to influence menu choices but also with an educational purpose (similar to work done by Saarinen et al. (2012)).

Secondly, as the tool includes a database (of secondary data) collected from literature, the choice of impact categories had to take into account the issue of data availability. Most studies of LCA of food products include amongst their results the impact category GWP (Teixeira, 2015), considered to be most appealing because of its simplicity, which makes it easy to communicate (Weidema et al., 2008). As for WF, an extensive collection of values of water footprint of most food items was published by the leading organization in this field, the Water Footprint Network (Mekonnen and Hoekstra, 2010a, b). This was used as the main source for the water footprint values included in the database.

4.2. How accurate are the values of CF calculated within the EATS database?

There is a great variability in the values of CF of food products, depending on many aspects, including the production method and the production country (Head et al., 2014, Elin Rööös et al., 2014, E. Rööös and Nylinder, 2013, Scholz et al., 2015). One example is the case of vegetables grown in open fields versus heated greenhouses. As shown in a study by González et al. (2011), the latter group has significantly higher carbon emissions, with the value of CF varying according to the type of fuel

used to heat the greenhouse. In the above-mentioned work, the following values of CF (in $\text{gCO}_2\text{/kg}$ product) are reported for cucumbers: 80 for open field, 750 for electrically heated greenhouse and 2600 for fuel heated greenhouse. These variations have to be kept in mind in the interpretation of the results. To record this level of variability in the database, and how this affects the results of the tool, a sensitivity analysis was conducted. To this end, for each food item, the average, minimum and maximum values of CF are recorded in the database.

Figure 9 shows this range for the ingredients in meal F1. The figure shows the high variability in CF related to Salmon. Figure 10 shows the range for each recipe considered within this paper, from which it can be seen that there is high variability (hence uncertainty) for M1 and vice versa for V4. The high value of M1 is in line with the findings of Teixeira (2015), who conducted a statistical analysis of an agri-food database, in which the meals that experienced the largest variations had beef as the main ingredient.

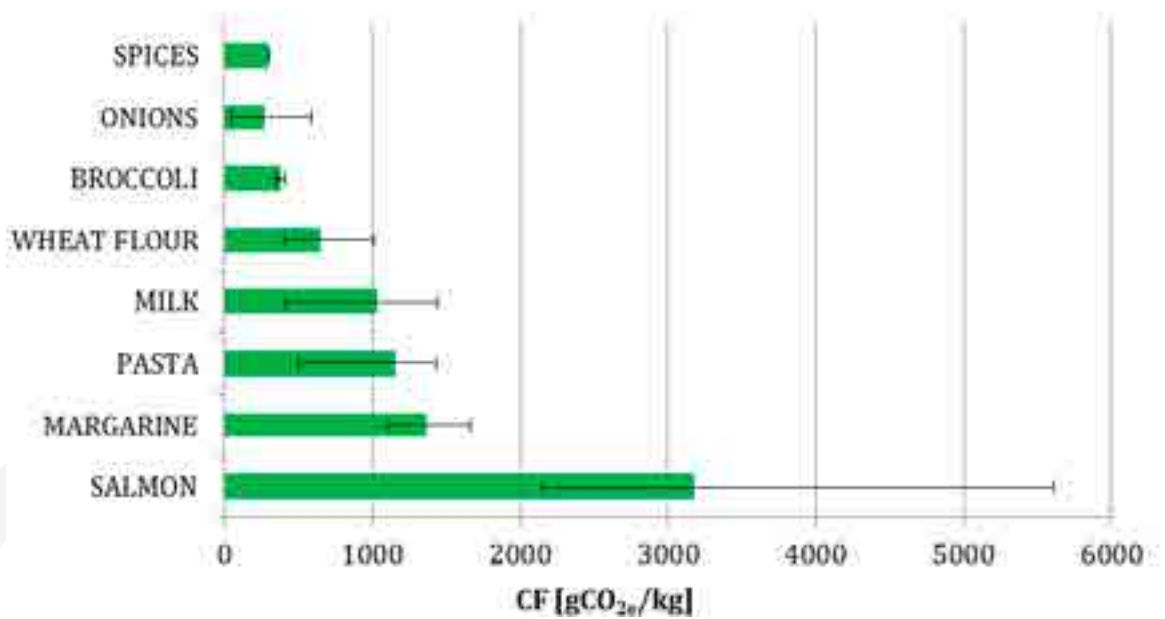


Figure 9: Carbon footprint of the ingredients of meal F1 with min, average and max values.

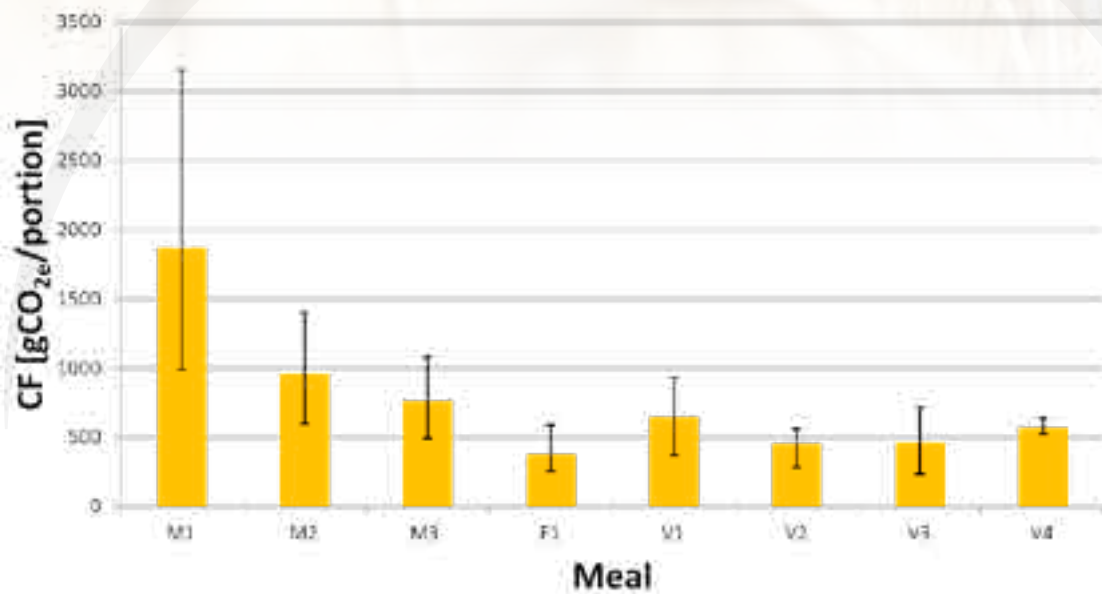


Figure 10: Carbon footprint of the eight meals calculated with average, minimum and maximum value for each meal.

4.3 What is the potential for future impact from the EATS tool?

Thanks to the ease with which users can engage with EATS and be provided with a visualization of the results, there is the potential for significant impact. Firstly in the way school menus are selected and secondly in the way students are introduced to and taught about sustainable food choices. Thirdly as a tool that enables easy identification of *hotspots*, in other words ingredients that are carbon / water intensive. This would allow those responsible for school menus to create alternative recipes, replacing these ingredients with similar ones that perform better from an environmental perspective. Notwithstanding this potential, one should keep in mind that the fundamental importance of school meals is to provide healthy and nourishing food to students. Therefore, EATS could either be used to compare existing recipes deemed to be equal in terms of nutritional value, or to suggest alternative recipes after testing their nutritional quality. This may be a valuable future addition to the tool.

5. Conclusions

This study presents EATS, a tool derived for schools and catering providers in order to self-assess the environmental performance of the menus being served. EATS enables its users to carry out a *cradle to plate* assessment of the carbon footprint and water footprint of a recipe with the purpose of identifying hotspots and suggesting better performing menus. The paper demonstrated the application of the tool to eight meals and compared them in terms of their carbon footprint and water footprint, showing substantial variations amongst them and a general trend of lower impact in the case of meat-free meals. These results prove that an accurate choice of the type of meals served by a caterer can have a significant impact on the overall environmental performance of the service provided.

EATS is a tool that can potentially be used by any school and catering provider in the UK to assess the performance of its menus and identify hotspots amongst its recipes. It does not require any prior knowledge of the LCA methodology and only basic informatics skills to be used. In order to meet these conditions, a number of simplifications had to be made, for instance the reduction of the impact categories to only carbon and water footprint and the use of secondary data as a starting point for the assessment. Hence, it is important to emphasize that it does not represent an alternative to a complete LCA study.

Thanks to a simple user interface EATS can be used to engage non-scientific audiences (including students) on the topic of sustainable food choices, and therefore has great potential both as a tool for decision making (i.e. menu improvement and creation) and education.

Acknowledgements

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214. Life cycle perspective on US dietary patterns

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ABSTRACT

Increasing global food scarcity, trends toward more protein-rich diets and resource competition underscore the importance to examine food production from both life cycle and diet perspectives. Previous studies have mainly focused on the impact of food production on greenhouse gases (GHGs) emissions and have often used European data. This study compares the impact of multiple environmental impacts (GHGs, land use, water use) of current and recommended United States (US) dietary patterns (Omnivore, Plant-Based, Lacto-Ovo and Vegan) using US data. Results show a trend towards increased land use and GHG emissions with diets that include more animal-based proteins. The opposite trend is shown for water use, due to the increased proportion of plant-based proteins such as nuts, many of which have a high water demand. Results from this study move towards holistic diet assessments for the development of evidence-based policies and strategies towards food sustainability.

Keywords: Diet, greenhouse gases, land use, water use, nutrition

1 Introduction

Over the past decade, there has been an increasing interest in using a life cycle assessment (LCA) perspective to address research questions regarding the sustainability of current food systems and dietary patterns. Although sustainability can be defined in multiple ways, a comprehensive view of sustainability encompasses low environmental impact, nutritional adequacy, cultural acceptability, optimization of human resources and economic affordability (Food and Agriculture Organization (FAO) 2010).

The European Union has dominated this research, with over 50% of the studies originating from Europe (Aston et al. 2012; Baroni et al. 2006; Baumann 2013; Berners-Lee et al. 2012; Capone et al. 2013; Fazeni et al. 2011; Gerbens-Leenes et al. 2002; Gerbens-Leenes et al. 2005; Macdiarmid et al. 2012; Masset et al. 2014; Meier et al. 2012; Pradhan et al. 2013; Van Dooren et al. 2014; Sáez-Almendros et al. 2013; Saxe et al. 2013; Scarborough et al. 2012; Temme et al. 2013; Vanham et al. 2013; Vanham 2013; Wolf et al. 2011). The remaining studies have been conducted either in the United States (Buzby et al. 2006; Peters et al. 2012; Peters et al. 2009; Peters et al. 2007; Pimentel et al. 2003; Eshel et al. 2006), Australia (Barosh et al. 2014; Hendrie et al. 2014), Brazil (de Carvalho et al. 2013), New Zealand (Wilson et al. 2013), China (Dong-Dong et al. 2010; Zhen et al. 2010).

In our review of the literature we found that most studies have focused on single environmental outcomes, including GHG emissions (Aston et al. 2012; de Carvalho et al. 2013; Hendrie et al. 2014; Macdiarmid et al. 2012; Meier et al. 2012; Pradhan et al. 2013; Sáez-Almendros et al. 2013; Scarborough et al. 2012; Van Dooren et al. 2014; Vieux et al. 2013; Wilson et al. 2013; Berners-Lee et al. 2012; Norja et al. 2009; Popp et al. 2010; Baumann 2013), land use (Fazeni et al. 2011; W. Gerbens-Leenes et al. 2005; P. Gerbens-Leenes et al. 2002; Temme et al. 2013; Peters et al. 2007; Meier et al. 2012; Zhen et al. 2010; Dong-Dong et al. 2010), and water use (Vanham et al. 2013; Vanham 2013; Capone et al. 2013; Meier et al. 2012).

From the diet perspective, studies have compared sustainability indices across habitual diets, and nationally or regionally recommended diets (Hendrie et al. 2014; Meier et al. 2012; Saxe et al. 2013; Van Dooren et al. 2014; Vanham et al. 2013; Vanham 2013; Sáez-Almendros et

al. 2013; Tukker et al. 2011; Barosh et al. 2014; Buzby et al. 2006; Capone et al. 2013; Fazeni et al. 2011), however, very few of these studies have examined multiple trade-offs between health benefits and environmental indices; these trade-offs move towards answering the question of whether or not these diets are ultimately sustainable.

In 2015 the United States (US) Scientific Advisory Committee for the development of the 2016-2020 Dietary Guidelines for Americans (DGAs) was given the task of reviewing research examining population level diet patterns and long-term food sustainability to inform the new DGAs. This resulted in a set of recommendations for sustainability guidelines that included “a focus on decreasing meat consumption, choosing seafood from non-threatened stocks, eating more plants and plant-based products, reducing energy intake, and reducing waste” (Millen et al. 2015). However, these recommendations were not included in the final DGAs (USDA DHHS 2015). Although there are many similarities between European countries and the US, climate, food production methods, resource availability, and dietary patterns differ. In order to communicate a full picture of the potential impact of diets and impact national level policy changes, further analysis with US data is warranted.

As noted above, US based studies have looked at GHG emissions, land use, water, and fossil energy use, often individually. Buzby et al. (2006) compared American’s habitual diet based on US National Diet Survey (2003-2004) and estimates of land use needs based on an increased consumption to meet 2005 DGAs recommendations for fruit, vegetables, whole grains and dairy. They did not look at any other aspects of sustainability. Pimentel and Pimentel (2013) compared multiple resource indicators of energy equivalent diets: cropland, water and fossil energy, but limited their comparison to a meat-based average US diet and a lacto-ovo vegetarian diet. Peters et al. (2007, 2009, 2012), modeled the potential impact of multiple theoretical diet patterns on GHG emissions, food miles, and land use needs in three studies limited to the New York Region. Lastly, Eshel and Martin (2006) compared Americans’ habitual diet with theoretical diets (lacto-ovo vegetarian, omnivore with fish, omnivore with red meat and omnivore with poultry – all with animal protein), in terms of projected GHG emissions (Eshel et al. 2006).

The purpose of this study is to estimate multiple environmental metrics [land use, water use and greenhouse gas (GHG)] of different dietary patterns, including the current and multiple USDA defined recommended diets. This study is unique in that very few studies have modeled multiple environmental indicators across a variety of USDA recommended diets using US data.

2 Methods

2.1 *USDA Dietary patterns*

In this study, we examine five dietary patterns, including the current US diet and four recommended dietary patterns (Omnivore, Plant-Based, Lacto-Ovo and Vegan) as described in the 2010-2015 Dietary Guidelines for Americans (USDA DHHS 2010). The patterns provide information on daily amounts of foods to eat from five major food groups (Fruits, Vegetables, Grains, Proteins, and Dairy) and their subgroups. As shown in Table 1, there are no changes in many components of the diet among patterns: e.g., the grain recommendations are constant across diets. However, the main changes in the dietary patterns are a shift from animal-based to nut/bean-based protein, and shift, in the vegan diet, from dairy products to a non-dairy alternative. The Omnivore diet includes all animal proteins, fruits, vegetables, grains, dairy, nuts. The Plant-Based diet includes more vegetables and fruit and slightly less animal protein, nuts, seeds and soy products than the Omnivore diet pattern. The Lacto-Ovo diet is a vegetarian diet pattern that includes eggs and dairy. The Vegan diet is a vegetarian diet pattern that does not include eggs or dairy. The Current US diet reflects estimates based on NHANES data (1999). Recommended amounts of foods in the four food patterns depend

on age, gender and physical activity levels, with the 2000-calorie level considered being the average (used in this study). Table 1 includes the actual consumed servings of Americans (Current US) and the recommended serving amounts for each of the four dietary patterns included in the study.

Table 1: Eating pattern comparison: current US diet, omnivore, plant-based, lacto-ovo, vegan food pattern, average daily intake at or adjusted to a 2,000 Calorie level. Units for each group are either c (cups) or oz (ounces).

Food groups (units) & subgroups	Diet Pattern				
	Current US	Omnivore	Plant-Based	Lacto-Ovo	Vegan
<i>Vegetables: total (c)</i>	1.5	2.5	2.5	2.5	2.5
Dark-green	0.1	0.2	0.2	0.2	0.2
Beans and peas	0.1	0.2	0.2	0.2	0.2
Red and orange	0.4	0.8	0.8	0.8	0.8
Other	0.5	0.6	0.6	0.6	0.6
Starchy	0.4	0.7	0.7	0.7	0.7
<i>Fruit and juices (c)</i>	1.1	2.0	2.0	2.0	2.0
<i>Grains: total (oz)</i>	6.3	6.0	6.0	6.0	6.0
Refined grains	5.5	3.0	3	3	3
Whole grains	0.8	3.0	3.0	3.0	3.0
<i>Dairy and non-dairy (c)</i>					
Dairy	1.5	3.0	3.0	3.0	0.0
Non-dairy, calcium fortified	0	0	0	0	3.0
<i>Protein foods (oz)</i>					
Meat	2.5	1.8	0.6	0.0	0.0
Poultry	1.2	1.5	0.4	0.0	0.0
Eggs	0.4	0.4	0.4	0.6	0.0
Fish/seafood	0.5	1.2	0.7	0.0	0.0
Beans and peas	0	0	1.4	1.4	1.9
Nuts, seeds, and soy products	0.5	0.6	2.0	3.6	3.6

2.2 Environmental Data

Data are taken from sources specific to each environmental variable. These sources were chosen to reflect spatial variability where possible, and also to provide a wide coverage of foods. Data for directly produced commodities (e.g., vegetables or grains) are described below, followed by a discussion of how data for secondary (e.g., livestock which is fed primary commodities) or processed (e.g., soymilk) foods are derived from primary information.

Land use data are taken from the US Census (USDA NASS 2009), which collects and reports a variety of data related to production of various crops across US states. In this case, total production and total acres harvested are used to calculate state specific values for yield and its inverse (m^2 / kg). Because yield varies across the country as a function of climate, cropping practices, etc., production was used as a weight to calculate national values.

Although the USDA NASS reports irrigation statistics in the Farm and Ranch Irrigation Survey (FRIS) (USDA NASS 2010), these data are offset from the census data by year, have reduced geographic coverage, and also have reduced crop coverage. Therefore, we use irrigation data from Pfister and Bayer (2014), who modeled 160 crops at a resolution of 5 arc minutes (~10 km at mid latitudes), based on CROPWAT (FAO 1999).

The importance of correctly capturing national values using weighting is illustrated by the use of irrigation water, which varies more from state to state than does land use. We consider water consumption for corn production: an arithmetic average of state water consumption for corn yields a value of 280 L water consumed / kg corn grain. However, a production – weighted average of the same data is 61 L water consumed / kg corn grain.

Finally, greenhouse gas data are taken from Heller and Keoleian (2014). This survey of greenhouse gas data associated with food production and consumption draws from a variety of LCA sources. Although greenhouse gas emissions associated with food production will indeed vary from country to country and, within the US, state to state, these variations are likely to be minimal relative to water use.

Data for livestock are either calculated or taken directly from the literature. In the case of dairy production, we combine a survey-based set of rations (kg feed / kg milk) (Thoma et al. 2013) to calculate the total land use and water use associated with production of the feeds necessary to produce a kilogram of milk. Land and water for beef production are taken directly from Capper (2011). Because of its importance in the non-lacto diets, soymilk was calculated specifically as a combination of soybeans and water, following the data provided in Birgersson et al. (2009).

This analysis is focused on a mix of inventory (land use and water use) and impact assessment (greenhouse gases). Certainly, it will be useful in future work to expand the analysis to take land and water all the way to impact assessment (e.g., Teillard et al. 2016). However, much land use in the U.S. is relatively stable, thus avoiding some of the major land-use impacts that come from conversion of land. Secondly, many of the non stress-based water endpoints relative to human health are also not as applicable in the U.S. As the LCA community is still developing ways to measure endpoint ecosystem impacts (Boulay et al. 2015), we have decided to keep our analysis at the inventory level.

2.3 Relationship of Environmental Data to LCA Data

As described above, the land use and water use data for feed production are not truly life cycle data. These data only capture environmental inventory that occurs on farm, and do not capture post farm stages, such as processing, distribution, or consumer. However, studies have indicated that, for these two inventory items, the on-farm stage is by far the dominant contributor to overall life cycle inventory flows (Meier and Christen 2012). In contrast, greenhouse gas emissions occur at several lifecycle stages, such as transportation and refrigeration, in addition to on-farm emissions associated with fertilizer. Therefore, for the GHG data, it is important to consider full life cycle data, rather than simply on farm. Overall, the data presented in this paper are expected to capture the main trends and differences between foods across all three environmental inventory categories. Any systematic errors, such as underestimation of water required to process vegetables, are expected to be rather small relative to changes in diet. Future work will systematically investigate this assumption.

2.4 Combining Environmental & Diet Data

Diet data are presented in terms of groups and subgroups of food, such as vegetables, which are broken down into leafy green vegetables, dark green vegetables, starchy vegetables, etc.

(Table 1). In order to connect these groups and subgroups to the specific commodities and foods for which environmental data are available, we use the following approach. First, subgroups are broken down further according to typical US consumption patterns (Marcoe et al. 2006). Data about US consumption make it possible to convert the vegetable subgroup ‘dark green vegetables’ into specific foods and a consumption fraction, such as broccoli (35.8% of dark-green vegetable consumption), romaine lettuce (27.1%), mustard greens (1.4%), etc. In this manner, specific foods’ environmental data can be aggregated to a subgroup and group level.

Second, we note that not all of the foods specified in the consumption patterns are reported in a given environmental inventory database. For example, the USDA Census may report data for broccoli and lettuce, but not other dark green vegetables. In priority, we look for reasonable substitutes (e.g., lettuce may be used as a substitute for romaine lettuce). When choosing a substitute is not feasible, we use a production-weighted average of the (sub)group of members that are reported. In this way, a diet pattern can be decomposed into specific food items for which substitutes or (sub)group averages are available. The environmental inventory data associated with these specific food items are then summed within groups and aggregated back to the diet level.

3 Results

Figure 1 presents results at the inventory level for the five dietary patterns presented in Table 1. For land use (Figure 1, top) the Current US diet had the highest land-use requirement, with the Omnivore, Plant-based, Lacto-ovo, and Vegan having sequentially lower demands, down to approximately 60% of the Current US diet. A similar trend is seen for greenhouse gas emissions (Figure 1, bottom), with the exception that the Omnivore diet has the largest impact on GHG emission. For GHG emissions, the impact of the Vegan diet is approximately 50% less than the impact of the Current US diet.

In contrast, water use associated with the different dietary patterns (Figure 1, middle), shows a different trend. The Lacto-ovo diet has the largest water use demands, with the Current US diet being approximately 55% of the Lacto-ovo demands.

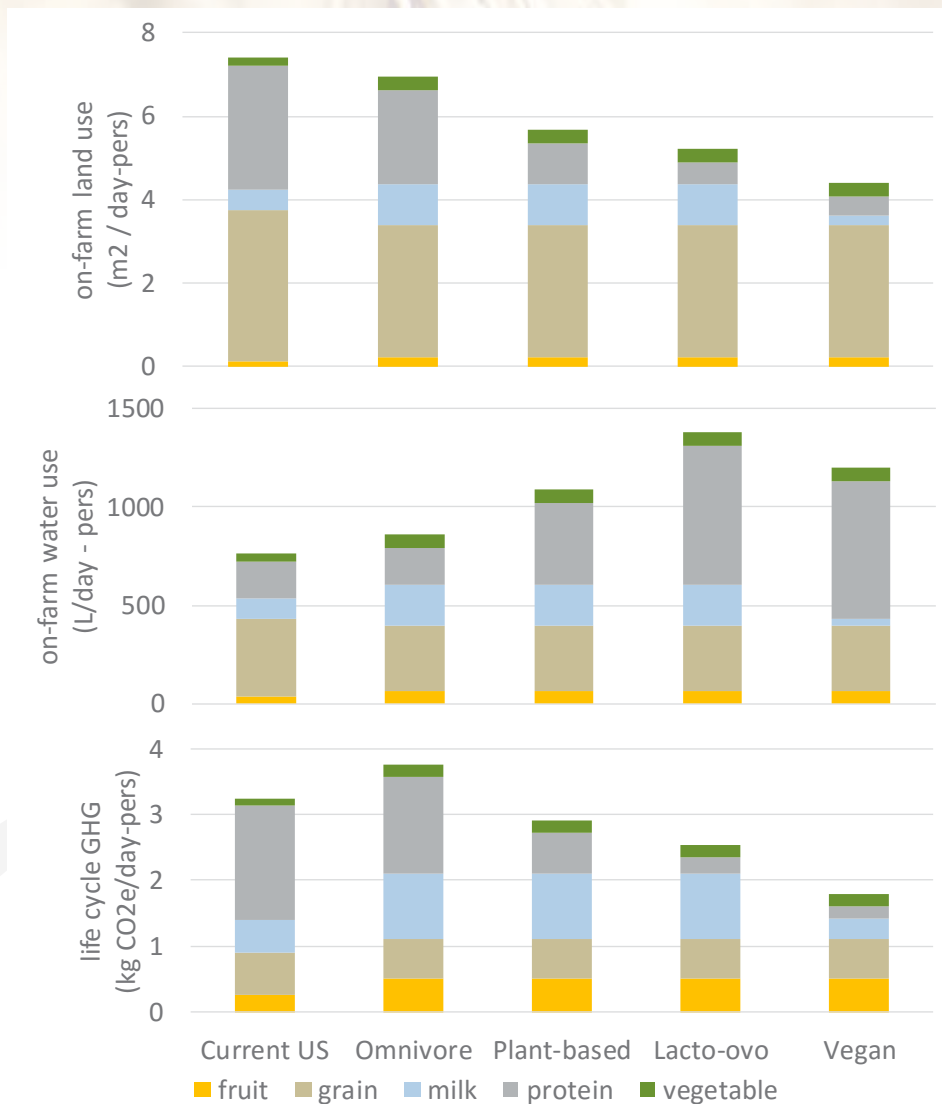


Figure 1: Inventory results for Current US diet and 4 recommended diets. Land use (top), water use (middle), and GHG (bottom).

While Figure 1 presents results at the diet level, it is also instructive to look at individual macro and micro-level nutritional components of foods. Figure 2 shows such result for land use, comparing the five food groups land use requirements for energy, protein, and calcium. In such an analysis, higher values represent more ‘productive’ uses of land. For example, grains produce the most calories per square meter, but also have the second-lowest production of calcium per area. Dairy products have the highest production of protein and calcium per area, but is only moderate with respect to energy production. This comparison shows the importance of considering all aspects of a diet.

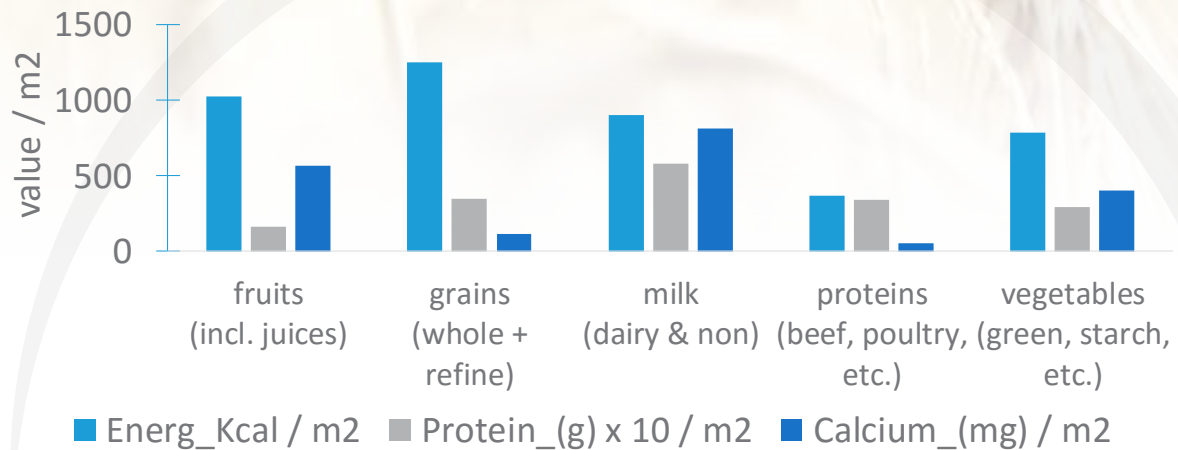


Figure 2: Land use requirements of individual components of nutrition (energy, protein, calcium) for the five major food groups in the Dietary Guidelines.

4 Discussion

The purpose of this study was to estimate the land use and water use demands, as well as greenhouse gas (GHG) emissions, associated with five US dietary patterns. As noted above, the main changes in the dietary patterns are a shift from meat-based to nut/bean-based protein, and shift, in the vegan diet, from dairy milk to a non-dairy alternative. Results indicate that the impact of the five different dietary patterns vary by environmental indicator. For example, dietary patterns composed of greater percentages of animal-based proteins use more land and emit more GHGs. Consequently, as the proportion of animal protein in the diet decreases, land use and GHG emissions also decrease. These results are consistent with those from previous studies (e.g., Netherlands: Van Dooren et al. (2014), Germany: Meier et al. (2013), US: Pimentel et al. (2013)).

The decreases in GHGs were due mainly to the differing amounts of protein and dairy in the diet. For protein sources, beef (discussed here only as an example of animal-based protein sources) has a land use requirement of approximately 50 m²/kg, whereas nuts have lower requirements (almonds ~5; walnuts ~3.5 m² / kg). This relationship is similar for greenhouse gases, where beef emissions are ~25 kg CO₂e / kg, and nuts are ~1.7 kg CO₂e / kg. For dairy, fluid milk requires 1.3 m²/kg and 1.3 kg CO₂e / kg, whereas soy milk requires 0.62 m² / kg and 0.9 kg CO₂e / kg.

Results show an opposite trend with on-farm water use, underscoring the importance of considering multiple environmental variables. Again, the shift in protein and dairy sources accounts for the upward trend in water use for diets with more plant-based foods. For water, beef requires approximately 1500 L / kg, while almonds require 3200 L / kg. The vegan diet uses slightly (13%) less water than the lacto-ovo diet because dairy milk requires 181 L / kg, and soy milk requires 90 L / kg. More than half of the relative contribution to on-farm water use of the Lacto-Ovo and Vegan diets can be attributed to the protein and dairy food groups. Specifically, nuts and seeds use the most water in these diets. Meier & Christian (2013) also found that increased water use is associated with higher fruits, nuts, and seeds.

A diet level perspective is an important complement to mass- or nutrient-based food analyses. As shown in Figure 2 (which is just for land use), an individual food may be highly efficient with respect to one component of the overall diet, and less so with respect to another.

Therefore, it is critical to analyze individual foods as well as overall dietary patterns. This sort of analysis allows one to consider how environmental demands might change if we were

to modify food groups' relative contribution to the diet sources while meeting nutritional requirements.

A diet-level perspective is also important with respect to public health implications. While diets are devised to balance a variety of caloric, nutritional, and cultural needs, there are both positive and negative health impacts to be considered. For example, Aston et al. (2012) found that if the number of vegetarians in the UK doubled and all others adopted the diet of the lowest red, processed meat quintile, there would be up to a 12% lower incidence of coronary heart disease, diabetes mellitus, and colorectal cancer.

5 Conclusions

This study quantified land use, water use, and greenhouse gas emissions associated with five US-based diets. With protein and dairy sources driving changes in the three environmental inventory variables measured in this study, the plant-based diet appears to have intermediate environmental inventory. Thus, an initial evaluation would indicate that this dietary pattern provides a balance between land-intensive proteins and water-intensive proteins and may be the dietary pattern to recommend for sustainability purposes. While the inclusion of three environmental variables has identified some tradeoffs, this study clearly points to the need for more holistic assessments.

Specifically, there are several points to consider in interpreting these data. First, we have not yet attempted to quantify uncertainty, so future work will determine whether the differences we see are meaningful. We have not presented a formal analysis of uncertainty for two reasons. First, the data sources for water (Pfister et al. 2014) and greenhouse gases (Heller et al. 2014) both state that it was not possible to calculate uncertainty formally. While we can estimate uncertainty on the USDA land-use data at less than 10% error, we cannot do this consistently across the data. Secondly, there is the challenge of linked processes. For example, if vegetable consumption land-use is underestimated for one diet, it will be underestimated for all diets. Therefore, putting error bars on the stacked bar graphs comparing diets could potentially be misleading, because the differences between these will not be captured with such error bars. In future work, a full Monte Carlo analysis with dependent and independent variables will allow us to better consider uncertainty.

Second, in this study we only assessed three environmental inventory variables. We have not captured information about many impact categories typically assessed in LCA, such as losses of nitrogen/phosphorus, particulate matter emissions, etc. Therefore, it is possible that any ranking of diets would change with the inclusion of more variables.

Third, because this LCA study only captures environmental inventory flows, we do not have information about potential impacts (e.g., biodiversity losses due to land or water use) caused by these flows. If, for example, the biodiversity characterization factor for water use was significantly higher than that for land use, then impact level assessment (e.g., biodiversity loss to the land and water use) would show that the water level impacts could be much more important in land use impacts. Furthermore, for those impact categories with spatially variable impacts, an impact analysis could identify hotspot production regions within the US. Finally, this study does not attempt to evaluate absolute sustainability, nor does it attempt to combine the environmental variables into a weighted metric. The assignment of weights to environmental variables is a question of stakeholder values, and the question of sustainability depends on the scale of analysis. For example, if water-intensive crop production were to increase in certain areas of the US, there would indeed be localized water stresses - and political problems in addition to the associated environmental stresses. However, at a larger scale, an analysis (based on stakeholder input) of the entire country could show that water stress is likely not a problem, given the resources of the Great Lakes. Globally, further study

will certainly be needed to provide perspectives on the question of whether food production systems and diets can be sustained or should be changed – and whether the environmental and political consequences of those practices are acceptable.

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141. A life cycle approach to modeling nutritionally balanced, climate-friendly and socially viable dietary patterns in Ontario, Canada.

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ABSTRACT

A growing body of literature demonstrates the broad impacts of food consumption on health and climate. Research to inform feasible dietary recommendations that reduce environmental impact while also promoting public health is urgently needed.

Prior work suggested that the two most popular dietary patterns among residents of Ontario, Canada have the largest Global Warming Potential (GWP) on a life cycle basis (Veeramani, Dias & Kirkpatrick, 2016). Commonly-consumed dietary patterns are excessive in calories and protein, particularly from animal sources, with the overconsumption of beef protein driving higher GWP. Building on this work, we employed simulations to assess the life cycle impacts on GWP of dietary changes intended to achieve nutritionally balanced, climate-friendly and socially viable dietary patterns. We considered seven food baskets that represent common dietary patterns among 10,723 Ontarians. Health Canada's Food Guide was used to assess nutritional quality and inform proposed changes to optimize consumption of key food groups (Health Canada, n.d.-a). Calorie and protein content was adjusted to the age and gender-weighted levels recommended by Health Canada (Health Canada, n.d.-b). Contribution analysis results from Veeramani et al. (2016) were used as a reference for minimizing the intake of foods with high GWP and substituting them with alternatives having lower GWP. To ensure the social viability of proposed changes, the types of food were maintained, but the amounts of the food items with the highest GWP were reduced by up to 50% of initial consumption levels.

The proposed dietary changes resulted in 5 to 34% reduction in GWP, with the largest reduction potential demonstrated by the two most popular diets, consumed by over 50% of Ontarians. Incorporating sustainability indicators such as GWP in dietary guidelines has tremendous potential to promote both public health and climate change mitigation.

Keywords: greenhouse gas emissions; LCA; dietary guidelines; dietary patterns; protein

1. Introduction

Over recent years, the sustainability of food consumption has been an area of increasing attention. A growing body of research provides insights into the environmental implications of the food system, including contributions to climate change, land use, water consumption, eutrophication and acidification potentials, and others (e.g. Carlsson-Kanyama, 1998; Sonesson, Mattsson, Nybrant, & Ohlsson, 2005; Baroni, Cenci, Tettamanti, & Berati, 2007; Jungbluth et al., 2000; Kramer, Moll, Nonhebel, & Wilting, 1999; Macdiarmid, 2013; Meier & Christen, 2012a, 2012b; Muñoz, Milà i Canals, & Fernández-Alba, 2010; Saxe, Larsen, & Mogensen, 2013; Vieux, Darmon, Touazi, & Soler, 2012). Debates and dialogues regarding sustainable food consumption also incorporate considerations regarding the interconnections between sustainability and health, particularly nutrition (FAO, 2010b). A sustainable diet presupposes low environmental impacts along with food and nutrition security as well as the promotion of health among current and future generations (FAO, 2010). Thus, studies increasingly examine nutritional quality along with environmental impacts to define a healthy and sustainable diet (Van Dooren et al., 2014).

Epidemiological studies indicate a strong link between dietary choices and health. Medical conditions such as type II diabetes, cardiovascular diseases, stroke, cancer, Parkinson's disease, hypertension, obesity, or some foodborne illnesses are often connected to consumption of animal-based diets (Barnard, Nicholson, & Howard, 1995; Goodland, 1997; Sabaté, 2003). High saturated fat, which is high in meat, cheese, milk, butter and eggs, along with high sodium intake can potentially increase the risk of cardiovascular diseases (Wilson et al., 2013; Hu et al., 1999). Dietary animal protein has been linked to various types of cancer (Campbell and Campbell, 2005; Youngman and Campbell, 1992; Schulsinger, Root and Campbell, 1989; Larsson, Bergkvist, & Wolk, 2004; Sieri et al., 2002; Chao et al., 2005; Fraser, 1999), hypertension, heart disease and gallbladder disease (Barnard et al., 1995), kidney stones (Breslau, Brinkley, Hill, & Pak, 1988), obesity and diabetes (Barnard et al., 1995), increased aging bone loss and hip fractures (Lanham-New, Lee, Torgerson, & Millward, 2007), Crohn's disease (Shoda, Matsueda, Yamato, & Umeda, 1996) and other NCDs. Plant-based foods, on the other hand, have been associated with fewer diseases and lower

mortality rates (Dunn-Emke et al., 2005; McCarty, 1999, 2001; Turner-McGrievy et al., 2008). Plant-based protein sources are found to reduce the risk of cancer, obesity, and cardiovascular diseases (McCarty, 1999). The World Health Organization's dietary recommendations suggest reduction of fat, animal-based foods and higher intake of fruits and vegetables which is believed to have a positive impact on both health and the environment in terms of diet-related sustainability (Reynolds et al. 2014). However, country-specific dietary recommendations should be studied to investigate health and environmental implications of food consumption in various countries.

Based on patterns of consumption observed using dietary intake data for residents of Ontario, Canada, the primary goal of the present study was to model nutritionally-balance, climate-friendly and feasible versions of existing dietary patterns and to assess the potential implications in terms of the implications for Global Warming Potential (GWP) using Life Cycle Assessment (LCA). A distinct feature of the Ontario province is the diversity of population, with over three quarters of its residents coming from diverse ethnic and cultural backgrounds (Ontario, 2011, n.d.). Diversity of the population is likely to entail significant variations in dietary preferences; thus, the proposed changes must be socially viable and realistic, in addition to aligning with dietary recommendations based on the evidence on nutrition and health.

2. Methods (or Goal and Scope)

The objectives of this paper include 1) assessing the nutritional quality of current dietary patterns in Ontario, with a particular focus on the two most popular dietary choices (Omnivorous diet and Omnivorous diet excluding pork - 'No Pork'), 2) modeling environmentally friendly versions of existing dietary patterns that meet dietary guidelines in place in Canada, and 3) quantifying potential changes in the carbon footprint associated with a transition to nutritionally balanced, climate-friendly and socially viable dietary patterns.

Seven types of diets (or dietary patterns) were identified in Ontario based on the actual one-day food reports of 10,723 Ontario residents. These included vegan, vegetarian, pescetarian and omnivorous diets as well as diets excluding red meat, pork or beef (Veeramani et al., 2016). The nutritional value of each dietary pattern was assessed based on the Canada's Food Guide and Health Canada's recommendations on calorie and protein intake. This information along with a prior examination of LCA associated with the diet patterns (Veeramani et al., 2016) was used to model nutritionally balanced, climate-friendly and socially viable diets.

Each of the seven dietary patterns was expressed as an annual food basket, representing the typical food consumption of a person exhibiting a particular dietary pattern (Veeramani et al., 2016). The food baskets were then modified to optimize their nutritional value while maintaining the key features of the dietary pattern and minimizing the associated carbon footprint. The changes included an increase or a reduction of calories and protein as well as adjustments to amounts of key food groups (fruit and vegetables, milk and alternatives, grain products and meat and alternatives) to achieve the optimal levels recommended by Canada's Food Guide; and substitution of high-impact food items such as cheese, butter, beef and other meat and dairy products to more environmentally favorable alternatives (**Figure 1**). To ensure social viability of the proposed changes, refinements were made for all seven dietary patterns to ensure that the key food groups and protein sources were neither eliminated nor reduced by more than 50%.

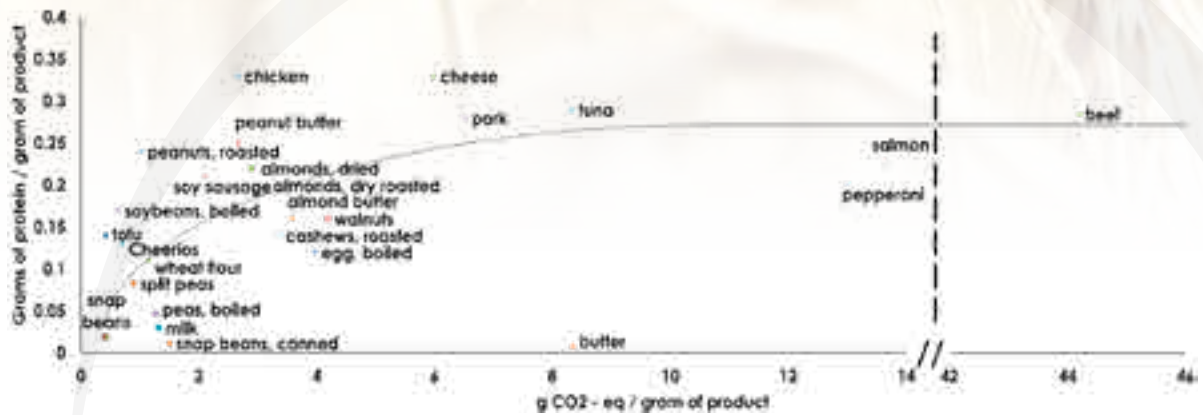


Figure 1: Protein sources and corresponding GWP, calculated by Veeramani et al. (2016) and used as a reference in formulating a nutritionally balanced, climate-friendly and socially viable diet.

The methodology described by Veeramani et al. (2016) was used to estimate the carbon footprint of the original and modified dietary patterns. LCA was carried out according to the ISO14040/14044 (2006) standards. The modeling was performed in SimaPro v. 8.0.2 software and carbon footprint was analyzed using the IPCC Global Warming Potential (GWP) 100-year method (IPCC 2007 GWP 100a V1.02). The food baskets representing seven dietary patterns served as the functional units in the analysis. The LCA of all foods and beverages in each of the food baskets was conducted on a farm-to-fork basis, encompassing raw material extraction, processing, farm-based activities, transportation to processing facilities and retail, processing, packaging, household transportation and other processes, including storage, food preparation and dishwashing.

3. Results (or LCI)

Results of the nutritional assessment indicated that all seven dietary patterns were initially unbalanced with regard to calorie and protein intake and consumption of key food groups. The two diet patterns common to over 50% of Ontario residents ('No Pork' and 'Omnivorous') contained up to 250% of the recommended protein level. The 'Omnivorous' food basket, which was consumed by 30% of the population on the day for which intake was reported, exceeded the optimal calorie level by 20%. The diets also largely lacked the recommended servings of fruit and vegetables, grain products and fish. These two most common diets also had the highest GWP (Veeramani et al., 2016).

Modeling of the food baskets to align more closely with dietary recommendations and reduce the consumption of high-impact foods resulted in significantly lower GWP for almost all dietary patterns. **Figure 2** demonstrates the potential changes in the GWP for each dietary pattern, ranging from 17% increase in GWP to 34% GWP reduction. The largest reduction potential was demonstrated for the two most popular diets ('No Pork' and 'Omnivorous'). The GWP was reduced by more than a third in both diets by increasing the consumption of fruit and vegetables, grain products and milk, and halving the consumption of high-impact products such as meat (particularly beef), butter, cheese and egg.

The only exception to the overall reduction trend was the 'Vegan' diet. Its GWP increased by 17%. This was primarily caused by the low calorie and protein content of the original food basket, which was half the recommended level, and a low intake of key food groups at around 60% of recommended values. Thus, increasing the content and thus the weight of the basket and adjusting the calories to the recommended level resulted in the overall increase of GWP. It should be noted that few Ontarians consumed a 'Vegan' pattern on the day that food intake was recorded and so there is limited food consumption data on which to base the nutritional assessment and subsequent LCA (Veeramani et al., 2016).

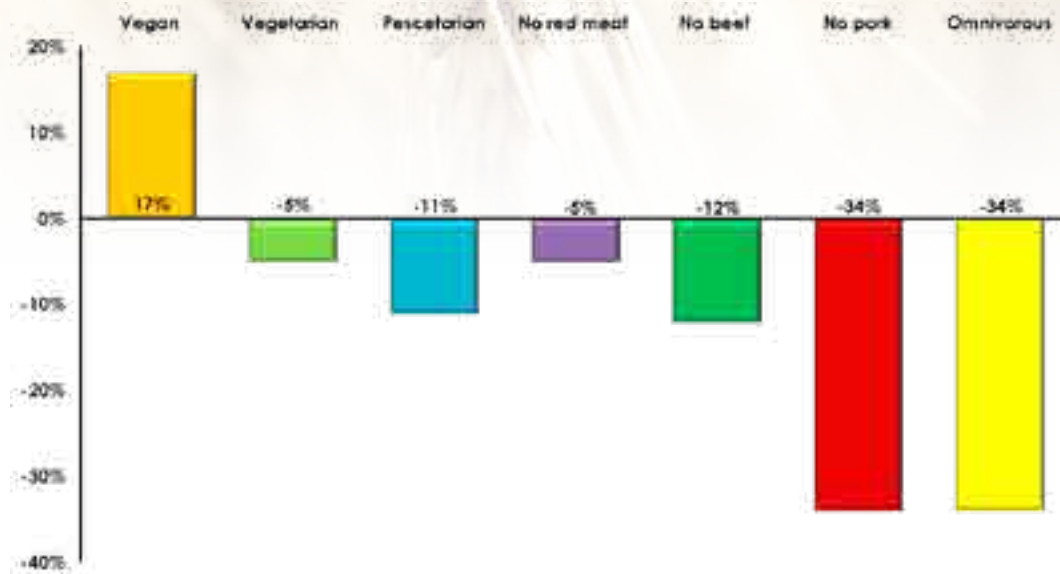


Figure 2: Potential changes in the GWP (%) associated with the transition to nutritionally balanced, climate-friendly and socially viable dietary patterns.

4. Discussion (or LCIA)

The present study contributes to an understanding of the nutritional quality of current dietary patterns in Ontario and a potential reduction in carbon footprint associated observed with a transition to nutritionally-balanced and climate-friendly diets. The interpretation of the results and insights into recommendations for research and policy should consider limitations of this study and their implications for reliability and generalizability of results.

The findings of the nutritional assessment may be affected by the composition of the food baskets, consumed amounts and recommended intakes, choice of reference for assessment, and population distribution within each basket. Importantly, the initial food baskets were based on reported diets for a single day whereas what we eat and drink is known to vary from day-to-day. The baseline consumption within each dietary pattern was based on the average amounts of consumed foods. However, the baseline average consumption provided sufficient basis for assessing the adequacy of the dietary pattern as a whole. Along with analyzing the key macronutrients such as protein, the study would benefit from additional assessment of macro-minerals and vitamins, and micronutrients. A more comprehensive nutritional assessment would need to incorporate the use of supplements to accurately assess the nutritional adequacy of the current dietary preferences in Ontario. Inclusion of the supplements in the analysis is also likely to affect the results of the LCA given the additional environmental impact associated with the production and the consumption of the dietary supplements.

The choice of the nutritional guideline used could have affected the modeling of nutritionally balanced food baskets and potentially affected the results of the subsequent LCA and proposed recommendations coming out of this work. Although Canada's Food Guide is a national dietary guideline, the dietary patterns can also be assessed against other standards with varying results.

There are also limitations in assessing the carbon footprint of healthy and climate-friendly diets associated with the LCA methodology, described by Veeramani et al. (2016).

5. Conclusions (or Interpretation)

The fundamental aspects of diets that are consistent with the evidence on nutrition and health are often presented and promoted through national dietary guidelines, such as the Dietary Guidelines for Americans, Australian Dietary Guidelines, Nordic Nutritional Recommendations, D-A-CH in Germany, Austria and Switzerland or UGB, Dutch Dietary Guidelines and Canada's Food Guide. However, at the present time, there is little incorporation of considerations regarding sustainability

into such guidelines (Merrigan et al., 2015). This study suggests that aligning diets more closely with Canada's Food Guide have the potential to reduce GWP while also promoting health.

This study contributes to interdisciplinary research supporting the nexus of nutritional and environmental sciences and policy-making. Food consumption has multidimensional implications ranging from nutrition and health, environment and food security to the agricultural traditions and innovations. Thus, research and related policy-making also need to be multidisciplinary to secure nutritional and food security and environmental sustainability. Diet-related research should facilitate and promote development of sustainable dietary guidelines in Canada and elsewhere through collaboration of nutritionists and environment professionals.

In addition to previous work by Veeramani et al. (2016), the present study provides a framework for identifying dietary patterns, assessing their nutritional quality and carbon footprint, and estimating potential impact reduction from transitioning to healthy, climate-friendly and socially-viable diets. The framework can be applied to assess the environmental footprint and nutritional adequacy of food consumption across Canada or in other jurisdictions and to inform diet-related changes to minimize environmental impact and improve nutrition and health.

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302. Environmental Impact Reduction from Integration of Plant-Based Hamburger Production to Replace Animal Farming

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ABSTRACT

Impossible Foods has developed a process to produce a completely plant-based replacement for ground beef with comparable or improved taste, nutrition, cost, and life cycle attributes to its animal-based counterpart. We have developed a rigorous Excel-based life cycle assessment (LCA) framework to assess the potential environmental consequences of replacing conventional cow-derived ground beef with this plant-based beef product. Replacing animal-based ground beef with an equal mass of the plant-based Impossible Foods (IF) ground beef would result in 86-91% less blue water used, 61-94% less green water used, 76-90% less greenhouse gas emissions, and 91-97% less land usage. By developing this single product into a portfolio of replacement products to those originating from animal farming, dramatic decreases in water usage, land usage, and greenhouse gas emissions can be realized.

Keywords: beef, alternative protein, land use, water use, carbon footprint

1. Introduction

Animal farming supplies meat and dairy products that are important components of the human diet around the world. Yet these products exert an outsize environmental impact. With per capita consumption of meat and dairy projected to increase with purchasing power growth and an expanding global population, dietary patterns represent a promising mitigation target for resource use. Despite the resource-use intensity of animal products, per capita meat consumption is expected to grow by 25% and dairy by 20% worldwide for a net meat production increase of over 70% as purchasing power increases in the developing world (Alexandratos and Bruinsma 2012 p. 44).

The livestock industry supplies only about 20% of global calories but occupies almost one third of the world's arable land area, nearly 40% of US domestic land area, and around 70% of all agricultural land globally (Delgado et al 1998; Steinfeld et al. 2006; Eshel et al. 2014). Globally, the sector generates 7.1 gigatons of CO₂eq, about the same amount of emissions as the entire transportation industry, predominantly as CO₂, CH₄, and N₂O (Gerber et al. 2013). Ruminants such as cows are the primary culprit, contributing 65% of the sector's emissions (Opio et al. 2013). Meat and dairy production together constitute almost 30% of humanity's global water footprint, as measured in terms of volume of freshwater appropriated throughout the supply chain (Hoekstra and Mekonnen 2012). Ninety eight percent of that 2422 Gm³/yr produces the feedcrops consumed by animals, with relatively little devoted to service water (Mekonnen and Hoekstra 2012).

Use of cattle to produce beef is a ubiquitous yet inefficient converter of energy to calories and protein, requiring 13 kg of feed to produce one kg of beef and an energy input to protein ratio of 40:1 (Pimentel and Pimentel 2003). By some estimates, production of beef in the US requires 88% of domestic agricultural land but contributes just 7% of all consumed calories per capita. These inefficiencies result in part from the use of energy by cattle for general metabolic purposes and conversion into both edible and nonedible tissues (Baroni et al. 2007). Ground beef makes up the largest fraction of total US beef consumption at around 50% (Beef Checkoff). Due to high per capita consumption and the disproportionately large environmental impact of beef, consumer shifts to consumption of a plant-based ground beef alternative confers a large environmental benefit.

A growing body of literature has addressed the environmental impact of cattle farming to produce beef, but few studies have investigated the relative efficiencies of novel protein alternatives. This Life Cycle Assessment (LCA) was developed to provide a comparison of environmental metrics between ground beef derived from cattle against the newly developed plant-based burger. In order to ensure methodological rigor and quality of results, sustainability consulting group Quantis has reviewed the IF LCA model. The review entailed importing IF's Excel-based life cycle inventory data into SimaPro v8.2 software and modeling and results from the IMPACT 2002+ framework. This review was

intended to provide high level vetting of IF results but was not intended to provide ISO compliance at this stage. For context, the results of the SimaPro model are presented along with the Excel model.

2. Methods

An attributional life cycle assessment has been constructed to assess the environmental impact of the Impossible Foods plant-based ground beef product for comparison against the published metrics for animal based beef. The system boundaries of the assessment start from the “cradle”, or the acquisition of raw materials, and terminate at the transportation of the finished good to retailers and/or food service. Distribution networks and end-stage preparation are assumed to be similar to those of cattle-derived ground beef and therefore are expected to have no significant differential impact for this comparative analysis.

Development of the functional unit

The goal of generating a Life Cycle Assessment of the alternative beef product was two-fold. First, the LCA facilitates identification of environmental hotspots across the supply chain. Second, it allows for a relative assessment of environmental impacts across different protein sources (specifically, the selected functional unit allows for a comparison to beef from the traditional cow-based production system).

In many food-based LCAs, the functional unit is based on the nutritional role of food as a vehicle for energy or protein. This permits comparison between disparate food types serving similar primary functions (Heller et al, 2013; Schau and Fet, 2008). The primary function of the IF product is to replicate animal-derived ground beef in all nutritional and culinary applications. Because this paper evaluates the IF ground beef product as an alternative production system to animal-based beef, and because the nutritional profiles of the two products diverge little, the functional unit employed here is mass-based.

Nutritional summary of animal-based ground beef and IF ground beef product			
	Unit	Value, Beef	Value, IF Burger
Water	g	656.6	531.2
Energy	kcal	2150	2600
Protein	g	185.9	227.2
Total lipid (Fat)	g	120	158
CHOs, by difference	g	0	67.5
Fiber	g	0	0
Sugars	g	0	0
Ca	mg	150	245
Fe	mg	20.9	33.5
Mg	mg	180	123
P	mg	1710	1220
K	mg	2950	1820
Na	mg	660	6405
Zn	mg	44.8	36.7
Thiamin	mg	0.42	270
Riboflavin	mg	1.5	1.8
Niacin	mg	46.49	57
Vitamin B-6	mg	2.46	3.2
Vitamin B-12	ug	21.7	41
Vitamin A, IU	IU	140	159
Vitamin E	mg	1.7	0
Vitamin D2 + D3	ug	1	0
Vitamin D	IU	38	0
Vitamin K	ug	15	0
Fatty acids, total saturated	g	57.15	129
Fatty acids, total trans	g	8.6	0
Cholesterol	mg	680	0

Ground beef nutritional information for 85% lean meat 17% fat, raw: USDA National Nutrient Database for Standard Reference Release 28, Basic Report 23567

Figure 1: Nutritional summary of animal-based ground beef and Impossible Foods ground beef

Life Cycle Inventory and Assessment

This initial assessment was developed by constructing a comprehensive Excel-based series of spreadsheets. Excel was selected due to the early development stage of the product. The Excel-based LCA housed mass balance information for formulation of the hamburger; energy balance information for heating, cooling, and electricity needs for production of the key intermediate as well as the final burger product; calculations with market allocation to appropriately identify relative impacts; and outputs for three environmental impact metrics. The specific recipe is not disclosed here due to intellectual property constraints; instead, comprehensive mass and energy balances from each ingredient group and each process are used to provide outputs of the model. This allows for identification of internal environmental hotspots and variation vis a vis cattle systems.

For the purposes of this initial in-house LCA study, the three most well-documented environmental categories associated with animal farming were considered: greenhouse gas emissions, land use, and water use. Water use is considered as both blue water (groundwater and surface water) and green water (plant-available water from precipitation). These metrics were those most commonly referenced within existing LCA and food systems literature for beef.

Data sources used to calculate our environmental impact across the three categories included primary data from supplier LCAs, publically available databases, co-product market allocations from industry publications, and peer reviewed literature. Total annual environmental impact of the production facility is calculated for all three metrics based on the mass balance of the current production prototype, energy balances, allowances for scrap, and the environmental impact metrics from the various aforementioned sources. These aggregate annual impacts are divided by the total production of that facility to determine each impact for direct comparison to animal-based beef.

Metrics for greenhouse gas emissions of ingredients, packaging, distribution, and energy usage originate from the publically available databases Carbon Calculations over the Life Cycle of Industrial Activities, EPA Power Profiler, and US GREET Database (Azapagic et al., 2010; United States Environmental Protection Agency, 2016; Argonne National Laboratory, 2015) and publications (Stocker et al., 2013; Dudley et al., 2014). Land use metrics originate from a range of reports, publications and websites as there is no available clearinghouse of data. (Ajinomoto Group Sustainability Report, 2013; United States Department of Agriculture, 2015; Bradley and Huang, 2006; Patel et al., 2006). Metrics for blue and green water usage are derived from Water Footprint Network (Water Footprint Network, 2016) and supplementary sources (Ajinomoto Group Sustainability Report, 2013; United States Environmental Protection Agency, 2014).

Animal-based beef comparisons

Several animal production studies are cited as benchmarks for comparison of animal vs plant based ground beef for discussion purposes. Each cited LCA study employs different methods in specific production systems. As such, they are included for context rather than acting as “apples to apples” comparisons. Care was taken to adjust for methodological inconsistencies wherever possible, and functional units were converted to finished product equivalency. For live weight to retail weight and live weight to carcass weight conversions, ratios of 0.43 and 0.70 were used, respectively.

Heme protein
Heme protein as referred to in this paper is a plant-derived leghemoglobin protein that serves as an analog to the myoglobin present in beef. This leghemoglobin protein was initially isolated from the root nodules of leguminous plants. Currently, the heme protein is expressed by yeast produced in culture, and isolated for inclusion in the plant-based ground beef. Environmental impacts associated with the production of the heme protein result from raw materials used to feed the fermentation culture (land) and from processes associated with mass fermentation (energy use, water use).

3. Results

The plant-based ground beef has a decreased adverse impact on the environment in comparison to beef originating from animal farming. The largest efficiency improvement is seen in land use, followed by water and emissions. The production of the Impossible Burger results in the release of 76-90% less kg CO₂eq / kg finished good than ground beef produced from cattle, decreases the impact of necessary land for food production by 91-97% and reduces blue water needed by a range of 86%-91% and green water by 61%-94% depending on the cattle production system.

Emissions

Figure 2 highlights the greenhouse gas emissions impact of the Impossible Burger against a number of cattle-based hamburger counterparts, and Table 1 lists the ingredients that have the highest contribution to greenhouse gas emissions according to both the preliminary internal analysis performed by the authors and the external review executed by Quantis.

The IF internal analysis resulted in an emissions impact per product that is approximately half that found by the review using LCA software. In terms of contributing factors, results are largely congruent (Table 1). The IF internal analysis indicated lower total impacts than were found by the Quantis review, and since both attributional analyses are based on the same comprehensive mass and energy balances around the modeled process, the source for any discrepancy must come from the scores attributed to the ingredients and energy usage. Although this 4.38 kg CO₂eq/kg difference in emissions totals is relatively small when compared to cattle system estimates in Figure 2, with a range

of 12 kg CO₂eq/kg, cattle models depict a diversity of production systems while both IF estimates reflect a uniform inventory. This highlights the significant role that database and model assumptions play in generation of LCA estimates.

Table 1: Contributions to greenhouse gas emissions by ingredient group and process stage

Ingredient Group	% Contribution	kg CO ₂ /kg product
Raw Materials		
Heme Protein Production	4.5%	0.26
Flavor Mix	1.7%	0.10
Potato Protein	6.4%	0.37
Coconut Oil	0.5%	0.03
Textured Wheat Protein	12.9%	0.74
All Other	0.3%	0.02
Inbound Logistics	21.7%	1.24
Production	32.7%	1.87
Food Packaging	3.0%	0.17
Outbound Logistics	16.2%	0.93
Total via IF	100%	5.73
Total via Quantis		10.11

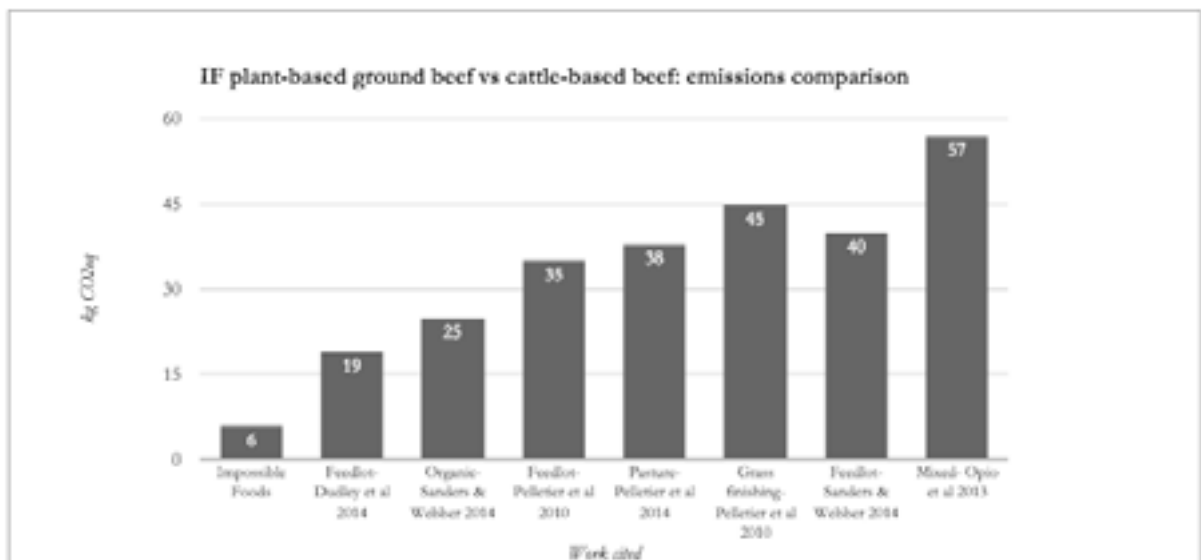


Figure 2: Greenhouse gas emissions of Impossible Foods ground beef vs. animal beef systems from cited studies

Emissions generated by the plant-based ground beef product are lower than beef produced in a cattle-based system by around 75 to 90 percent (Figure 2). Animal systems producing beef diverge widely along geographic and management lines. In general, emissions estimates are highest for cattle production relying on marginal quality forage, for beef production derived from specialized herds without dairy coproducts, and for the mixed systems employed by much of the developing world, in which crop byproducts comprise more than 10% of animal diet (Steinfeld et al, 2006).

Land Use

While the internal IF LCA from the Excel based model found that each kg produced required just over six square meters of arable land, the SimaPro model employed by Quantis estimates that around four square meters are required. The three primary drivers of IF product land use are raw-ingredient based, and with the exception of heme, align with the proportions used in the ground beef bill of materials (BOM). Since texturized wheat protein is the largest ingredient component other than water, it occupies the most arable land required by the IF product, followed by coconut oil. Heme comprises only a small portion of the bill of materials. However as a fermented product, the land footprint of heme reflects that of the agricultural products used to generate the yeast's carbohydrate substrate.

Across all cattle production systems, the IF plant based ground beef relies on dramatically less land to produce a burger product. Figure 3 compares the land use of the Impossible Burger to multiple life cycle studies involving animal-based analogs. Intuitively, grass-fed beef occupies a much higher area of arable land than does beef from an industrial system. Neither system, however, is landless, as even non-pastured beef require large expanses of cropland to supply the feed (usually field corn and soymeal in the US). The land use impact calculated by Quantis is in the same order of magnitude and reinforces the notion that the Impossible Burger uses much less land than its animal-based counterpart.

Table 2: Contributors to land use by ingredient group and process stage

Ingredient Group	% Contribution	m ² /kg product
Raw Materials		
Heme Protein Production	11.9%	0.73
Flavor Mix	1.6%	0.10
Potato Protein	2.0%	0.12
Coconut Oil	31.2%	1.90
Textured Wheat Protein	53.2%	3.25
All Other	0.1%	0.01
Total	100%	6.10
Total via Quantis		3.67

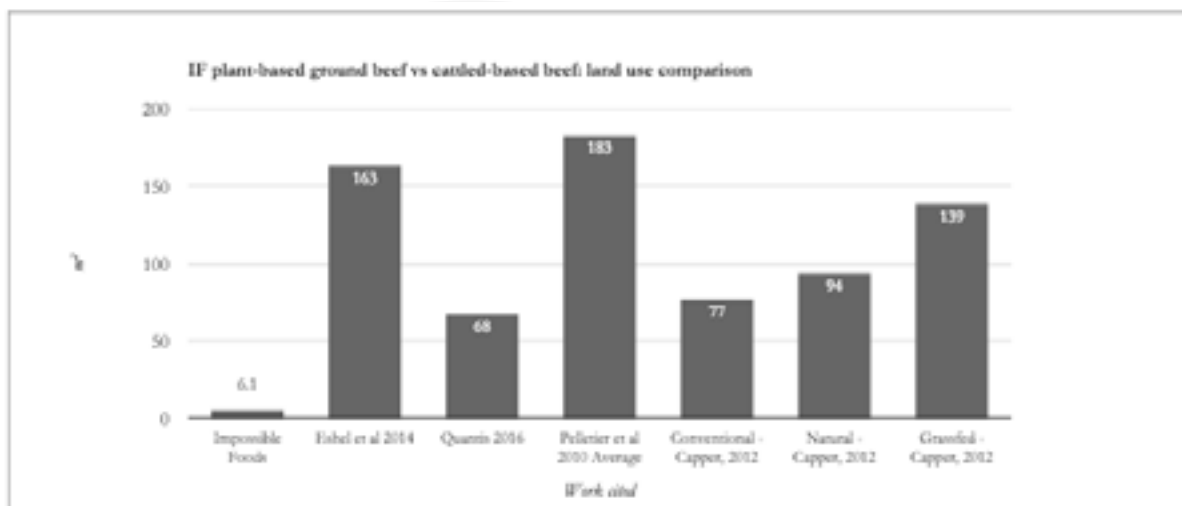


Figure 3: Land use of Impossible Foods ground beef vs. animal based beef from cited studies

Water Use

The widest divergence between results of the internal Excel LCA and the Quantis-reviewed SimaPro LCA are the water use estimates (Table 3). In terms of surface and ground water appropriated throughout the supply chain (blue water), the internal IF model found 70 liters per water required for a finished product, compared to 209 liters of water found by the reviewed model. Modeling assumptions provide the cause of the discrepancy. While the initial review estimated zero irrigation water devoted to coconut oil production, the SimaPro model assumed significant irrigation outlays. Based on regional coconut oil supplier feedback, irrigation infrastructure in the coconut production region is minimal. The SimaPro model's coconut water irrigation parameters was thus adjusted to a conservative 30% of the default, based on total FAO irrigated land as a percent of total agricultural land for the source country. Still, this adjusted parameter for coconut crop irrigation eclipses that of heme, which was the fermentation culture water-use hotspot in the initial model. The rain-fed production assumption for coconut can be seen in the Table 4, which depicts green water consumption. Considering precipitation water rather than irrigation water, coconut oil holds the highest footprint at nearly 90% of total green water.

Figure 4 depicts the relative water use efficiencies of the plant-based ground beef product compared to cattle based beef production systems. Grazing systems rely on pasture irrigation, and while industrial systems are more water efficient, feed crop production still requires irrigation. Industrial beef production generally uses more water, the higher the proportion of concentrates that are used in the cattle feed program (Eshel et al, 2014). As seen in the emissions estimates, the wide range (700 liters/kg) of cattle water use estimates reflect a diversity of production systems. Though the water use estimate range (139.5 liters/kg) for the plant-based ground beef is much smaller, it reflects a single production system represented by different model assumptions, with variation largely driven by a single parameter for irrigation.

Table 3: Contributors to blue and green water usage by ingredient group and process stage

Ingredient Group	Blue Water		Green Water	
	% Contribution	L H ₂ O/kg product	% Contribution	L H ₂ O/kg product
Raw Materials				
Heme Protein Production	87.7%	61.40	3.0%	49
Flavor Mix	1.5%	1.04	0.0%	0
Potato Protein	0.4%	0.26	0.1%	1
Coconut Oil	0.0%	0.00	89.5%	1471
Textured Wheat Protein	8.3%	5.80	7.1%	117
All Other	1.1%	0.76	0.0%	0
Food Packaging				
	1.1%	0.74	0.3%	5
Total	100%	69.99	100%	1644
Total via Quantis		209.4		

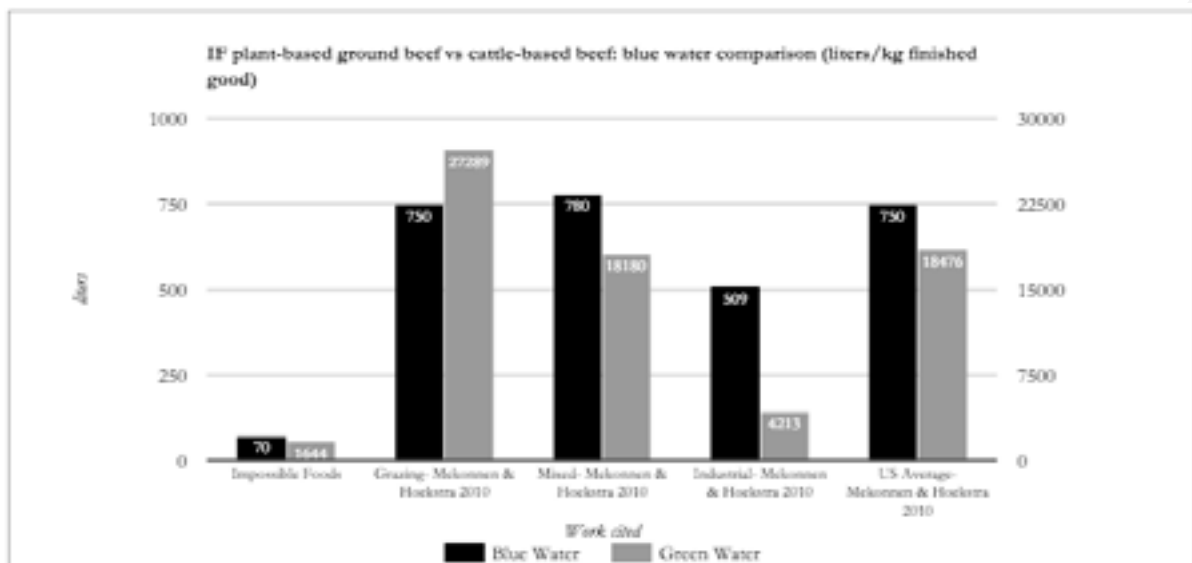


Figure 4: Water use of Impossible Foods ground beef vs animal based beef from cited studies

4. Discussion

Development of a burger from plant-based ingredients is a more efficient use of land, air, and water resources than the cattle-derived alternative. From an engineering perspective, animal conversion of plants into fat and protein adds several unit operations that sap process efficiency when observing the amount of “food” produced at the end of the animal’s life. These inefficiencies are removed when harvesting said plants directly and making a plant-based hamburger product that provides the same nutritional and sensory experience as the animal-based ground beef.

In drawing comparisons to cattle-based systems, it bears noting that beef production systems are widely divergent. While the IF plant-based burger performs well across all estimates of all types of beef production, efficiencies are slightly less dramatic when contrasted with an industrial “landless” system in the United States compared to a mixed system in the developing world, or a pastured system occupying cleared forest land in Brazil. Additionally, different impact metrics are affected by different drivers. While quality of feed is a strong determinant of emissions, water use is impacted by proportion of concentrates in feed, and whether the animal was raised in a stocker system or pastured during part of its life cycle. Finally, any non-specialized beef production system will share environmental impacts across other significant co-products such as dairy. Beef produced in the US will reflect nearly the full environmental impact of that animal, as only about 12% of beef supplied is sourced from the dairy herd (Capper, 2012). Contrast this with a Russian beef production system, in which nearly all beef is sourced from culled dairy animals and breeding overhead of the dairy industry (Alexandratos and Bruinsma, 2012), and specialized beef herd impacts are generally higher.

Regardless of the beef production system, this LCA shows that there is tremendous potential to address ecosystem degradation via popular adoption of novel substitutes to beef products. A large-scale conversion of animal-product consumption into plant-based consumption could reduce environmental stressors while increasing resource efficiency, as demand for animal products grows worldwide. Results would benefit from a stronger accounting of supply chain specificities. Transparency across the entire supply chain would allow for incorporation of known agronomic practices and yields from raw materials, which drive the majority of impacts.

5. Conclusions

The plant-based Impossible Foods ground beef performed better across all metrics than benchmarks for animal-based beef in both the initial Excel-based LCA and in same-inventory results modeled by a third party in LCA software. Plant-based meat products provide a promising mitigation mechanism in addressing disproportionate demand increases for meat products in a growing world population. Future work should expand the suite of impacts investigated in addition to refining those already known, through inclusion of eutrophication, toxicity, and biodiversity and a full understanding of the supply chain. Additionally, given the land use requirements of beef specifically, ecological impact is a challenging but critical metric to understand.

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93. Life Cycle Assessment Perspectives of Insect Proteins for Feed and Food

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ABSTRACT

The emergence of the insect production as a source of alternative proteins indicated the need for the sustainability assessment with a holistic approach and on a fair functional basis. The study aimed at life cycle assessment (LCA) of insect production at industrial level and further insect biomass processing for feed and food purposes.

LCA was performed with the use of ReCiPe methodology, according to the classical scheme of ISO standards. It was based on industrial insect production data, with further scenarios modelling based on data from databases and literature. The LCA results of insect-based products were compared to the appropriate benchmarks (protein feeds, whey protein powder, chicken meat). The comparison was done between similar in protein and moisture content products on a weight (mass) basis.

Results indicated that environmental performance of insect based feed and food could be beneficial, provided that suitable diet for insects was selected. High protein diets were responsible for high environmental impacts and high insect biomass yields. The utilization of manure was environmentally beneficial, but did not result in high insect biomass yields (feed purposes). In result, the scenarios of low value food processing by-products (e.g. distilled grains) and high-impacting waste streams were confirmed to be among the best insect diets for more sustainable for food and feed production accordingly. The production of insect-based protein powder and meat substitutes based on food by-products was 2-5 times more environmentally beneficial than traditional products. As the study was based on the industrial production data for *Hermetia illucens* as a model insect, the results cannot be universally applicable for all the cases of insect biomass application. The need for more extended studies of different insect species use in combination with identified beneficial options is necessary to ensure the practical applicability of the results.

Keywords: alternative proteins, insect-based products, sustainability of insect products, *Hermetia illucens*.

1. Introduction

Food production is the oldest and the most impacting human industry. It is responsible for more than 25 billion tonne CO₂-eq., which brings to the number of more than 50% of overall GHG emissions from all the sources globally (49 billion tonne CO₂-eq.) (Pachauri et al. 2014; Schmidt and Merciai 2014). Agriculture itself is responsible for 70–85 % of water footprint (Shiklomanov 2003; Bellarby et al. 2008; Pfister et al. 2011; Hoekstra and Mekonnen 2012; Vermeulen et al. 2012; Garnett 2014; Pfister and Bayer 2014). Expected increase of food production by 70% till 2050 (FAO 2009) imposes additional risks for the environmental security and human health.

Meat production as valuable protein supply is recognized as the most impacting field in food production (Steinfeld et al. 2006; Schmidt and Merciai 2014). High impacts are associated with high demand for feed resources, cattle enteric fermentation, and inefficient manure management. Multiple publications expressed the need to increase the supply of high-quality protein sources for the growing population, reducing associated environmental impacts (Tilman et al. 2002; Tilman et al. 2011; Cao and Li 2013; Boland et al. 2013; Tilman and Clark 2014). Two main ways in dealing with the problem are foreseen as a part of a solution. The first solution aims to decrease the impacts of feed for food animals. It would decrease the impact of meat production as it accumulates the impact of feed (Herrero and Thornton 2013; Eisler et al. 2014). Another part of the solution could be the improvements in the breeding techniques, which could lead towards the development of environmentally low-impacting livestock (Tuomisto and de Mattos 2011; Herrero and Thornton 2013; Eisler et al. 2014). The path of predicted solution targets the substitution of existing high impacting meats with alternatives, based on plant material (soy, peas, lupine, rice, pulses, etc.), animal produced proteins (milk, insects), and other sources (mycoprotein, algae, lab-grown) (Raats 2007; Blonk et al. 2008; Finnigan et al. 2010). Some of the meat substitutes were developed to the product level and successfully applied to markets (Blonk et al. 2008; Tijhuis et al. 2011).

The main issue of both solutions progression is associated with the determination of their sustainable benefits or drawbacks. It is especially obvious with the use of insects as a source of proteins for feed and food purposes. There is a lack of studies on the sustainability of insects as protein rich food and feed sources, mentioned also in literature (van Huis et al. 2013; Windisch et al. 2013; Mlcek et al. 2014; Lundy and Parrella 2015). Some authors made previous assessments of

insect production at lab scale, analyzed feed conversion and land use change (Ooninx and de Boer 2012; van Huis et al. 2013; Smetana et al. 2015), assessed direct GHG emissions of insects metabolism (Ooninx et al. 2010; Ooninx and de Boer 2012), evaluated waste treatment with insect based system, including plastics utilization (Amatya 2009; Alvarez 2012; Muys et al. 2014; van Zanten et al. 2015; Komakech et al. 2015; Roffeis et al. 2015; Yang et al. 2015) and performed generic assessment of insect production without further processing (Muys et al. 2014). While the studies provided previous estimates on possible sustainable benefits of insects use for feed (Ooninx et al. 2010; van Zanten et al. 2015; Verbeke et al. 2015; de Marco et al. 2015) and food purposes (Ooninx et al. 2010; Ooninx and de Boer 2012; van Broekhoven et al. 2015), no complete Life Cycle Assessment (LCA) of insects harvesting and processing at industrial scale for feed and food purposes was reported up to the date. Moreover, the judgement of environmental aspects of insects production is highly complicated due to the absence of system vision of their growing, processing and use.

This paper is concentrated on life cycle assessment of insect use for feed and food purposes, with basis in industrial scale insect harvesting and processing. It highlights a few main issues associated with low technology readiness levels (TRL) in upstream and downstream processes of insect-based feed and food products. Multiple insect production techniques and processing technologies make the sustainability assessment challenging and often unpredictable. The aim of this article is the systematic LCA of generic insect production (with model insect: *Hermetia illucens*) for feed and food purposes. Even though the application of *Hermetia illucens* is considered in most literature for feed purposes only, we haven't found a sound reason in the literature for not using *Hermetia* larvae or prepupae for food purposes. Nevertheless, this paper is not aimed to promote the use of *Hermetia illucens* for food purposes, but rather uses it as a model insect for the fair comparison of production options. The outcomes of the study should indicate the most promising scenarios for the sustainable insect production for feed and food purposes in environmental perspective.

2. Methods

The goal of the study was to conduct a life cycle assessment of insect production and processing at industrial scale and compare the results with alternative scenarios of insect-based defatted formulation production (powder used as intermediate for feed and food purposes). Additionally, a task of insect-based intermediate comparison with benchmark products was set to identify the potential of insects as a more sustainable source of proteins for food purposes. This was achieved by setting the proper functional units (FU), system boundaries and use of industrially gathered data. The functional units were based on the "cradle to processing gate" perspective with the expansion of the system to include the treatment of generated waste (or production of by-products) at insect harvesting and processing stages.

The LCA was divided into three main stages: raw resource generation needed to feed insects; insects feeding and breeding with further primary preprocessing (harvesting, killing, drying, defatting and milling), and additional processing (used for extended food scenarios). It was based on the standardized LCA approach (ISO 14040 2006; ISO 14044 2006), which allows reviewing insects production and processing as a complete system and as a part of a more complex supply chain. In order to complete the calculations, the research relied on professional SimaPro8 software and use of adapted ecoinvent 3.1 database processes for background data. The study used two midpoint impact assessment methods (ReCiPe V1.08 and IMPACT 2002+) (Goedkoop et al. 2013) to get the possible variability of midpoint characterization factors and their input in the endpoint factors.

Following "prospective attributional approach" (Hospido et al. 2010) the paper aimed to assess and compare innovative insect production and processing technologies, rather than establishing the consequences of their application towards the change of the agricultural and food systems (as it is for the use of consequential approach). Insects processing was considered to the point of intermediate product creation, applicable for different purposes (feed and food). It led to the first FU considered as 1 kg of dried defatted insect powder (with normalized protein content), which was useful to compare various production techniques with benchmark analogues (animal feed, protein additives). Further consideration of insect produced intermediate for food purposes required its combination with other components to get the product comparable to conventional meat. Such considerations triggered the

consideration of the second FU: 1 kg of ready for consumption fresh product at the processing gate (without packaging). The sensitivity analysis included the consideration of feed and food protein content for relative comparison.

The study included two main system boundaries associated with insect production and processing: (1) raw material production for insect diet (inputs), insect breeding harvesting and processing to the intermediate product (“cradle to gate”); (2) further insect biomass processing to the meat-like intermediate suitable for human consumption and comparable with qualities to fresh meat (“cradle to gate”). Only one possible path for insects processing to the intermediate was included in the study (based on available industrial data from *Hermetia Baruth GmbH*). Therefore, it did not intend to analyze the benefits and disadvantages of numerous possible options for the insect biomass production.

3. Results

In order to perform a comparative analysis of modelled scenarios with “traditional” protein feed sources, a number of generic scenarios were used. They represent the most common varieties of animal protein feeds available on the market. Generic protein feed scenario was acquired from ecoinvent 3 database, and represented a combination of side-streams from multiple industries. Fishmeal scenario is a very common option for the animal feeding and included the mix of side-streams and by-products from the fishing industry. It was modelled with the use of Danish LCA database adapted to German conditions and ecoinvent 3 database. Chicken feed scenario was modelled after de Marco et al. (de Marco et al. 2015), and included a production of a feed composed of maize meal (58%), soybean meal (34%), soybean oil (4.5%), dicalcium phosphate (1.2%), calcium carbonate (1.1%) and other additives, minerals and vitamins (1.2%). Not surprisingly the results of their LCA were better than most modelled scenarios, as there was no additional plant raw resources transformation to animal biomass, indicated for the other modelled scenarios. The impacts were within the range of 90 – 218 mPt (mPt – millipoints, single score combined unit of normalized weighted values of impact categories). Only three scenarios of insect protein powder production (fed on cattle manure, on DDGS and on municipal organic waste) had more environmentally sustainable scores: -250 mPt, 130 mPt and -190 mPt accordingly.

As previously mentioned, only a few insect-based intermediates (powder meals) from the analyzed scenarios could be used for the human consumption. The use of manure and waste for food production hold an additional risk for food safety. That’s why scenarios, based on manure and municipal waste, were excluded from further food production modelling. The study of food intermediates production relied on the industrial process of high moisture extrusion, usable to produce “fresh meat texturized substitutes” in order to bring insect-based products in fair comparison with a benchmark (meat). High moisture extrusion data were acquired from industrial scale equipment available in German Institute of Food Technologies (DIL e.V.; Quakenbrueck, Germany). It was modelled that the moisture content of the final product will be around 60%, insect powder would account for 20% and soybean meal powder will be responsible for the other 20%. The use of soybean meal was necessary to provide the final product with fiber-like structure due to the certain viscoelastic properties. As this processing stage was the same for all the intermediates, it was responsible for the similar environmental impacts at the different scenarios (Figure 1).

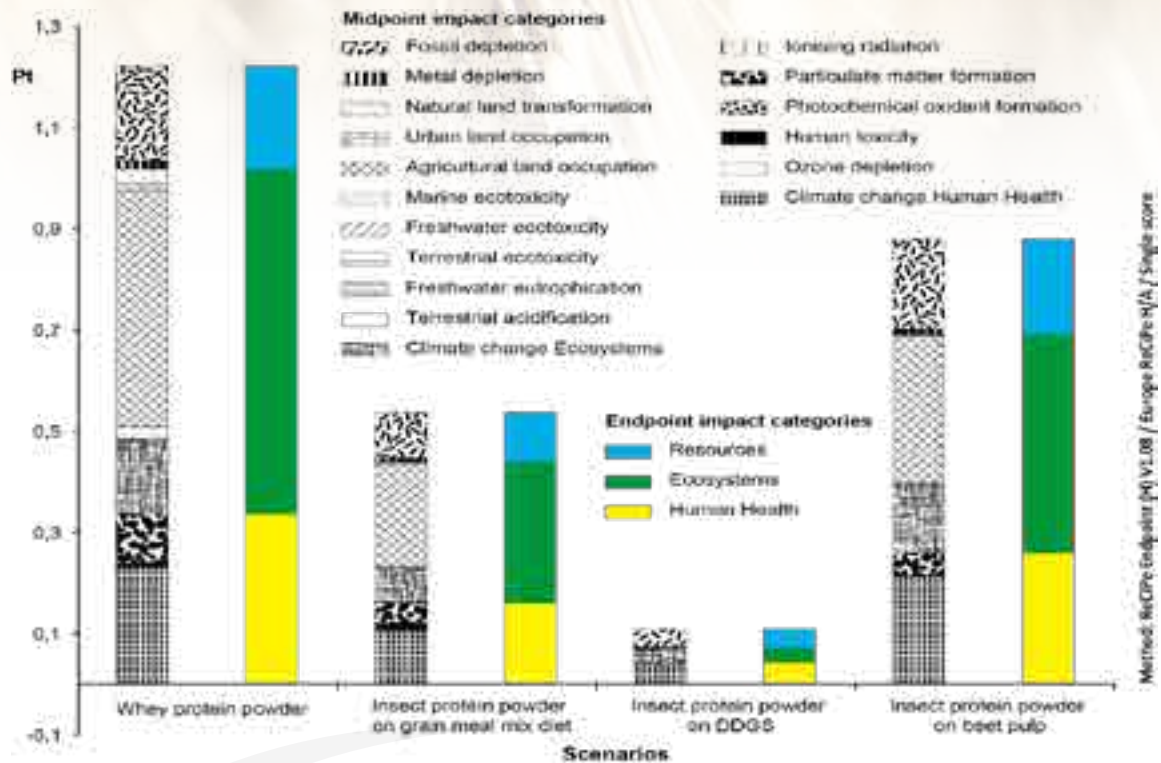


Figure 1: Single-score comparison of powdered intermediates environmental impact with whey protein benchmark (FU: 1 kg of defatted protein powder, from cradle-to-gate)

Whey protein powder production was included in the study with the data modelled from ecoinvent 3 database, as whey protein powder is a co-product of ethanol production (Figure 1). Despite a statement of whey protein powder being of a low economic value, it is available on the market as a comparatively expensive sport additive. The proper allocation of impacts demonstrated whey powder as the least environmentally sustainable option among compared intermediates, intended for human consumption. As a dried product with concentrated protein content (around 60%), whey powder had higher impact than alternative insect-protein based scenarios. The best case scenario of protein powder production was the use of insect protein grown on DDGS. However, even the use of baseline insect meal scenario and meal powder from beet diet insects (which were previously discussed as not the most sustainable options) were more environmentally beneficial than whey powder.

In order to compare insect-based products, suitable for human consumption, with meat, it was necessary to create high moisture texturized products with similar structures and protein content. That's why two scenarios based on the use of soybean meal and insect meal powders were compared to chicken meat (Figure 2). All three products contained around 60% of moisture and 20-30% of proteins. The results indicated that the overall performance of meat substitutes were 4-6 times better than meat (Figure 2). The best performing scenario of moisturized products was the use of soybeanmeal powder for the production of the substitute. In case of insect protein scenario the increased impact was associated with the use of insect powder from the baseline scenario. This scenario was selected as the most assured for the human use.

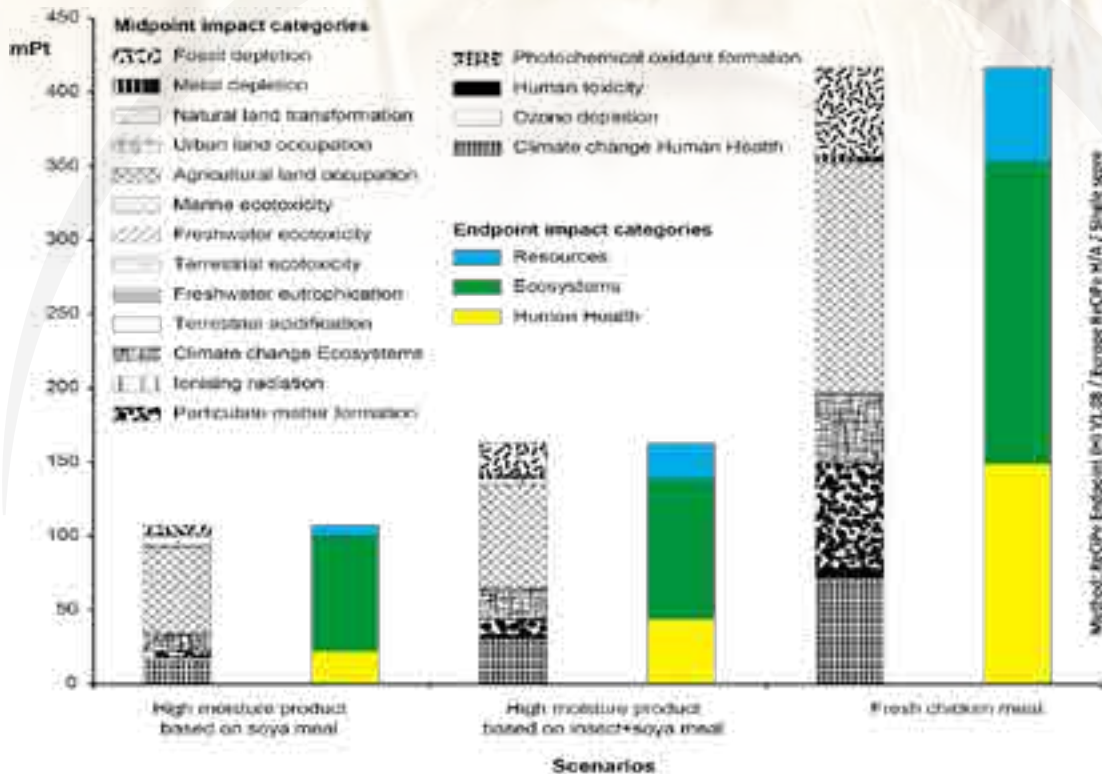


Figure 2: Single-score environmental impact comparison of high moisture intermediates with chicken meat benchmark (FU: 1 kg of high-moisture protein product, from cradle-to-gate)

4. Discussion

The comparison of midpoint impact categories results with calculated values from the available literature sources (in order to perform a fair basis comparison) demonstrated that the range of scenarios analyzed in this article covered the values available in other literature sources (Table 1). Such a diversity of results from the literature sources could be caused by the variations in insect species (house fly larvae, black soldier larvae, mealworm), feeding diet, and production designs. Using only one species as a model we presented variations of diet impacts with insect meal application for feed purposes and further extension for meat substitutes. Even though, the results demonstrated the biggest impact of insect diet on the environmental performance of the final product, we assume that use of different insect species would bring additional variations to the environmental footprint of insect based products.

Table 1: Comparison of midpoint impact categories results with literature sources (FU 1 kg of insect protein meal)

Midpoint impact categories	Sources of LCA studies						
	Housefly larvae (van Zanten et al. 2015)	Housefly larvae (Roffeis et al. 2015)	BSF (Komakech et al. 2015)	BSF (Muys et al. 2014)	Mealworm (Ooninx and de Boer 2012)	Mealworm (Smetana et al. 2015)	BSF This study
GWP, kg CO ₂ eq.	0.77	-	1.24	-	3.5	7.1-7.55	1.36-15.1
Energy use, MJ	9.3	159.85-288.15	1.512	13.4-64.06	44.32	80.0-101.0	21.2-99.6
Land use, m ² arable	0.032	2.79-5.32	-	0.01-0.04	4.68	3.75-3.8	0.032-7.03

Note: BSF – black soldier fly (*Hermetia Illucens*)

Despite a sophisticated spectrum of scenarios, presented in this paper, the study did not account for the complete variations of insect production facilities, insect species, feed and food processing technologies. Moreover, emerging food processing technologies could potentially lower the environmental impact of insect-based products. The study also lacks the reflection on the water footprint of insect production for feed and food, which would be the next step to tackle by the authors.

5. Conclusions

Insect production for feed and food purposes is currently emerging in Western countries. Multiple issues, connected with poor knowledge of the risks associated with potential insect toxicity, allergenicity, nutritional quality, and diseases transferability are triggering the limitation of their use by the regulations. The only force, capable to shift the state of the art is the identification of insects as a more sustainable source of proteins than traditional and other emerging sources. Without this claim interlinked to the profitability of their production, insects will have difficulties on the market of feed and food products.

The study aimed to analyze a few insect production scenarios, indicated insects as not as sustainable source of protein (from environmental perspective) as it is often considered. Insect use for feed purposes could be sustainable, but it depended on the type of insect diet. On one hand, a higher insect yield (and protein content) was achieved via the use of feed with comparatively good nutritional quality (e.g. rye meal, soybean meal), but then the final product was associated with high environmental impacts. On the other hand, low quality feeds, based on manure, had low efficiency for insect biomass and protein yields. It caused an increase in the use of resources at the insect growing stage, which overcame the benefits from manure utilization. Therefore, the most promising for feed production are those scenarios, which were based on low value agri-food products with good nutritional profile (e.g. DDGS) or based on waste utilization with high environmental impact, so the impact of increased use of resources should not overcome the benefits from the manure utilization. The application of insects for feed purposes is controversial and depends on a few factors, which calls for the need to proof the environmental performance of insect diets, growing and processing technologies, so the ranges of more sustainable options of insect production can be identified.

The comparison of food products was based on the weight of the product (1 kg) with a similar nutritional profile (water, protein, fat content). Insect-based protein products demonstrated surprisingly better performance than traditional food analogues. The environmental impact of insect protein powder (high quality diet) was at least twice better than whey protein powder used as sport dietary supplement. Insect biomass grown and harvested on low quality diets (manure, municipal organic wastes) was not considered for food purposes. Similarly, insect-based meat substitutes were two times more environmentally friendly than chicken meat. At the same time, the comparison of the insect-based meat substitute with a soymeal-based meat substitute revealed environmental benefits of the last one.

In general, the application of insects for food purposes is environmentally more beneficial than traditional sources of proteins, but detailed study of insect based food comparison with the other emerging food products is needed. In order to align with the new EU Novel Foods Regulation (Regulation (EU) No. 2015/2283) the insect-based products should be analyzed in comparison with benchmark via nutritional studies, which would be of high importance for upcoming life cycle assessment studies.

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ORAL PAPERS IMPACTS

17. Impacts

53. Relevance of the Spatialized Territorial LCA method to assess environmental impacts: case study of eutrophication in a French catchment

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ABSTRACT

To help local stakeholders managing agricultural activities, environmental impacts should be assessed at the territorial level by considering its biophysical characteristics. The Spatialized Territorial Life Cycle Assessment (STLCA) method (Nitschelm et al. 2015), goal and scope are to integrate: (1) at the Life Cycle Inventory (LCI) stage, the locations of direct emissions as well as biophysical characteristics of the surroundings (e.g., soil type) into calculations of direct emissions, and (2) at the Life Cycle Impact Assessment (LCIA) stage, fate of pollutants as well as the territory's sensitivity (i.e. the biophysical context) to the impact(s). The objective of our study was to prove the usefulness of STLCA by applying it to the Lieue de Grève catchment, France. We present the results of LCA and STLCA of this territory. We focus on eutrophication since it is a local impact on this territory. Non-spatialized emissions for crops and livestock were determined using emission models of the French agricultural database Agri-BALYSE (Koch and Salou 2015). Spatialized nitrogen emissions (i.e. NO₃⁻, NH₃ and N₂O) are predicted using Syst^N, a simulation model that includes agricultural practices and biophysical parameters (Parnaudeau et al. 2012). Non-spatialized eutrophication impacts are determined using the CML-IA characterization factors (Udo de Haes et al. 2002). Two spatialized characterization factors are derived from CML-IA for marine and freshwater eutrophication (Nitschelm et al., in prep.). Spatialized characterization factors include fate factors as well as a sensitivity factor specific to the territory. Three main results arose from this study. First, with spatialized results, a map can be generated showing the locations of emission and/or impact hotspots within the territory. Second, at the territorial level, there was no difference in magnitude between non-spatialized impacts and impacts spatialized at the LCI stage. Including fate and sensitivity of surroundings in the characterization factor decreased direct eutrophication impacts (i.e. within the territory) by 10 kg PO₄³⁻ eq. per ha. The decrease in eutrophication from non-spatialized to spatialized LCIA is significant, but this is mainly due to the integration of fate and sensibility in the characterization factor. Third, at the farm level, eutrophication impacts per ha differ depending on whether they were non-spatialized, spatialized at the LCI stage, or spatialized at the LCI and LCIA stages. These differences are explained by differences in farm practices and biophysical characteristics of the territory. From these results, we can conclude that spatialization is not always necessary and depends on the goal and scope of the study. For a study at the territorial level, non-spatialized methods can be sufficient. For a study considering farms and their practices, spatialized methods can better represent farm diversity, as well as hotspot location, within a territory.

Keywords: life cycle assessment, regional, impact assessment, agriculture

1. Introduction

Loiseau et al. (2012) identified life cycle assessment (LCA) as a suitable method to perform environmental analysis of an agricultural territory due to its life-cycle and multicriteria perspectives. Recently, Loiseau et al. (2013) developed a methodological framework for territorial LCA. This methodological framework is, however, non-spatialized, and considers the territory as a black box. When assessing environmental impacts of an agricultural territory, integrating spatial differentiation is important because: (1) changes in agricultural land-use influence environmental impacts of agriculture (Cederberg et al. 2013) and (2) several emissions (e.g., nitrogen, phosphorus) and impacts (e.g., eutrophication, acidification) due to agricultural activities vary depending on the biophysical characteristics of the surroundings (e.g., weather, soil type) as well as farming practices (e.g., fertilization). In the literature, regionalized LCA has been performed for agricultural regions (e.g., Anton et al. 2014; de Vries et al. 2015) but without spatial differentiation both at the Life Cycle Inventory (LCI) and Life Cycle Impact Assessment (LCIA) stages. Extensive work has been done, however, to integrate spatial differentiation both in LI and LCIA (e.g., Mutel and Hellweg 2009; Wegener Sleeswijk and Heijungs 2010; Yang 2016). At the LCI stage, several methods have been developed to assess spatially explicit emissions from agriculture (e.g., Bessou et al. 2013; Scherer and Pfister 2015). At the LCIA stage, a massive amount of work has been done in spatialization (O'Keefe et al. 2016). For example, several methods for spatialized characterization of acidification (e.g., Civit

et al. 2014; Roy et al. 2014) and eutrophication (e.g., Azevedo et al. 2013; Struijs et al. 2011) have been developed at different spatial levels (e.g., country, catchment) since the 1990s.

To encompass the “spatialized territorial” approach, Nitschelm et al. (2015) developed the Spatialized Territorial Life Cycle Assessment (STLCA) method, which integrates spatial information of the surroundings in a territorial LCA. In this paper, we apply the STLCA method at the LCI and LCIA stages. At the LCI stage, the locations of direct emissions as well as biophysical characteristics of the surroundings (e.g., soil type) are included in calculations of direct emissions, using a simulation model. Syst’N model (Parnaudeau et al. 2012) was used to calculate nitrogen emissions (NO_3^- , NH_3 and N_2O) at the plot level. At the LCIA stage, the fate of pollutants and the territory’s sensitivity (i.e. the biophysical context) to the impact(s) are integrated in characterization factors. In this paper, we focus on nitrogen and phosphorus emissions because they are the major emissions in the territory under study (Gascuel-Oudoux et al. 2015). In consequence, we focus on eutrophication since it is the major impact arising from nitrate and phosphorous emissions. Moreover, these emissions and impacts are localized, and can benefit from spatial differentiation. The main objective of this paper is to obtain spatialized eutrophication calculation using the STLCA method to verify its relevance and applicability through its application to a case study, the Lieue de Grève catchment located in Brittany, France.

2. Methods

2.1. Goal and scope

The goal of this LCA is to compare eutrophication potentials due to agricultural activities in the Lieue de Grève catchment: (1) using a global LCA approach, (2) using a spatialized LCI and (3) using a spatialized LCI and LCIA. We defined the territory’s boundaries as the surface area, whether inside or outside the catchment, of all farms with at least 30% of their surface area located inside the catchment. Although this resulted in a territory with non-contiguous boundaries, it was important to retain information for entire farms, since many choices that influence impacts are made at the farm level.

2.2. LCI

2.2.1. Farm clusters conception and description

There are 112 farms located within the territory’s boundaries defined in 2.1. Based on those 112 farms, we built 13 farm clusters representative of the 112 farms. The 13 clusters are defined from statistical analysis (principal component analysis and hierarchical clustering on principal components) of a database of 112 farms in the catchment. These 13 farm clusters differ in agricultural production and practices (Table 1).

Table 1. Cluster types representing the 112 farms of the Lieue de Grève catchment. Bovine meat: meat from beef cows; Dairy milk: milk from dairy cows; grass: feed based on grass; maize: feed based on maize; maize and grass: feed based on a mixture of maize and grass.

Farm cluster	Production	No. of farms	Cluster area (ha)
Bovine meat / grass	Bovine meat	3	218
Bovine meat and cash crops	Bovine meat	5	342
Bovine meat / maize and grass	Bovine meat	10	570
Dairy milk / grass	Bovine milk	6	321
Dairy milk / maize	Dairy milk	15	983
Dairy milk and cash crops	Dairy milk	12	970
Dairy milk / maize and grass	Dairy milk	24	1530
Dairy milk and meat / grass	Dairy milk and meat	3	299
Dairy milk and meat / maize and grass	Dairy milk and meat	5	351
Dairy milk and meat / maize	Dairy milk and meat	13	1070
Dairy milk and meat and cash crops	Dairy milk and meat	8	586
Pork and cash crops	Pork	6	210

Each cluster was represented by a real farm representative of the cluster (“typical farm”). For each typical farm, we had access to an extended set of farm-survey data that included management practices for crops (e.g., yields, sowing and harvesting dates, amounts of fertilisers and pesticides used) and livestock (e.g., production levels, number of animals, types and quantities of feed, grazing dates) as well as background data from Ecoinvent v2. LCI of each cluster was performed using the data obtained for these typical farms. Typical farm within a farm cluster were chosen based on data availability.

2.2.2. Emission calculations

Non-spatialized nitrogen emissions are determined using emission models of the French agricultural database Agri-BALYSE (Koch and Salou 2015), based on the non-dynamic COMIFER 2002 method (Cattin et al. 2002). This method estimates nitrate emission levels based on four risk levels: (1) bare soil duration, (2) biomass quantity of residues from previous crop, (3) nitrogen quantity in residues from previous crop and (4) nitrogen uptake from next crop.

Spatialized nitrogen emissions (i.e. nitrate, ammonia and nitrous oxide) from crop fields are predicted at the field level using the dynamic model Syst’N (Parnaudeau et al. 2012), which simulates nitrogen emissions at the scale of crop rotations. Syst’N is chosen because it can simulate nitrogen emissions for various crop systems (crop management, type and rotation) along with various soil types (texture, depth, organic matter content). Nitrate leaching for each crop within a rotation is set equal to the quantity of nitrate leached from drainage beginning in the Lieue de Grève catchment (November) of the year the crop was harvested (Liao et al. 2015). While climate is considered to be the same throughout the catchment, soil type (e.g., depth, structure, organic matter content) varied depending on the field location. Based on the database “Sols de Bretagne” (Lemerancier 2010), we identified 8 soil types on the Lieue de Grève catchment.

The SALCA-P model (Prasuhn 2006) is used to predict non-spatialized and spatialized total phosphorous emissions (eroded, particulate and dissolved) at the plot level. LCI is determined at the farm level.

2.3. LCIA

Non-spatialized eutrophication impacts are determined using the CML-IA method (Guinée et al. 2002). CML-IA does not distinguish marine eutrophication from freshwater eutrophication.

Spatialized eutrophication impacts are estimated using two spatialized characterization factors derived from CML-IA for marine and freshwater eutrophication (Nitschelm et al., in prep.). These factors include, in addition to eutrophication potentials specific to nitrogen and phosphorous compounds, fate factors and a sensitivity factor specific to the territory. Eutrophication potential (EP) represents the potential of each compound to participate in the eutrophication impact and is given in kg PO₄³⁻ eq (EP_{NO₃-} = 0.10 and EP_{PO₄3-} = 3.06). Fate factors for nitrate and phosphorus are determined using the Nutting-N and Nutting-P models (Dupas et al. 2015). As a result, the nitrate fate factor equalled 0.6, meaning that 60% of the nitrate emitted from Lieue de Grève farms reaches the catchment outlet; the phosphorus fate factor equalled 0.22, meaning that 22% of the phosphorus emitted from the Lieue de Grève farms reaches the river network. Sensitivity factors are derived using (1) the Håkanson (2008) method for marine eutrophication and (2) the French SYRAH method (Fabre and Pelte 2013) for freshwater eutrophication. The sensitivity factor for the Lieue de Grève river network is determined using the BD TOPO© database (IGN France, Paris) at a resolution of 1:10,000. Based on the fate and sensibility factors described previously, we calculate for the Lieue de Grève river network a sensitivity factor of 0, indicating that it is not sensitive to freshwater eutrophication. For the Lieue de Grève bay, the width of the bay mouth and depth were determined from a digital elevation model at a resolution of 100×100 m. We calculate a sensitivity factor of 0.76, indicating that the bay is highly sensitive to eutrophication. Multiplying each set of factors together, the Lieue de Grève’s characterization factor for freshwater eutrophication is 0, while its characterization factor for marine eutrophication is 0.046. Since freshwater eutrophication equalled 0, this paper shows results only for marine eutrophication. Regional impact of the Lieue de Grève territory is estimated by

multiplying mean per-ha impacts of each typical farm by the surface area of the corresponding farm cluster.

3. Results

3.1. Mapping marine eutrophication sources

A map of sources of marine eutrophication, showing hotspots locations within the territory (Fig. 1), identifies farms with the highest marine eutrophication potential per ha, located mainly at the western border of the territory, and those with the lowest, located in the north and the centre of the territory.

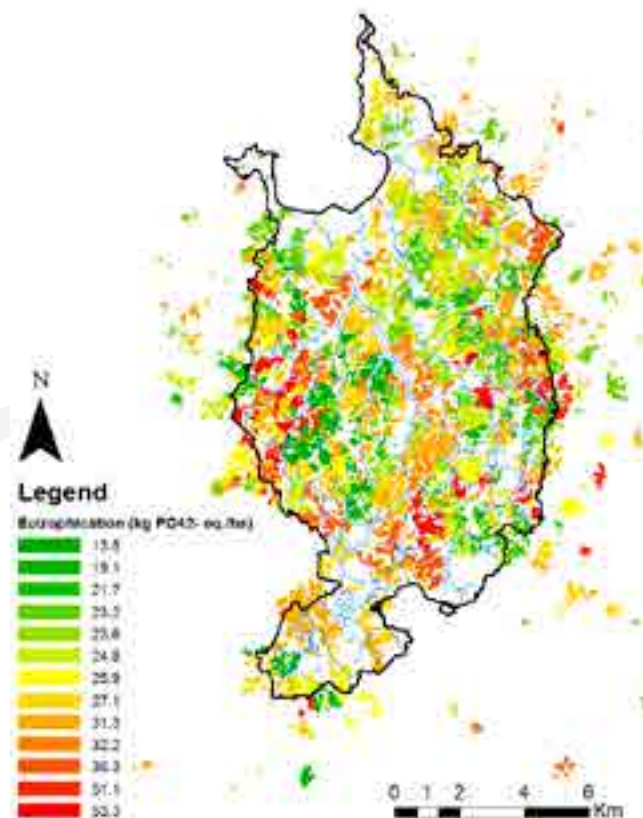


Figure 1. Map of sources of spatialized marine eutrophication impact for the 13 farm clusters of the Lieve de Grève catchment, Brittany, France. These results were obtained by adding spatial differentiation in LCI and LCIA stages.

3.2. Non-spatialized vs. spatialized eutrophication results

At the farm level, eutrophication impacts per ha differ depending on whether they are non-spatialized, spatialized at the LCI stage, or spatialized at the LCI and LCIA stages (Fig. 2). These differences are explained by differences in farm practices and biophysical characteristics of the surroundings such as soil type as well as interaction between these two parameters.

At the territorial level, there is no significant difference (p -value > 0.05), in magnitude between non-spatialized impacts and impacts spatialized at the LCI stage. This result can be explained by a compensatory phenomenon at the farm level: while eutrophication impact decreases in certain farms (e.g., “milk + meat grass” farm, Fig. 2), the same impact increases in other farms (e.g., “milk + meat + pork + CC” farm, Fig. 2). As the results, the mean eutrophication level is the same for non-spatialized and spatialized-LCI results. When spatialized at both LCI and LCIA stages, direct eutrophication impacts (i.e. within the territory) decrease by 10 kg PO_4^{3-} eq. per ha (Fig. 3). The change in eutrophication from non-spatialized to spatialized LCI and LCIA is significant (p -value < 0.05). In the Lieve de Grève’s case, changes between non-spatialized and spatialized LCI and LCIA are therefore driven by the change in the characterisation factor, and not by the change in spatialized emission. At the territorial level, there is therefore no interest to spatialize LCI. Spatialized LCIA can

however benefit from spatial differentiation at this level since it allows a better differentiation between marine and freshwater eutrophication.

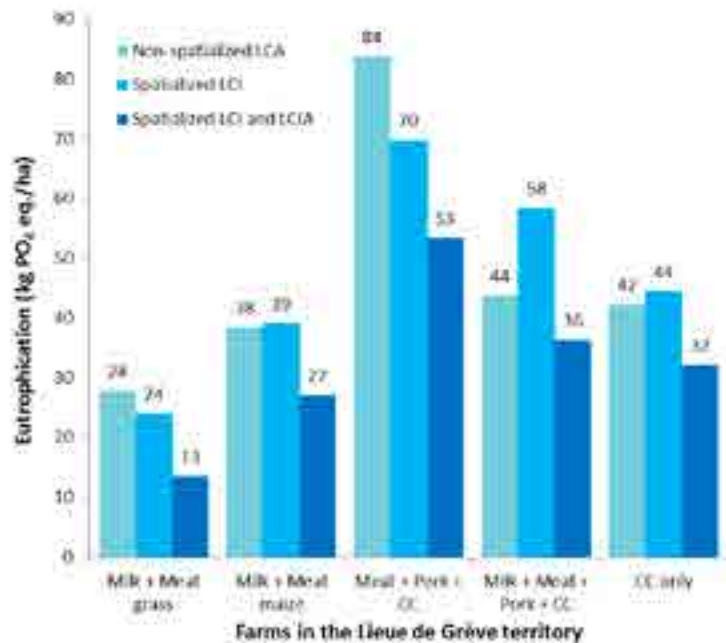


Figure 2. Marine eutrophication impacts (kg PO₄³⁻ eq.) per hectare of farm cluster. Milk: milk from dairy cows; Meat: from dairy or beef cows; CC: cash crops; grass: livestock feeding based on grass; maize: livestock feeding based on maize.

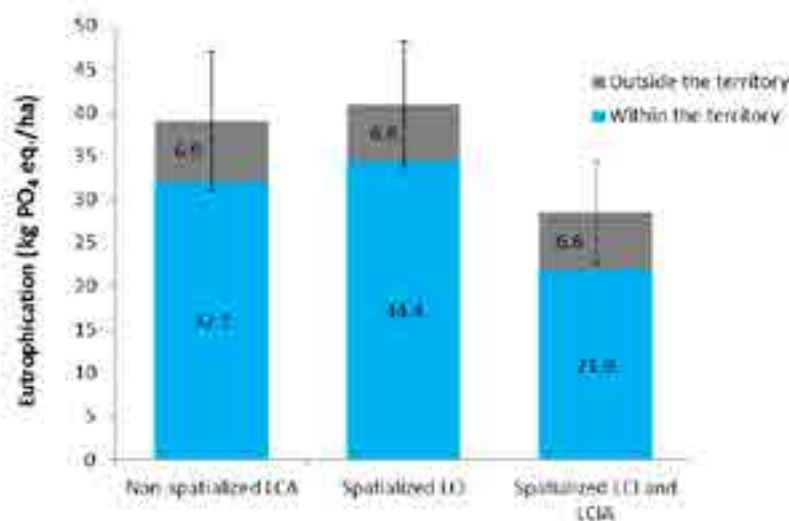


Figure 3. Mean marine eutrophication impacts (kg PO₄³⁻ eq.) per hectare of the Lieu de Grève territory. Non-spatialized LCA: Agri-BALYSE for emissions and CML-IA for eutrophication; Spatialized LCI: Syst'N for nitrogen emissions, Agri-BALYSE for other emissions and CML-IA for eutrophication; Spatialized LCI and LCIA: Syst'N for nitrogen emissions, Agri-BALYSE for other emissions and spatialized characterization factors for marine eutrophication in the Lieu de Grève catchment. Error bars represent range between maximum and minimum impacts of farm types in the territory.

3.3. Variability in impacts

For farms in the four “dairy milk” clusters (Table 1), we perform spatialized LCA that considers differences in farm size, and therefore milk production, and the soil types present in the farms, since they are a major driver of nitrogen emissions. We assume that milk production is linearly correlated with fodder production of farms within the same cluster. Results indicate that the four dairy clusters have the same range of potential median marine eutrophication impact: ca. 21-24 kg PO₄³⁻ eq/ha. Results extent varies depending on the farm clusters. The “milk / grass and maize” farm cluster have the highest extent ranging from 15 to 30 kg PO₄³⁻ eq/ha while the “milk and cash crops” farms have a lower extent ranging from 20 to 27 kg PO₄³⁻ eq/ha. Indeed, the “milk / grass and maize” farm cluster area is almost twice the size of the “milk and cash crops” farm cluster and covers more soil type. Hence, location is an important factor to take into account the impact variability within a cluster.

4. Discussion

Our method combines a simulation model (Syst’N) to predict spatialized nitrogen emissions at the field level with spatialized characterization factors for marine and freshwater eutrophication. From this study’s results, we can conclude that spatial differentiation is not always necessary, since non-spatialized and spatialized LCI had the same direct eutrophication impact results at the territorial level. Comparing non-spatialized and spatialized LCI results, “compensation phenomena” appear to exist: some farms have higher impacts with spatialized LCI, while others have lower impacts, which lead to the same results at the catchment level. Non-spatialized LCIs (i.e., national averages) are sufficient at the catchment level. However, the change in eutrophication from non-spatialized to spatialized LCI and LCIA is significant. This decrease in eutrophication is mainly due to the change in the characterization factor from global to local. In the spatialized characterization factor, nitrogen and phosphorous fate as well as the territory sensibility are taken into account.

If the goal and scope consider farms and their practices, spatialized methods can better represent farm diversity and variability. In this study, dairy farms had a wide range of variability in marine eutrophication impact, which depends on milk production levels and soil types. We can therefore conclude that each cluster has a combination of practices + biophysical characteristics that lead to low eutrophication impacts.

These results have practical consequences: (1) displaying results in maps can greatly aid decision-making processes; (2) variability in practices and biophysical characteristics can help identify combinations of them that lead to lower impacts, as well as hotspot location within the territory; (3) this variability can help in building territorial land-planning scenarios (e.g., Acosta-Alba et al. 2012) and (4) integrating spatial variability helps explain uncertainties (Chen 2014).

There are, however, practical limitations to implement such methods. According to O’Keeffe et al. (2016), these limitations include (1) the possible lack of data at a territorial level; (2) lack of appropriate spatially dependent models to estimate direct emissions and impacts; (3) the impossibility to spatialize every stage of an LCA (in terms of knowledge, data availability and computational technique) and (4) the validity and representativeness of hybrid life cycle approaches such as use of both spatialized and non-spatialized LCA (van Zelm and Huijbregts 2013). Moreover, spatialized assessment is time consuming, and is, for now, difficult to implement in LCA software. As Jeswani et al. (2010) pointed out, adding too much complexity to LCA can decrease decision makers’ confidence in its results. There is a need to refine and standardize spatialized LCI and LCIA approaches. For the former, Yang (2016) propose solutions to implement more accurate spatialized LCI using regional output percentage or multi-regional Input-Output models. For the latter, the characterization method Impact World+ (Bulle et al. 2014), currently under development, aims to develop a wide range of spatially explicit characterization factors at the lowest spatial level for several impact categories (e.g., acidification, eutrophication, (eco)toxicity, water use, land use). In the meantime, we suggest spatializing well-known processes of foreground processes only if the goal and scope of the study demand it.

5. Conclusions

Several results arise from this study. First, a map that shows the locations of impact hotspots within the territory is generated. This first result can help decision makers visualize sources of

impacts. Indeed, mapping results is a powerful way to communicate information, especially when they include spatial characteristics. Second, we demonstrated that using a spatialized characterization factor adjust direct eutrophication impacts (i.e. within the territory) depending on the territory's sensitivity to this impact. Third, at the cluster level, eutrophication impacts per ha differed depending on whether they were non-spatialized, spatialized at the LCI stage, or spatialized at the LCI and LCIA stages. And finally, for each cluster, one can identify combinations of practices/biophysical characteristics of the surroundings that have lower eutrophication potential impacts. From these results, we conclude that spatialization is not always necessary and depends on the goal and scope of the study. For a study at the territorial level, non-spatialized methods can be sufficient. For a study considering farms and their practices, spatialized methods can better represent farm diversity, as well as hotspot location, within a territory. Since spatialized analyses take a large amount of time, we recommend spatializing inventory and impact assessment only if the goal and scope of the study requires it.

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36. Application of Eutrophication Potential Indicators in a Case Study Catchment in New Zealand

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ABSTRACT

Objective: The objective of this work was to determine the effects and implications of generic and site-specific Eutrophication Potential indicators in the LCA of livestock farm systems using a New Zealand (NZ) lake catchment case study. **Method:** Average dairy and sheep & beef farm systems (based on primary data) in the Lake Taupo catchment have been studied. Emissions of nitrogen (N) and phosphorus (P) to waterways from these farms have been calculated using the site-specific OVERSEER nutrient budgets model hereafter called OVERSEER. Water quality data has also been collected for the lake catchment and has defined the extent of N and/or P limitation for algal growth. In the Lake Taupo catchment, maximum catchment loads of N have been defined and used to set regulated maximum N leaching losses from farms, in order to maintain acceptable levels of water quality for the community. A range of different Eutrophication Potential indicators have been assessed and compared. These have then been related to the actual nutrient limitations of the lake to understand the effects of choice of indicator and its relevance to specific catchments. **Results:** For sheep & beef and dairy farm systems, N leaching was 13 and 49 kg N/ha/year, and P runoff losses were 1.1 and 3.0 kg P/ha/year, respectively. Ammonia emissions were also calculated. These emissions data were then used to calculate the increase in nutrients in water bodies and Eutrophication Potential using methods that vary in the inclusion of N and/or P, including ILCD (Freshwater Eutrophication Potential, Marine Eutrophication Potential calculated with ReCiPe 2008), Freshwater Eutrophication Potential calculated with ReCiPe 2015, and Eutrophication Potential calculated with CML. **Conclusions:** Freshwater Eutrophication Potential indicators that focused only on one nutrient can be inappropriate, as illustrated in NZ, where many freshwater bodies are co-limited by N and P (in terms of algal growth). Generic indicators based on P only may sometimes be irrelevant at a site-specific level. New Zealand's largest lake is a good illustration: Lake Taupo is co-limited, and water quality concerns and regulations are not focused on P, but solely on N due to increasing N levels over time. Ideally, site-specific eutrophication indicators should be used at a catchment level to account for water body specificity in LCA.

Keywords: Life Cycle Assessment, Water, Nutrients, Nitrogen, Phosphorus

1. Introduction

Aquatic eutrophication is one of the major water quality issues throughout the world (Khan and Mohammad 2014). Eutrophication covers all impacts of excessively high environmental levels of macronutrients, the most important of which are nitrogen (N) and phosphorus (P) (Guinee et al. 2002). An excess of these nutrients can lead to uncontrolled phytoplankton (algae) growth. In NZ, the National Policy Statement for Freshwater Management sets out the objectives and policies for freshwater management, aiming to protect quality of waterways (MfE, 2014). Minimum acceptable values for water trophic state were defined, and have to be reached within a reasonable timeframe. In this context, assessing the contribution of NZ livestock farms to aquatic eutrophication is highly relevant, especially since they are the main anthropogenic source of nutrients in NZ water bodies (Scarsbrook and Melland 2015).

Owing to the significant concern about the eutrophication impacts of agriculture, Life Cycle Assessment (LCA) studies on agricultural systems often characterise this impact. Various methods exist for this impact category, but they differ in terms of inventory requirements, geographical coverage, spatial resolution, and emission pathways modelled. Eutrophication impacts calculated with CML (Heijungs et al. 1992) assess both terrestrial and aquatic eutrophication in a single indicator, where all emissions (N and P to air, water, soil, and organic matter to water) are aggregated using the Redfield ratio which provides "equivalency factors" (Redfield et al. 1963). Thus, the characterisation factor is independent of whatever substances happen to be the limiting factor of algae growth in a given location (Guinee 2002). This method assumes that 100% of the emissions to water will contribute to eutrophication, meaning that the fate (transport and transformation/attenuation) of the nutrient is not modelled. As a result, CML corresponds to a "worst case scenario" since it ignores that only a fraction of the emissions will be transported to the aquatic environment (Struijs et al. 2009).

Eutrophication impacts calculated with ReCiPe 2008 (Struijs et al. 2009) assess aquatic eutrophication through two distinct impact indicators: marine eutrophication and freshwater eutrophication. This method was recommended by the European Commission (JRC-IES 2011), notably because it accounts for the sensitivity of the receiving water body: marine water is considered to be sensitive to N (i.e.: N is the limiting nutrient for marine biomass growth), whereas freshwater is considered to be sensitive to P (i.e.: P is the limiting nutrient for freshwater biomass growth) (Struijs et al. 2009). In ReCiPe, the Fate Factors (FF) for N and P to marine and freshwaters are site generic, but are derived from a European model (CARMEN & EUTREND) which make them specific to Europe. Eutrophication impacts calculated with more recent methods model the fate of N and P with two distinct spatially explicit models. For assessing freshwater eutrophication, the fate modelling of P was improved from a European model to a global model in ReCiPe 2015 (Helmes et al. 2012), which also assesses the persistence of P in the freshwaters. For assessing marine eutrophication, the generic fate of N is calculated for large marine ecosystems (Cosme et al. submitted).

Assuming that marine and freshwater ecosystems have distinct single limiting nutrients, being N and P, respectively, may be a methodology weakness. Indeed, Elser et al. (2007) showed that freshwater and marine ecosystems are similar in terms of N and P limitation. In NZ, freshwaters can be N-limited, P-limited or co-limited (McDowell and Larned 2008). Lake Taupo for example, NZ's largest lake, is co-limited by N and P (Pearson et al. 2016) and local government regulations are currently focused on limiting N inputs due to increasing N levels in the lake over time (WRC 2016).

The objective of this work was to evaluate different Life Cycle Impact Assessment (LCIA) methods and their relevance to estimating eutrophication impacts of freshwater using NZ's largest lake as a case study. Using case study farms, we calculated eutrophication indicators with different methods, and determined the implications of using generic and site-specific Eutrophication Potential indicators in the LCA of livestock farm systems.

2. Methods

Emissions and impacts were calculated for the farm stage only, on a per-hectare basis.

2.1. Case study farms

The livestock farm systems studied were an average dairy farm and an average sheep & beef farm from the Lake Taupo catchment (volcanic soil with rainfall of 1300 mm/year) (Thorrold and Betteridge 2006). All farms have livestock grazing perennial grass/clover pastures all year round. The 100 ha dairy farm has 270 cows, uses no brought-in feed and applies fertilisers at 100 kg N/ha/year and 46 kg P/ha/year. The 480 ha sheep & beef farm is stocked at 11.5 sheep-equivalents/ha, with a 70:30 sheep:cattle ratio and applies fertilisers at 17 kg N/ha/year and 22 kg P/ha/year.

2.2. Inventory of nutrient flows

Based on primary data for inputs on farm, field emissions of N leaching and P runoff were calculated with the OVERSEER[®] nutrient budget model (Wheeler et al. 2007), and for ammonia and nitrous oxide, NZ-specific emissions factors from the NZ Greenhouse Gas Inventory (MfE 2015) were used (for details, refer to Table 1). OVERSEER is a nutrient model which has been validated against field site measurements from throughout NZ (McDowell et al. 2005, Wheeler et al. 2007). An attenuation factor of 50% was used for nitrate from soil (below root-zone) to freshwaters (rivers and lakes) (Elliot et al. 2014).

2.3. Eutrophication impact assessment

Eutrophication impacts were calculated with CML, ReCiPe 2008, and ReCiPe 2015 (Huijbregts et al. 2015). We distinguished three stages in the impact assessment models; (i) the increase in nutrients in the receiving water body (the emission multiplied by a Fate Factor (FF)), (ii) divided by a reference

emission in order to obtain a eutrophication potential indicator (midpoint), or (iii) multiplied by an Effect Factor (EF) as an indicator for ecosystem damage (endpoint).

Nutrient fate modelling – To calculate an increase in nutrients in a water body, each method relies on different inventory requirements. CML and ReCiPe 2015 rely on net emissions of nutrients to freshwater, whereas ReCiPe 2008 is based on gross supply of fertilisers and manure to agricultural soil. In other words, the FF of ReCiPe 2015 accounts for the fate of nutrients from freshwater to final receiving water compartment (freshwater or sea), whereas the FF_{gross supply} of ReCiPe 2008 accounts for the fate from agricultural topsoil to final receiving water compartment (Fig. 1).

CML does not provide FF, but assumes 100% of emissions will contribute to the eutrophication potential (pathway A in Fig. 1). The use of ReCiPe 2008 FF requires caution: to account for NZ-specific volatilisation rates, we did not apply the composite FF for N to soil+air, but applied two separate FF to soil and to air (Goedkoop et al. 2009). In this study, we compared ReCiPe 2008 using impacts assessed based on gross supply (B in Fig. 1) or based on net emissions that account for our site-specific nutrient emissions modelling (C in Fig. 1). In pathway C, we applied ReCiPe 2008 by multiplying net emissions of nutrients to freshwater (NO₃⁻) and to air (NH₃, N₂O), with FF_{net emission} for nutrient emission to freshwater from a sewage treatment plant (corresponding to direct emissions in freshwater (Struijs et al. 2009)), and NH₃ and N₂O emission to air. To apply ReCiPe 2015, we used the FF developed by Helmes et al. (2012): on average in NZ, the persistence time of P in freshwaters is 6.2 days (D in Fig. 1).

Eutrophication potential indicator – With CML, eutrophication potential (terrestrial and aquatic) is calculated based on an equivalency factor to convert all nutrient flows in terms of phosphate equivalents (PO₄³⁻_{eq}) (Heijungs et al. 1992). With ReCiPe 2008, freshwater eutrophication potential is calculated using P emissions in freshwater from a sewage treatment plant as the reference emission (with an eutrophication potential equal to 1). Similarly, marine water eutrophication potential is calculated using N emitted in freshwater from a sewage treatment plant as the reference emission. Eutrophication potential assessed with ReCiPe 2015 is similar to ReCiPe 2008, but does not consider marine eutrophication anymore since an endpoint model is lacking (Huijbregts et al. 2015).

Endpoint effects modelling – CML method does not assess effects from a nutrient increase. ReCiPe 2008 and 2015 methods evaluate effects on freshwater eutrophication only, focusing on P, using an EF developed by Struijs et al (2011). The effect model is based on a stressor-response relationship for Dutch freshwater ecosystems. More recently, the effects modelling for P emissions has been improved by accounting for more species and freshwater types (Azevedo et al. 2013a, b). Regarding marine eutrophication (focused on N), very recent work seems promising, but is only partially published so far (Cosme et al. 2015, Cosme and Hauschild 2016) (Fig. 1).

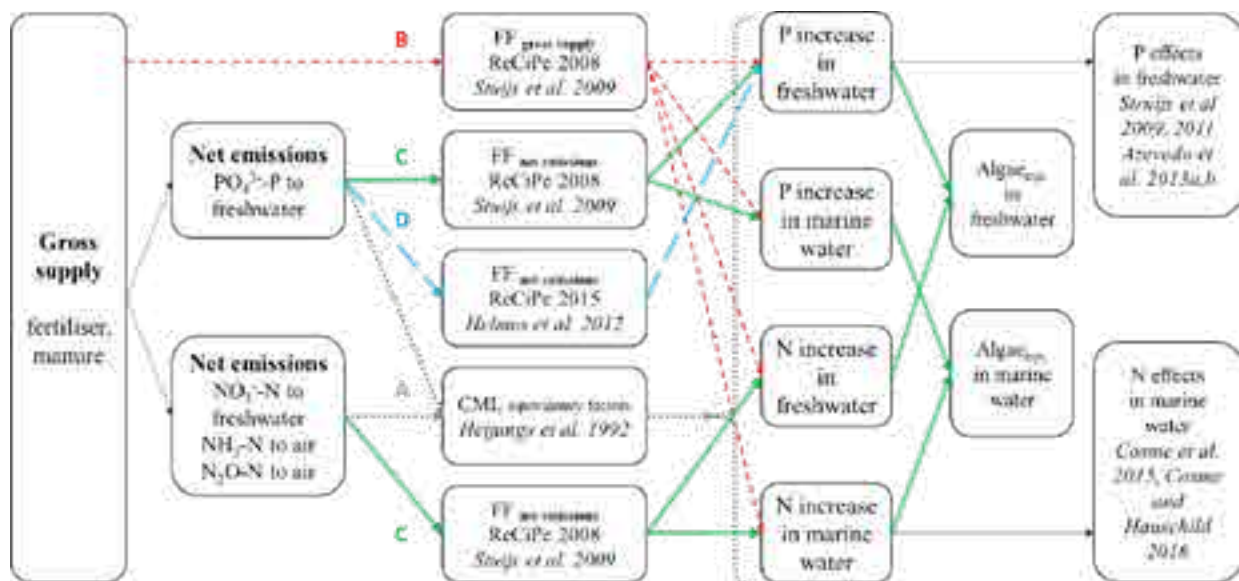


Figure 1: Nutrient inventory flow requirements, fate factors (FF) and effects modelling with different methods throughout the eutrophication cause-and-effect chain in LCA. Capital letters refer to pathways and methods applied in this work.

3. Results and discussion

3.1. Inventory of nutrient flows

Table 1 shows the inventory of nutrient flows for the sheep & beef and dairy farms. N and P emissions are 3.6 times higher per-hectare for dairy farms, but dairying land only represent less than 3% of the pastoral land area of the Lake Taupo catchment (most is in sheep & beef farming; Vant and Huser (2000). Converting sheep & beef pasture to dairying would increase the overall N load to the lake by 20-60% (Vant 2000).

Table 1: Nutrient flows inventory results [kg N or P/ha/year] and emission compartment.

Nutrient flow	Compartment	Sheep & Beef		Source
		Sheep	Dairy	
N fertiliser	agri. soil	17	103	Primary data
N fixation (clover/rain)	agri. soil	63	122	
N excreta dung & urine	agri. soil	162	368.6	
N to farm dairy effluent (FDE)	agri. soil	-	19.4	OVERSEER
N out in wool, meat, milk		19	62	
NO ₃ ⁻ -N leaching (below root-zone)	soil	13	49	
NH ₃ -N to atmosphere	air	17.9	49.1	(MfE 2015)
N ₂ O-N (direct) to atmosphere	air	1.3	3.7	(MfE 2015)
N ₂ O-N (indirect) to atmosphere	air	0.3	0.8	(MfE 2015)
P in excreta (manure P)	agri. soil	15.1	30	Function of DMI & P out
P fertiliser	agri. soil	22	45	Primary data
PO ₄ ³⁻ -P runoff to water	freshwater	1.1	3.0	
P out in wool, meat, milk		3	11	OVERSEER

Where DMI=dry matter intake

3.2. Eutrophication impacts

Comparison of the results from different methods is not straightforward; not only because the methods address different processes of nutrient fate, but also because the rationale and units of indicators are different.

Nutrient fate – We compared fate estimates from ReCiPe 2008 using two methods, one based on gross supply of fertiliser and manure (B in Fig. 1) and the other using net emissions to air and freshwater (C in Fig. 1). N fate estimates were lower when based on net emissions (by 24% on average). The leaching fraction estimated with OVERSEER combined with the 50% attenuation factor from root zone to freshwater was less than N estimated to reach freshwaters using the CARMEN model. Also, P fate estimates were lower when based on net emissions (by 22% on average), highlighting that except for plant uptake and topsoil binding, no other P transport or attenuation process is accounted for in CARMEN, whereas our estimate of P runoff accounts for P attached to sediments lost from soil and P accumulation in the soil.

In the following section, we focus on the fate of P because ReCiPe 2008 (CARMEN) only provides an estimate of N fate, not allowing a comparison between methods. First, we compared the FF for net emissions in freshwater with the FF for gross supply of manure and fertiliser. Struijs et al. (2011) reported a difference of a factor of 18 higher for freshwater emissions of P versus agricultural

emissions of P, which is similar to the factor of 20 from Huijbregts and Seppälä (2001), but is higher than the factor of seven reported by Potting et al. (2005). We cannot do a similar comparison with Helmes et al. (2012) because this method does not provide FF from agricultural emissions, since it focuses on direct emission to freshwater. We also compared the P increase in freshwater estimated with ReCiPe 2008 vs. 2015. To allow a comparison, ReCiPe 2008 FF has to be multiplied by the total volume of European freshwater (885 km³, according to Struijs et al. 2009), to convert a dimension of concentration (kg P/km³) to a dimension of mass (kg P). Results showed that the P fate impact using ReCiPe 2008 was 18 times higher than that using Helmes et al. (2012) (Table 2). This is because P fate modelling in the CARMEN model only accounts for the advective transport of P and does not include the P removal processes through retention and water use modeled by Helmes et al. (2012).

Eutrophication potential – CML eutrophication potential corresponds to a nutrient emission expressed in phosphate equivalents, and is totally compartment-generic. Conversely, ReCiPe methods are specific to the receiving compartment, and focus on P emissions to freshwater (disregarding N emissions), and N emissions to marine water (disregarding P emissions). Thus, any comparison of these indicators would be inappropriate (Table 2).

Endpoint effects – It was not possible to assess effects (or damages) of an increase in nutrients in aquatic compartments with the current methods because we are outside the domain of validity of the equations for P effect (Struijs et al 2009, 2011, Azevedo et al. 2013a), and equations for N effects have just been published (Cosme and Hauschild 2016).

Table 2: Increase in nutrients in water bodies and eutrophication potential impact indicator results for different methods, expressed per ha

Method	Pathway on Fig.1	Impact indicator	Dimension	Compartment	Sheep & Beef	Dairy
CML	A	Eutrophication potential	kg PO ₄ ³⁻ _{eq}	Terrestrial & aquatic	14.38	42.27
ReCiPe 2008 using FF for net emissions	C	Increase in Phosphorus (Emission x FF _{net emission})	kg P	Freshwater	0.34	0.91
		Nutrient increase and N&P aggregation in algae equivalent	kg algae/km ³	Marine water	0.02	0.06
		Marine eutrophication potential (N increase/FF _{net N emission to freshwater})	kg N _{eq}	Marine water	8.70	30.55
		Freshwater eutrophication potential (P increase/FF _{net P emission to freshwater})	kg P _{eq}	Freshwater	1.10	3.00
ReCiPe 2015	D	Increase in Phosphorus	kg P	Freshwater	0.019	0.05

4. General discussion and implications

4.1. Differences in inventory of nutrient flows

Default factors are not appropriate for field-specific estimates of emissions: Default volatilisation rates in ReCiPe are 21% of N in manure and 7% in fertiliser, whereas our NZ-specific volatilisation rates were 10% of N in manure and fertiliser (MfE 2015). Differences in terms of technosphere and ecosphere boundary, depending on the method, are confusing for the LCA practitioner, since they rely on different inventory requirements. There is a lack of guidelines for good practices. The second Pellston workshop on “Global guidance for LCIA indicators and methods” to be held in 2017 will help in this direction.

4.2. Differences in fate modelling

There is a need for a globally valid model, but with site-specific characterisation factors. ReCiPe 2008 is not appropriate for NZ since it is specific to Europe. Nevertheless, in the absence of FF for

other continents, this method has been used outside Europe, such as in Central or South America recently (Huerta et al. 2016, Willers et al. 2016). ReCiPe 2008 is recommended by the European Commission but is not transparent as the modelled fate processes and associated assumptions used in the CARMEN model have not been published (Beusen 2005). The Helmes et al. (2012) method is a significant improvement toward a global nutrient fate model, but it only focuses on P and freshwater. The FF for NZ (6.2) showed a standard deviation of 19.6 days (persistence time of P in freshwater). As a result, a NZ country average FF is not appropriate: we should use a finer resolution such as the Lake Taupo catchment scale, because local hydrological properties have the largest effects on these fate factors (Helmes et al. 2012).

N fate in freshwater - The long time lag for leached N to groundwater of around 40 years observed in the Lake Taupo catchment (Vant 2013) is not reflected by the fate factor. This time lag is related to the deep groundwater in the catchment. In NZ, there is ongoing research on the characterisation of N attenuation according to the site-hydrogeological specificities. The reported uncertainty for attenuation factor ranges from 0 to 0.8 (Elliot et al. 2014). This uncertainty (due to natural variation of denitrification processes) has an influence on the eutrophication impact result. Future work should use site-specific fate modelling of nutrients currently under development in several catchments in NZ (Stenger et al. 2016).

P fate in freshwater - Sediment in lakes can act as an internal source of P, but varies with lake properties (Özkundakci et al. 2010). Similarly, Scherer and Pfister (2015) recently showed that the site-dependent P concentration in soil is one of the most important parameters influencing P emissions to water from agriculture. This illustrates the preference for use of spatially-explicit fate models.

4.3. Sensitivity of receiving water bodies

Accounting for the sensitivity of water bodies to eutrophication drivers is relevant, but doing so by focusing on a single nutrient may be inappropriate. Lake Taupo is N-and P-limited, so the freshwater effect model focused on P is only capturing part of the problem. To avoid using the concept of limiting nutrient, N and P nutrients in each receiving compartment (marine and freshwater) were aggregated using conversion factors for P and N in terms of algae, based on the Redfield Ratio (Redfield 1963 used by Goedkoop et al. 2008), assuming that one mole of algae biomass contains one mole of P and 16 moles of N (Table 2). This allows an impact indicator to be obtained that reflects an increase of both N and P nutrient in a water compartment.

4.4. Nutrient effects modelling and policy

The concentration of nutrients in Lake Taupo is lower than most European freshwaters at 0.079 mg.L⁻¹ total N, and 0.0052 mg.L⁻¹ total P on average between the years 2010 - 2014 (Vant 2013). These concentrations are so low that they are outside the domain of validity of the effect factor equation. Thus, it was not possible to assess any effect from an increase in nutrients in Lake Taupo with actual methods. The effect factor equations are valid for concentration of total P above 0.1 mg.L⁻¹ (Struijs et al. 2011) or above 0.05 mg.L⁻¹ (for temperate lake, according to Azevedo et al. 2013a). These concentrations correspond to optimum nutrient level for ecosystems, and are consistent with current water quality policies. Indeed, Struijs et al. (2011) found the highest number of species for an average total P concentration of 0.1 mg.L⁻¹, which is just below the regulatory water quality objectives for European lakes (0.15 mg.L⁻¹) (European Commission 2000). In NZ, the national bottom lines were set at 0.05 mg.L⁻¹ for total P and 0.75 mg.L⁻¹ for total N in lakes (MfE 2014). But for Lake Taupo, the objectives of water quality are more strict: the objectives is to stay below 0.0703 mg.L⁻¹ total N and 0.0056 mg.L⁻¹ total P (WRC 2016). In this case, LCA fails to account for a high standard of water quality that is in a near-pristine state. Nevertheless, quantifying the eutrophication impacts of dairy and sheep & beef farms in the Lake Taupo catchment is highly relevant because these farms are monitored and have a maximum nitrogen discharge allowance (WRC 2016).

5. Conclusions

The application of eutrophication potential indicators suffers from a lack of transparency of methods, what processes to account for, and a lack of clear guidelines of inventory requirements for LCA practitioners. The inventory of nutrient flows at a farm scale and fate factors modelled at a catchment scale should be site-specific (the relevant scale has to be determined). The farm inputs play an important role in the impact, but the fate modelling (transport, attenuation) and the sensitivity of the receiving compartment plays an important role as well. Since LCA involves inventories across many countries on a global scale, the challenge is to have site-specific characterisation factors that are defined with a global coverage.

Considering that the main currently-accepted freshwater eutrophication indicators are only based on P, we could not assess impacts on Lake Taupo, which is co-limited by N and P, and thus could not use LCA as a tool to support current policies on water quality regulation.

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202. Identifying hotspots to improve the environmental performance of farms: Main drivers and their variability

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ABSTRACT

The high variability of environmental impacts among farms is a challenge for the mitigation of the environmental impact of agriculture. Often, the difference between farms of the same type is larger than differences between means of farm types. In this paper, we studied in detail how individual groups of input data contribute to various environmental impacts, and the variability of these input groups among farms. With this, we suggest an approach to identify areas with a high improvement potential.

We used LCA results from a sample of 51 farms in Austria, which we obtained in the FarmLife project. The farm sample comprised 11 arable, 6 winegrowing, 12 cattle and pig fattening, and 22 dairy farms; 30 farms were organic, 21 conventional. The functional unit was one hectare of agricultural land. The impact categories we analyzed were non-renewable energy use, use of phosphorus resources, global warming potential, and terrestrial ecotoxicity.

Special focus areas became apparent using our approach. Fertilizers and field emissions were a prominent hotspot in the environmental impacts of arable and wine farms. At farms with livestock, purchased goods could be considered as hotspots for environmental impacts, above all purchased concentrate feed, but also fertilizers and animals. Direct animal and field emissions were also important with a high variability, but only in some impact categories. A special hotspot of the conventional farms were the concentrates purchased, while on the organic farms, each impact category had its own decisive input group.

We were able to show that in order to reduce the environmental impact of agriculture, it is advisable not only to address the absolute values of contributing factors, but also their variability in a larger sample of farms. This yields in-depth information for advising farmers, for decision makers, but also for more focused research activities.

Keywords: input groups, types of farming, farm sample, standard deviation, contributing factors

1. Introduction

The agricultural sector contributes largely to the environmental impact of European economies (Jungbluth et al., 2011; Tukker et al. 2006). Defining ways to reduce this impact is hampered by the fact that there is a huge number of individual farms rather than a few big companies, as would be the case in some other sectors of the economy. Environmental impacts of individual farms vary highly, even among farms of the same farming type (Hersener et al., 2011). While the type of farming is often considered explanatory for environmental effects, and choosing another farming system such as organic farming is seen as a way to improve at least some of the environmental impacts of farms (Tuomisto et al., 2012, Mondelaers et al., 2009), the variability of impacts among individual farms is indeed often larger than differences between mean values of farm types or farming systems (Hersener et al., 2011, Küstermann & Hülsbergen, 2008). This suggests that there might be high improvement potential for individual farms, even without changing the orientation of production. As information on the farming type alone cannot explain the wide range of results, it is necessary to analyze farms at a deeper level than that in order to improve the environmental impact of farming in general. In order to find areas with a high improvement potential, it is necessary to trace back environmental impacts to different inputs and activities on farms. Analyzing these factors should reveal the effect of farmers' activities on environmental impacts, so that ultimately, practicable solutions can be found.

Using LCA results from a sample of 51 farms in Austria, we studied in detail individual factors contributing to impact categories and the variability of the impact of these factors among farms. Our approach was to find groups of life cycle inventory data that contribute highly to impact indicator results and, at the same time, have a high standard deviation between farms. This points to areas with a high potential for optimization (Hersener et al., 2011). In this paper, we want to find whether this approach indeed leads to finding hotspots for improving the environmental performance of agriculture. We want to identify special fields of improvement in our sample farms, and to find

whether these differ between farming types with and without livestock, and between conventional and organic farms.

2. Methods

The LCA data from the 51 Austrian farms we investigated were obtained during the project “FarmLife”, which ran from 2012 to 2015. The farm sample comprised 11 arable farms, 6 winegrowing farms, 12 cattle and pig-fattening farms, and 22 dairy farms. 30 farms were organic, 21 were managed conventionally. Five of the arable farms mainly cultivated bread cereals, two mainly grain maize, while the others were more mixed (with cereals, maize, grass, root crops, and other crops). The winegrowing farms produced grapes exclusively, except one that produced also fruit. Among the fattening farms, four had suckler cows, four had only fattening cattle, four had fattening pigs. Eight of the fattening and twelve of the dairy farms did arable farming as well. Five of the dairy farms had pigs in addition to cattle.

On average, the farms had 32 ha of agricultural land and were therefore smaller than the average of farms in Austria (44 ha, BMLFUW, 2014). The winegrowing farms were smallest (6-15 ha), two of the arable farms with 115 and 156 ha were the largest by far. Winter wheat yields were 3.8 t ha^{-1} (3 t ha^{-1} on organic, 5.6 t ha^{-1} on conventional farms), grain maize yields were 7 t ha^{-1} , and silage maize yields were 15.3 t ha^{-1} on average, which was below the average yields in Austria. The lower productivity was because our sample contained a higher percentage of organic farms than the country average, but could also be attributed to the generally low fertilization level of the farms. On average, they applied 81 kg N ha^{-1} , varying between 0 and 224 kg N ha^{-1} (organic farms used 54 kg N ha^{-1} on average, conventional farms 120 kg N ha^{-1}).

The temporal system boundaries were one year, *i.e.* the period from the 2012 harvest until the 2013 harvest for annual crops, and the calendar year 2013 for perennial crops and all farm activities that were not related to crop management. The spatial system boundary was the farm gate; that means we did not consider any downstream processes. In the project FarmLife, we calculated LCA results with two functional units, ha of agricultural land managed by each farm, and MJ digestible energy contained in the farms’ products. All LCA results are presented in Bystricky et al. (2015). For this paper, we used the results per hectare of agricultural land to test our approach; this functional unit serves to answer questions on how to use the available area in a sustainable way, or how to minimize the environmental impact of a country’s agricultural land. In future, we are planning to do the same analysis with the results per MJ digestible energy.

A full life cycle assessment was done of each farm, adapting the SALCA method (Swiss Life Cycle Assessment, Gaillard & Nemecek, 2009) to Austrian conditions as part of a newly developed tool set “FarmLife”, which uses LCA for advisory services (Bystricky et al., 2014). Derived from SALCA, the FarmLife tool set comprises models for direct field and animal emissions, a life cycle inventory database for special agriculture-related background processes that are compatible with the ecoinvent database, and a set of impact categories. The FarmLife tool set has its own newly developed data collection tools, as well as a concept for evaluation and communication (see Baumgartner et al., 2016). Adapting the SALCA method to be applicable in Austria affected above all the emission models, but we also added inventories to the background database. We calculated potential emissions of phosphorus, nitrate, heavy metals, ammonia, nitrogen oxides, and methane.

As given by SALCA/FarmLife, we assessed non-renewable energy use, use of phosphorus and potassium resources, land use, deforestation, water use, global warming potential, ozone depletion, ozone formation, terrestrial eutrophication, aquatic eutrophication with nitrogen and phosphorus, acidification, terrestrial and aquatic ecotoxicity, and human toxicity. In order to group these impact categories and select representative results, we calculated the correlation of all pairs of impact category results using Pearson’s correlation coefficient. In addition to this, we analyzed the contribution of different groups of life cycle inventory data (“input groups”) to each impact category.

The input groups are given as default by SALCA; they are: Buildings & equipment, machinery, energy carriers, fertilizers & field emissions, pesticides, seeds purchased, concentrates purchased, roughage purchased, animals purchased, animal husbandry on farm, others. These two steps allowed us to form the following groups of impact categories (see Bystricky et al., 2015): energy- and infrastructure-related, nutrient-resource-related, and animal-husbandry-related impact categories. Aquatic eutrophication with nitrogen and phosphorus, terrestrial ecotoxicity, deforestation, and land use were unrelated to other impacts and had to be considered separately. For the three groups, we chose non-renewable energy use, use of phosphorus resources, and global warming potential as representative impact categories (Bystricky et al., 2015); in this paper, we present results of these three plus terrestrial ecotoxicity.

We present the contribution of different input groups to the results, and the standard deviation within each input group as a measure for variability. A large contribution to an impact combined with a high standard deviation denotes production sectors with high potential for environmental improvement. For our analysis, we looked at the sample of farms as a whole, and separately at different groups of farms (conventional/organic, arable/winegrowing/fattening/dairy). We expected thus to find more detailed explanations for the variability of farm results, as some input groups are only relevant for farms of a specific farming type.

3. Results

As an example for our LCA results, figure 1 shows the global warming potential of the 51 farms. It illustrates the variability of farms, and the overlapping of different farm types. The global warming potential of arable and winegrowing farms was mainly determined by fertilization, which takes into account the production of purchased organic and mineral fertilizers and greenhouse gas field emissions. At the fattening farms, the global warming potential caused by infrastructure and fuels correlated with the total global warming potential. Apart from that, different input groups played a different role at each fattening farm. At the dairy farms, the percentage contribution of many input groups remained more or less constant, that means the global warming potential of different input groups was proportional to the total.

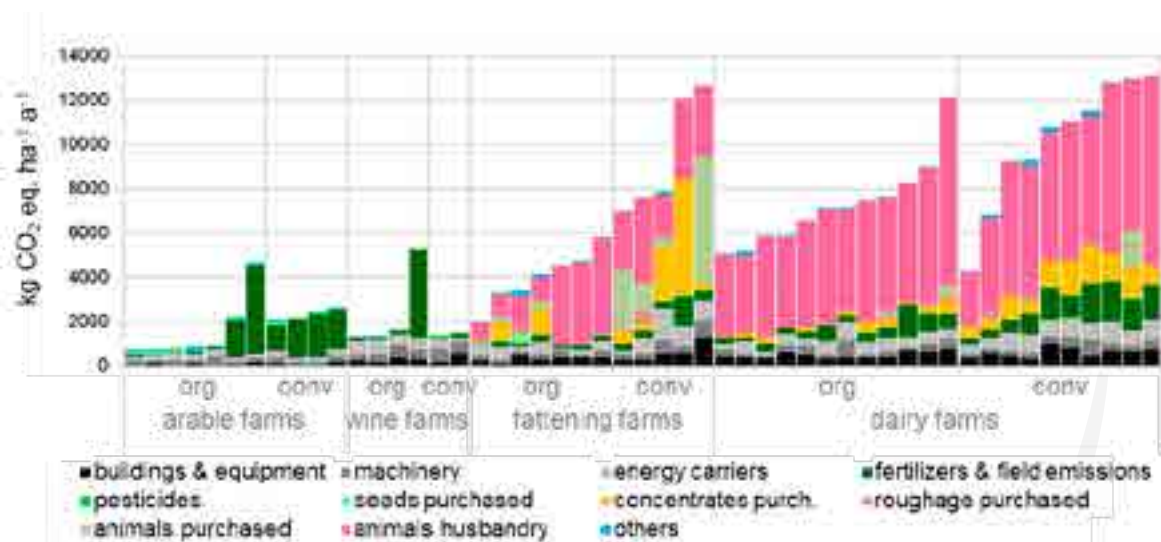


Figure 1: Global warming potential per hectare of agricultural land and per year of 51 farms.

Table 1 shows the contribution and the standard deviation of different input groups for non-renewable energy use, use of phosphorus resources, global warming potential, and terrestrial ecotoxicity. Looking at the total sample of farms, each impact category had its own decisive input groups, which already might indicate some special focus areas for improvement. While buildings, equipment, machinery, and energy carriers had a large contribution to non-renewable energy use,

concentrates purchased had the highest variability among farms. These, together with the fertilizers used, were also decisive for use of phosphorus resources. Regarding global warming potential, the input group “animal husbandry” had the highest average contribution and standard deviation. Terrestrial ecotoxicity had the highest variability at heavy metal input via fertilizers and at the production of purchased feedstuff. The variability was particularly high at input groups related to animal husbandry, and also to some extent at input groups related to mineral fertilizer use. Therefore, the variability of the total farm sample was largely caused by the fact that different farming types were put together in our sample, which confirmed our approach to look at different farming types separately.

This led to a more differentiated picture. In all farm types, buildings, equipment, machinery and energy carriers had a high contribution to non-renewable energy consumption, but the standard deviation there was comparatively low. In organic farms, with a low contribution and standard deviation of all other input groups, these three input groups still had the strongest influence on variability of results. In conventional farms, conversely, the highest variability arose from the purchase of concentrates. This also applied to all livestock-keeping farms in general. Fertilizer use had a high contribution to total non-renewable energy consumption of arable farms and a high standard deviation, while in winegrowing farms, the standard deviation was rather low in all input groups.

For the use of phosphorus resources, use of mineral fertilizers was decisive throughout, with a high standard deviation. In the livestock-keeping farms, concentrates purchased were important, too, but this could also be traced back to fertilizers, which were used in producing feedstuff. The winegrowing farms in our sample, as well as all organic farms, were all rather extensive and did not use much fertilizer, so all the figures were very low.

At arable and winegrowing farms, the input group “fertilizers and field emissions”, containing N_2O , CO_2 , and CH_4 emissions, had both the highest contribution to the total global warming potential and a high standard deviation among farms, with N_2O being most dominant. In fattening farms, the standard deviation of global warming potential was highest for concentrates and animals purchased, but animal husbandry also had a high standard deviation, together with the highest overall contribution. Animal husbandry was also the input group with the highest influence at dairy farms. The global warming potential of organic farms, while being mostly driven by animal husbandry, had some variability concerning fertilizers and field emissions, which was caused by very few of our farms which had purchased compost. The conventional farms were influenced by livestock-keeping in that concentrates and animals purchased showed some variability, while animal husbandry was the most important contributor to global warming potential.

Terrestrial ecotoxicity at arable and winegrowing farms was affected mostly by heavy metal field emissions. The farms’ heavy metal surplus was derived from imports via purchased goods such as fertilizers, and exports via the products. The influence of fertilizers and field emissions was also visible at livestock-keeping farms, but was exceeded by the contribution and standard deviation of concentrates purchased. The use of pesticides played a minor role, too. At organic farms, field emissions were more important, in terms of percentage contribution and standard deviation, than at conventional farms; this was caused, again, by those organic farms that purchased compost as fertilizer. At the conventional farms, concentrates purchased, with a high standard deviation, were more important.

Table 1: Non-renewable energy use (in MJ ha⁻¹ a⁻¹), use of phosphorus resources (in kg P ha⁻¹ a⁻¹), global warming potential (in kg CO₂ eq. ha⁻¹ a⁻¹), and terrestrial ecotoxicity potential (in kg 1,4-DB eq. ha⁻¹ a⁻¹) of the complete farm sample (n = 51) and of different groups of farms. Mean values and standard deviation (SD) for each input group.

Non-renewable energy consumption		Buildings & equipment	Machinery	Energy carriers	Fertilizers & field emissions	Pesticides	Seeds purchased	Concentrates purchased	Roughage purchased	Animals purchased	Animal husbandry	Others
All farms	Mean	5'424.751	6'212.610	7'573.572	1'351.780	102.017	366.532	4'365.467	692.525	1'069.687	0.000	831.276
	SD	3'867.434	3'099.095	4'134.230	2'464.566	286.336	653.075	7'684.084	1'266.112	3'433.097	0.000	1'095.395
Arable farms	Mean	640.426	4'410.467	3'854.948	3'585.880	52.763	713.097	0.000	0.000	0.000	0.000	807.439
	SD	620.127	1'702.330	1'088.523	3'931.896	91.352	497.073	0.000	0.000	0.000	0.000	867.644
Wine farms	Mean	4'323.924	9'095.414	7'380.862	908.312	674.084	128.148	0.000	0.000	0.000	0.000	139.999
	SD	2'270.955	2'307.635	2'426.734	2'224.900	590.239	220.708	0.000	0.000	0.000	0.000	342.925
Fattening farms	Mean	5'158.324	6'485.934	8'240.158	420.282	23.310	593.756	7'431.653	1'143.521	3'848.162	0.000	878.617
	SD	2'750.356	3'506.441	5'819.126	1'275.368	43.782	1'150.455	12'348.284	1'942.571	6'275.809	0.000	1'130.414
Dairy farms	Mean	8'262.464	6'178.376	9'121.850	863.766	13.556	134.322	6'066.317	981.660	380.731	0.000	1'005.902
	SD	3'070.163	3'108.034	3'312.403	1'285.971	27.788	166.755	6'073.126	1'117.173	1'263.604	0.000	1'277.108
Organic farms	Mean	4'720.227	5'540.151	6'278.231	522.105	43.676	403.330	1'231.146	563.533	328.726	0.000	715.551
	SD	3'474.451	2'787.112	2'639.813	1'422.316	132.229	812.154	1'968.231	977.474	987.976	0.000	833.893
Convent. farms	Mean	6'431.215	7'173.265	9'424.061	2'537.030	185.360	313.962	8'843.069	876.800	2'128.202	0.000	996.597
	SD	4'252.099	3'331.678	5'149.056	3'121.584	408.925	323.916	10'305.743	1'600.492	5'103.923	0.000	1'393.788

Use of phosphorus resources		Buildings & equipment	Machinery	Energy carriers	Fertilizers & field emissions	Pesticides	Seeds purchased	Concentrates purchased	Roughage purchased	Animals purchased	Animal husbandry	Others
All farms	Mean	0.010	0.004	0.081	2.678	0.060	0.182	2.010	0.246	0.094	0.000	0.000
	SD	0.007	0.002	0.479	5.665	0.235	0.262	5.005	0.596	0.316	0.000	0.000
Arable farms	Mean	0.003	0.003	0.308	6.018	0.026	0.305	0.000	0.000	0.000	0.000	0.000
	SD	0.004	0.001	1.016	9.337	0.076	0.273	0.000	0.000	0.000	0.000	0.000
Wine farms	Mean	0.006	0.005	0.002	0.000	0.406	0.221	0.000	0.000	0.000	0.000	0.000
	SD	0.003	0.001	0.001	0.001	0.607	0.360	0.000	0.000	0.000	0.000	0.000
Fattening farms	Mean	0.013	0.004	0.003	1.480	0.007	0.213	4.476	0.410	0.353	0.000	0.000
	SD	0.008	0.002	0.002	2.843	0.016	0.321	9.259	0.907	0.580	0.000	0.000
Dairy farms	Mean	0.013	0.004	0.033	2.391	0.013	0.094	2.217	0.346	0.026	0.000	0.000
	SD	0.005	0.002	0.142	4.675	0.042	0.160	2.819	0.582	0.106	0.000	0.001
Organic farms	Mean	0.008	0.003	0.024	0.187	0.010	0.149	0.332	0.071	0.007	0.000	0.000
	SD	0.006	0.002	0.122	1.025	0.039	0.260	0.993	0.156	0.026	0.000	0.000
Convent. farms	Mean	0.012	0.004	0.163	6.235	0.132	0.231	4.406	0.495	0.219	0.000	0.000
	SD	0.008	0.002	0.735	7.491	0.355	0.263	7.138	0.861	0.470	0.000	0.001

Global warming potential		Buildings & equipment	Machinery	Energy carriers	Fertilizers & field emissions	Pesticides	Seeds purchased	Concentrates purchased	Roughage purchased	Animals purchased	Animal husbandry	Others
All farms	Mean	391.567	319.515	510.934	751.129	5.537	43.589	471.172	70.524	255.215	2'956.118	48.514
	SD	276.045	160.400	274.194	885.649	15.451	69.764	870.082	129.281	911.247	2'668.546	64.172
Arable farms	Mean	64.852	227.714	268.611	1'147.213	2.826	87.683	0.000	0.000	0.000	0.000	49.365
	SD	60.267	89.478	71.487	1'257.880	4.882	51.385	0.000	0.000	0.000	0.000	54.530
Wine farms	Mean	310.000	459.461	496.977	757.741	36.812	17.771	0.000	0.000	0.000	0.000	8.558
	SD	163.535	116.268	163.296	1'555.633	31.333	29.923	0.000	0.000	0.000	0.000	20.962
Fattening farms	Mean	438.344	336.279	552.478	327.199	1.223	66.156	895.960	117.129	901.868	2'569.460	52.092
	SD	308.124	183.862	387.535	342.179	2.319	119.081	1'522.699	196.372	1'717.669	1'306.219	68.432
Dairy farms	Mean	551.656	318.105	613.241	782.517	0.717	16.274	603.557	99.600	99.707	5'451.296	57.035
	SD	194.138	162.316	220.887	539.131	1.498	20.208	535.520	115.940	338.557	1'542.536	72.741
Organic farms	Mean	323.044	285.492	422.635	596.287	2.468	49.269	156.933	56.325	46.484	2'546.628	43.462
	SD	210.296	145.398	176.712	995.820	7.302	86.380	266.555	99.816	125.143	2'503.849	51.892
Convent. farms	Mean	489.458	368.120	637.075	972.331	9.922	35.474	920.085	90.810	553.401	3'541.104	55.732
	SD	330.563	171.609	338.045	660.068	22.028	35.083	1'195.768	163.104	1'376.410	2'846.327	79.357

Terrestrial ecotoxicity		Buildings & equipment	Machinery	Energy carriers	Fertilizers & field emissions	Pesticides	Seeds purchased	Concentrates purchased	Roughage purchased	Animals purchased	Animal husbandry	Others
All farms	Mean	0.261	0.095	0.054	1.448	0.653	0.074	1.828	0.140	0.115	0.000	0.009
	SD	0.190	0.047	0.051	3.232	1.221	0.170	3.430	0.559	0.324	0.000	0.014
Arable farms	Mean	0.022	0.069	0.036	3.751	0.686	0.105	0.000	0.000	0.000	0.000	0.008
	SD	0.021	0.027	0.068	5.087	1.550	0.199	0.000	0.000	0.000	0.000	0.008
Wine farms	Mean	0.385	0.137	0.028	1.871	2.235	-0.017	0.000	0.000	0.000	0.000	0.002
	SD	0.278	0.033	0.009	4.583	1.043	0.041	0.000	0.000	0.000	0.000	0.006
Fattening farms	Mean	0.245	0.101	0.061	0.707	0.504	0.156	2.391	0.355	0.433	0.000	0.007
	SD	0.134	0.056	0.072	2.167	1.169	0.242	4.517	0.946	0.558	0.000	0.008
Dairy farms	Mean	0.356	0.094	0.067	0.584	0.286	0.040	2.934	0.132	0.031	0.000	0.013
	SD	0.119	0.046	0.024	1.080	0.750	0.105	3.628	0.483	0.105	0.000	0.018
Organic farms	Mean	0.219	0.086	0.039	1.643	0.269	0.022	0.358	0.052	0.054	0.000	0.008
	SD	0.160	0.042	0.022	3.996	0.769	0.106	0.599	0.528	0.187	0.000	0.009
Convent. farms	Mean	0.322	0.109	0.076	1.168	1.201	0.149	3.929	0.266	0.203	0.000	0.012
	SD	0.215	0.051	0.069	1.680	1.528	0.215	4.585	0.590	0.445	0.000	0.018

4. Discussion

Our farm sample showed that LCA results of different farm types are overlapping, so that the type of farming cannot be used to generally explain environmental impacts of individual farms. Results from other authors also permit this conclusion (Paulsen et al., 2014; Küstermann & Hülsbergen, 2008). Nevertheless, looking in more detail, we found it useful to consider different types of farming separately when looking for solutions to reduce environmental impacts of individual farms, as the contribution of input groups was specific for different farm types.

Our approach showed some hotspot input groups, which were different for each impact category and farm type. The high variability of our farms regarding these indicates possibilities for improvement of individual farms. Fertilizers and field emissions were a prominent hotspot in the environmental impacts of arable and wine farms. For the resource-use-related impacts (non-renewable energy use and use of phosphorus resources), the amount of fertilizers purchased and their production was decisive, while global warming potential and terrestrial ecotoxicity were determined by a variety of direct emissions. As this input group comprises all direct field emissions that SALCA calculates next to the impacts of fertilizer production, it might be advisable for future research to divide it into several input and emission groups. At farms with livestock, purchased goods could be considered as hotspots for environmental impacts, above all purchased concentrate feed, but also fertilizers and animals; that is, mostly, inputs from agricultural production outside the own farm. Direct animal and field emissions were also important with a high variability, but only at some of our considered impact categories. Looking at the farming system, a special hotspot of our conventional farms were the concentrates purchased, while on the organic farms, each impact category had its own decisive input group. Regarding all results, the functional unit has to be taken into consideration. Using another functional unit, for example product-related, could lead to different conclusions regarding variability of farms and input groups (Samson et al., 2012). We plan to do the same analysis with MJ digestible as functional unit in the future.

The other input groups could be considered less important than those whose overall contribution was comparatively large throughout and whose standard deviation was always larger than the average contribution to any environmental impact. Buildings and equipment, machinery and energy carriers were only important for use of non-renewable energy, and there, the standard deviation, that is the variability, was lower than the average contribution of the input groups. The use of pesticides on the farms played a role only at terrestrial ecotoxicity, and was exceeded by heavy metal emissions and by emissions from purchased feedstuff. Seeds and roughage purchased featured neither in terms of percentage contribution nor in terms of variability.

The input groups that we analyzed showed fields of activity that the farmers can influence to quite different degrees. While field and animal emissions occur directly on the farm, they have complex causes, and the farmers might have several points where they can adjust these emissions. Using less nitrogen fertilizer would lead to lower greenhouse gas emissions, and using less purchased fertilizer in general would reduce use of non-renewable energy and of phosphorus resources. As it is, our farms were already comparatively extensive regarding the amount of fertilization, and had rather low crop yields to begin with, so this might present an option only for the more intensive farms. However, it is necessary to do the same analysis as we did in this paper with a functional unit representing productivity (e.g. digestible energy), thus addressing the question of the eco-efficiency of farms, and which input group is the best starting point to improve eco-efficiency. Emissions from animal husbandry have a huge variety of causes, too, from the feeding strategy to restocking rate to liveweight gain to manure management. As a hotspot for global warming potential, this needs a detailed analysis of reasons for high or low emissions in the farm sample in order to find solutions for individual farms. An easier point of improvement might be pesticide use, as the impact of pesticides mostly depends not on the amount but on the active ingredients used, which are interchangeable to some degree. This would, however, only reduce ecotoxicity, and only partly.

Other, more widespread hotspot input groups were purchased goods from agricultural production elsewhere. Farmers do not have an influence over the environmental burden carried by these, but only have an influence via the amount they use. Here, again, the correlation of these inputs with the productivity of the farms has to be considered and could be interesting for further analyses. Nevertheless, the variability of the impact of these inputs suggest an improvement potential for individual farms. Apart from that, it shows the general interconnectedness of agricultural production on different farms and in different countries, indicating that the environmental performance of individual farms depends on the impacts of production elsewhere.

5. Conclusions

We were able to show that in order to reduce the environmental impact of agriculture, it is advisable not only to address the absolute values of contributing factors, but also their variability in a larger sample of farms. On the one hand, our approach might be useful for improving individual farms by identifying input groups that have a higher improvement potential than others because of their high variability among farms. On the other hand, this approach can also be used by decision makers to identify tangible starting points for improving the environmental impact of the agricultural sector as a whole. The approach also yields in-depth information for more focused research activities. In our case, the most important hotspots were fertilizer use, direct field and animal emissions, and the impact caused by purchased concentrate feed. Furthermore, some differences became apparent between organic and conventional farms, and between farms with and without livestock. Our sample of 51 farms was, however, too small to draw general conclusions. It enabled us, however, to test this approach and its applicability. Apart from using larger samples of farms, further research is needed using other functional units and dividing some of our input groups into more specific input groups that would better reflect cause-effect relationships in agriculture.

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168. Farmers would use agriculture-specific LCI-methods – Comparing the Cereal Unit allocation and the Crop Rotation approach versus LCI-results of established modeling options

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ABSTRACT

Aiming at bringing LCA modeling closer to agricultural reality, we compare the Life Cycle Inventory (LCI)-results of established LCI methods with respective results of the two approaches: 1. Cereal Unit allocation approach (Brankatschk and Finkbeiner 2014) and 2. Approach for consideration of agricultural crop rotations (Brankatschk and Finkbeiner 2015).

Wheat, barley, rapeseed and pea (a legume) were considered with average statistical yield, composition and grain/straw-ratios. Inputs of N, P₂O₅, K₂O, MgO, crop protection agents and diesel were used from agricultural planning tables. In order to measure methodological differences, inputs and outputs were fixed. Mass, energy, economic and Cereal Unit allocation results were compared. Using aforementioned crops, four crop rotations (*a – d*) were built (see Figure 1). Additionally, straw harvest scenarios 1%, 50% and 100% were considered. LCI-results (specific input per ton of product; e. g. kilograms N per ton of wheat grains) were calculated and compared for all crops in all combinations of: no rotation (modeled in 1-year system boundary), crop rotations, straw harvest scenarios and presence of legume.

Especially for high straw harvest scenarios, substantial differences in LCI-results were found: -42.1 % for rapeseeds in mass allocation compared to Cereal Unit allocation (see Figure 2). Specific N inputs for wheat range from 22.03 kg N/t wheat grains (modeled in 1-year system boundary, that means no crop rotation considered) to 16.70 kg N/t wheat grains (in crop rotation *d*; see Figure 3). Significant differences were found between LCI-results for crops, modeled in 1-year system boundary (no crop rotation) versus grown in rotations. Rapeseeds, grown in crop rotation *d* need 53.8% lower specific N input compared rapeseeds without rotation.

Differences in LCI-results were found for different crop rotations, straw harvest and legumes in rotations. Farmers use these tools in their daily work; they will call LCA practitioners for addressing these issues in LCA-modeling. This work quantifies the relevance of addressing crop rotations and different allocation approaches. The tested new tools help including agricultural practice in LCI and thus strengthen LCA as an appropriate tool towards sustainable agriculture.

Keywords: Modeling Agricultural Practice, Crop rotations, Climate-smart agriculture, United Nations Sustainable Development Goals, Food Security and Climate action

1. Introduction

Interrelationship between climate and agriculture is widely taken as a matter of course. It is worth the effort to understand the versatility and multiple dimensions of these connections. On the one hand, climate change affects agricultural productivity, increases vulnerability of food production and makes adaption measures necessary (IPCC 2014; Porter and others 2014; Thornton and others 2014); on the other hand, agriculture has a particular potential for combating climate change (FAO 2010; 2013; Lipper and others 2014). As a consequence, agriculture is key to achieve both of the United Nations Sustainable Development Goals (SDG): *Food Security* and *Climate Action* (United Nations 2015). This paper focuses on the methods for assessing environmental impacts of agriculture and further improvement of these methods. Agriculture needs assistance in identifying climate-smart practices in order to mitigate its environmental impacts, e.g. climate change, and simultaneously bearing in mind the productive function of agriculture.

Since centuries, farmers are improving land use efficiency, yields, phytosanitary conditions and maintaining soil fertility. A fundamental tool to achieve these aims is the targeted use of crop rotations (Wrightson 1921). Furthermore, remaining crop residues on the field play a significant role for the humus balance and serve as nutritional basis for soil organisms and thus improve soil structure (Brankatschk and Finkbeiner 2015). Whereas provision of sufficient amounts of food can be considered as traditional task of agriculture, the aim of climate action is relatively new for agriculture. For achieving compatibility between both food provision and climate action, robust and reliable

recommendations are needed, as both of the targets are of critical importance. For deriving such recommendations, both targets need to be considered at the same time.

In order to assess environmental interventions, e.g. greenhouse gas emissions, the internationally standardized Life Cycle Assessment (LCA) has been developed (ISO 14040 2006; ISO 14044 2006) and serves as tool to identify environmentally sound options amongst various options. Even though performed since many years and already providing valuable results (Schenck and Huizenga 2014; van der Werf and others 2014), methodological challenges still remain (Grabowski and others 2015). Two of these challenges were selected and proposed solutions tested.

First challenge is the method for allocating environmental burdens between co-products. Such step is needed for several agricultural activities (e.g. harvesting process creates to grains and straw) or processing agricultural products (e.g. grain milling creates to flour and bran). Arbitrariness in the selection of allocation approaches might lead to methodological cherry picking among initiators of LCAs and thus affects LCA results. As a potential solution, the Cereal Unit allocation approach has been proposed as agriculture-specific allocation approach (Brankatschk and Finkbeiner 2014).

Second challenge is the consideration of crop rotation effects between crops grown on the same field in temporal sequence – including yield increase and nutrient shift. Current LCA practice has limited ability to depict crop rotation effects (Brankatschk and Finkbeiner 2015); crops are assessed within one year system boundary, even though they are grown in crop rotations. Although this problem is described since 1990's (Audsley and others 1997; Cowell and others 1995), no consensus has been found yet to integrate these effects. A potential solution to perform a system boundary expansion during the LCI and allocating impacts among products using an agriculture-specific allocation approach as common denominator has been proposed (Brankatschk and Finkbeiner 2015).

Previously described approaches were used in LCI modeling and respective results compared to the results, obtained using current modeling practices.

2. Methods

Agronomic data for establishing the Life Cycle Inventory (LCI) were obtained from publicly available databases, official statistics and publications (Albrecht and Guddat 2004; BioGrace 2015; BMEL and BLE 2015; Brankatschk and Finkbeiner 2014; Eurostat 2015; Eurostat and European Union 2007; Kaltschmitt and others 2009; KTBL 2015; LfL 2013; Richthofen and others 2006; TLL and others 2010).

Using winter wheat (W), winter rapeseed (R), winter barley (B) and spring pea (P), four different **crop rotations** were designed (see Figure 1). In order to avoid unnecessary complex contexts, relatively simple rotations were chosen. In practice much more complex rotations are conceivable. The presented four rotations differ only slightly and are closely related to each other. This allows comparing the effects between growing the same crop in various rotations. A short rotation is represented by Crop rotation *a*. Crop rotation *b* and *c* contain the same crops, whilst wheat is being included several times. Crop rotation *d* is based on rotation *c* and the second wheat has been replaced by Spring pea as a legume.



Figure 1: Assessed Crop rotations a, b, c and d; Winter rapeseed (Rapeseed), Winter wheat (Wheat), Winter barley (Barley) and Spring pea (Pea); schematic representation based on Castellazzi et al. (2008)

Existing approaches to include **crop rotation effects**, such as yield increase, improved phytosanitary conditions and nutrient shift are described in Brankatschk and Finkbeiner (2015). Current LCA practice allows considering a shift of selected main nutrients from current crop to the succeeding crop in the crop sequence of crop rotation. Long-term fertilization strategies and crop rotation planning are not yet considered in current LCA practice at all – even though crop rotation planning is essential for the success of agricultural farms. Brankatschk and Finkbeiner proposed an approach for expanding the system boundaries during the Life Cycle Inventory calculation in order to include the entire crop rotation and allocating the various inputs and outputs using an agriculture-specific allocation approach; for example, this can be accomplished by the Cereal Unit allocation (Brankatschk and Finkbeiner 2015).

Established **co-product allocation** approaches for agricultural systems are energy allocation (based on lower heating value), mass allocation (based on mass of products) and economic allocation (based on market price). Advantages and disadvantages of those approaches are discussed by Brankatschk and Finkbeiner (2014). They proposed a new biophysical and agriculture-specific allocation approach, based on the animal feeding value of various agricultural products. The so called Cereal Unit allocation approach uses the animal feeding value and hereby introduces a common use perspective for all agricultural products and co-products. The Cereal Unit serves as a common denominator for all agricultural products (Brankatschk and Finkbeiner 2014). The Cereal Unit allocation has been included as a benchmark allocation approach in order to compare LCI results.

Traditionally, **straw** has been used for animal bedding, animal nutrition and improving soil fertility via introducing organic matter in the soil as feed for soil organisms and as a component for soil fertilization. The amount of straw, needed to ensure these functions, depends on specific soil conditions. To some extent, there can be a surplus of straw on the field, that is available without affecting long-term soil fertility. In order to depict the influence of straw harvest to the overall LCI results, scenarios for 1%, 50% and 100% straw harvest were included. It is assumed harvested straw that is used outside the agronomic system should carry environmental burden. On the contrary, straw remaining on the field fulfills functions to the soil-fertility and therefore, no environmental burden is allocated to it.

Comprehensive Life Cycle Inventories (LCI) are built for both, the current LCA modeling practice and previously described approaches. LCI results were calculated and compared to each other. The comparison is focused on the following aspects:

- Different allocation approaches: Cereal Unit allocation versus established allocation approaches (energy allocation, mass allocation and economic allocation)
- Burden sharing between straw and grain: quantification of straw harvest effects for 1% straw harvest versus 50 % straw harvest and 100% straw harvest
- Modeling crops in 1-year system boundaries (no crop rotation) versus the same crop in various crop rotations (*a, b, c* and *d*)
- Environmental benefits of nitrogen-fixing legumes: Crop rotation *d* with legume versus rotation *c* without legume.

3. Results

Applying different allocation approaches leads to deviating LCI-results for all crops and all straw harvest scenarios. Relative differences are presented in Figure 2; LCI-results using the Cereal Unit allocation serve as benchmark. As allocation is accomplished regardless of specific inputs, the relative differences between various are the same.

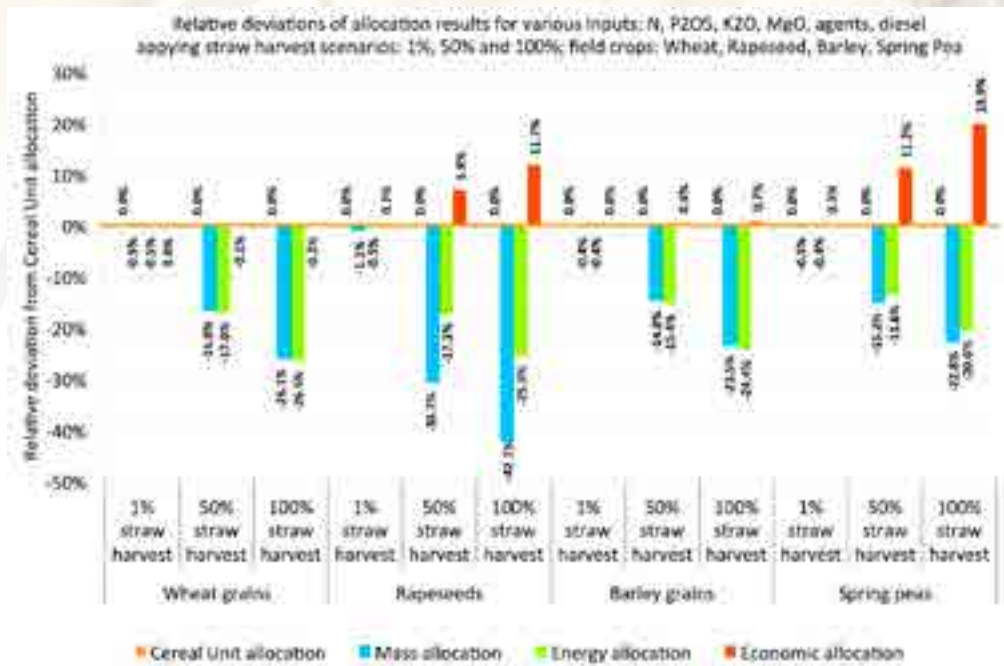


Figure 2: Relative deviations of LCI results using different allocation approaches, compared to Cereal Unit allocation (as benchmark); for wheat grains, rapeseeds, barley grains and spring peas and in each case for the straw harvest scenarios 1%, 50% and 100%

Differences between allocation approaches can be observed. Applying mass allocation and energy allocation leads to lower environmental burdens for all crops in all straw harvest scenarios. For larger shares of straw harvest, bigger differences were observed. For 1% straw harvest, differences in allocation results are less than -1%. For the 50% straw harvest scenario, differences vary between at least -14.8% (Barley grain, 50% straw harvest, mass allocation) and -42.1% (Rapeseeds, 100%, mass allocation). Economic allocation leads to higher environmental burdens for rapeseeds and spring peas. For 50%- and 100%-straw harvest scenario, the economic allocation shows 6.8% to 19.9% higher environmental burdens. For wheat grains and barley grains differences are below +/- 1% in all straw harvest scenarios. As a consequence the straw harvest share is essential to the environmental burden of the grains / seeds / peas. Table 1 provides an overview on LCI-results for the example of specific nitrogen inputs.

Table 1: Specific nitrogen inputs [kg N / t] for grains and straw from wheat, barley, rapeseed and pea; applying different allocation approaches (mA = mass allocation, enA = energy allocation, ecA = economic allocation, cuA = Cereal Unit allocation) and straw harvest scenarios (1%, 50%, 100%)

Allocation:	mA	enA	ecA	cuA	mA	enA	ecA	cuA	mA	enA	ecA	cuA
Wheat												
	1% straw harvest				50% straw harvest				100% straw harvest			
kg N / t grain	21.9	21.9	22.0	22.0	15.8	15.7	18.9	19.0	12.3	12.2	16.6	16.6
kg N / t straw	21.9	22.2	9.2	9.1	15.8	15.9	7.9	7.8	12.3	12.4	6.9	6.9
Barley												
	1% straw harvest				50% straw harvest				100% straw harvest			
kg N / t grain	19.9	19.9	19.9	19.9	14.8	14.7	17.5	17.4	11.8	11.6	15.5	15.4
kg N / t straw	19.9	20.4	8.3	8.6	14.8	15.1	7.3	7.5	11.8	12.0	6.5	15.4
Rapeseed												
	1% straw harvest				50% straw harvest				100% straw harvest			
kg N / t grain	44.6	44.9	45.2	45.1	24.5	29.3	37.9	35.4	16.8	21.7	32.5	29.1
kg N / t straw	44.6	29.0	10.6	14.9	24.5	18.9	8.9	11.7	16.8	14.0	7.6	9.6
Pea												
	1% straw harvest				50% straw harvest				100% straw harvest			
kg N / t grain	-6.4	-6.4	-6.5	-6.5	-4.3	-4.4	-5.7	-5.1	-3.3	-3.3	-5.0	-4.2
kg N / t straw	-6.4	-6.1	-1.9	-3.5	-4.3	-4.2	-1.6	-2.8	-3.3	-3.2	-1.5	-2.3

The example of wheat was chosen to present deviations in LCI-results for modeling crops in 1-year system boundaries versus crop rotations *a*, *b*, *c* and *d*. Figure 3 presents a comparison of specific inputs, needed to grow wheat within crop rotations *a*, *b*, *c* and *d* versus modeling in 1-year system boundary and thus without considering crop rotation effects. Each line within the net-diagram represents the specific input for one ton of wheat grain or wheat straw. This illustration allows comparing differences for each individual specific input.

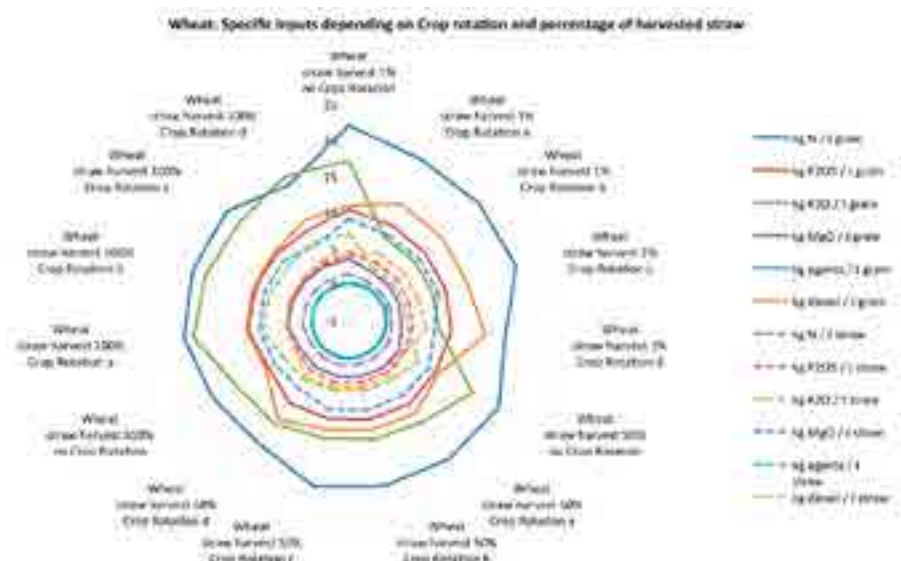


Figure 3: Specific inputs for wheat grain (continuous lines) and wheat straw (dashed lines): influence of Crop rotation and percentage of straw harvest

Specific N inputs for the production of 1 t wheat grains range from 22.03 kg N/t wheat grains (no crop rotation and 1% straw harvest scenario) to 16.70 kg N/t wheat grain (in crop rotation *d* and 1% straw harvest scenario), in relative numbers 24.2% smaller. Specific N inputs for the production of 1 t wheat straw range from 9.11 kg N/t wheat grain (no crop rotation and 1% straw harvest scenario) to 6.91 kg N/t wheat straw (in crop rotation *d* and 1% straw harvest scenario), in relative numbers 24.1% smaller.

Specific K₂O inputs for the production of 1 ton wheat grains range from 17.14 kg K₂O/t wheat grains (no crop rotation in 1 % straw harvest scenario) to 7.23 kg K₂O/t wheat grain (1 % straw harvest scenario; in both Crop rotations *b* and *c*), in relative numbers 57.8% smaller. Specific K₂O inputs for the production of 1 ton wheat straw range from 7.09 kg K₂O/t wheat straw (no crop rotation in 1% straw harvest scenario) to 2.99 kg K₂O/t wheat straw (1% straw harvest scenario; in both crop rotations *b* and *c*), in relative numbers 57.8% smaller.

Explaining all differences for all crops and all nutrients in writing would be too extensive for this paper. Therefore, in Figure 4 a graphical representation of the specific inputs (N, P₂O₅, K₂O, MgO, agents and diesel) for barley, rapeseed and spring pea for all crop rotation options and straw harvest scenarios is provided.



Figure 4: Specific inputs (N, P₂O₅, K₂O, MgO, agents and diesel) for barley grain and barley straw (left); rapeseeds and rapeseed straw (center); spring peas and pea straw (right): influence of Crop rotation and percentage of straw

As N-fertilization has biggest impact to climate change, differences are explained for N inputs only. For barley, the specific N input ranges from 19.94 kg N/t barley grains (no crop rotation, 1% straw harvest) to 16.06 kg N/t barley grains (in crop rotation *d*, 1% straw harvest scenario), in relative numbers 19.5 % smaller. For rapeseed, the specific N input ranges from 45.15 kg N/t rapeseeds (no crop rotation, 1% straw harvest) to 20.88 kg N/t rapeseeds (in crop rotation *d*, 1% straw harvest scenario), in relative numbers 53.8% smaller. For spring pea, the specific N input ranges from -6.46 kg N/t spring peas (no crop rotation, 1% straw harvest) to 12.69 kg N/t spring peas (in crop rotation *d*, 1% straw harvest scenario), in relative numbers 296 % higher.

Figure 5 presents the specific inputs for rapeseed, wheat and barley, grown both, in a rotation without spring pea in crop rotation *c* and with spring pea in crop rotation *d*. Since pea is a nitrogen-fixing legume, this allows comparing the effect of including legumes in crop rotations towards the environmental performance of the other crops in the rotation.



Figure 5: Crop rotation effect of spring pea (nitrogen fixing legume) on LCI-results of rapeseeds, wheat grains and barley grains; crop rotation *c* without pea (dark bars) versus crop rotation *d* with pea (light bars)

The results indicate that specific nitrogen inputs for rapeseeds, wheat grains and barley grains are 13.5% lower, when grown in a rotation with a legume (crop rotation *d*). An opposite effect was observed for the specific inputs of phosphorus pentoxide, potassium oxide and diesel. Whereas specific nitrogen inputs are 13.5% lower in crop rotation *d*, inversely, the specific diesel consumption is 15.1% higher for crops in rotation *d*.

4. Discussion

For agricultural LCAs the selection of an appropriate allocation approach is difficult, since many products and co-products occur along the agricultural supply chain of a large number of products. Alignment of allocation procedures between different product groups is far away from trivial, because producers tend to optimize their LCA towards better environmental performance for their products. The Cereal Unit allocation was introduced as a biophysical and sector-specific allocation approach for all agricultural supply chains. Comparison between mass, energy, economic and Cereal Unit allocation was performed. In many cases, economic allocation and Cereal Unit allocation shows comparable LCI-results. Reason might be the production potential that is expressed on the one hand by the animal nutritional value (Cereal Unit allocation) and on the other hand by the market price (economic allocation).

In the 1% straw-harvest scenario, the specific inputs for all main products of the crops are almost identical in all allocation approaches. The reason is, that almost total amount of inputs is allocated to the main product (grains / seeds / peas). An allocation to the co-product straw only takes place, to the amount of straw that is being harvested. Straw remaining on the field fulfills functions to the soil fertility. The higher the amounts of harvested straw are, the lower is the specific input for the grain.

Significantly different results for the specific nitrogen inputs – up to 42% – were found for the 50%- and 100%-straw harvest-scenarios between: first, the Cereal Unit allocation results and second, the mass and energy allocation results. Reason behind these differences is the fact, that mass allocation and energy allocation do neither take the nutritional nor the economic value of co-products into account. The Cereal Unit allocation allows differentiating nutritional properties between 1 ton of straw and 1 ton of grains, whereas the mass allocation allocates considers the weight of products only. The energy allocation has a similar limitation in using the lower heating value. Economic allocation is

suitable to depict the current market price, but is subject to fluctuations and might be flawed by subsidies.

For rapeseed, reduced specific inputs were found for N, P₂O₅, K₂O, MgO and diesel in every crop rotation compared to no crop rotation (modeling with 1-year system boundary). Deviations can be explained by the fact that rapeseed straw is relatively rich in nutrients. Especially, the concentrations of nitrogen and potassium oxide in rapeseed straw and straw/corn-ratio are very high for rapeseed compared to other crops – leading to greater amounts of nutrients, remaining in crop residues on the field (46.2 kg N/ha rapeseed vs. 23.6 kg N/ha barley and 155.0 kg K₂O/ha rapeseed vs. 80.1 kg K₂O/ha barley).

High concentrations of nitrogen and potassium oxide in rapeseed straw are (amongst others like improved soil structure) relevant aspects for increased yield of succeeding crops. These effects are covered by the tested approach of including the crop rotations. The other way around, these effects are not considered, if rapeseed is modeled in a 1-year system boundary (no crop rotation). Within these assessments, the amounts of nitrogen fertilizer and potassium fertilizer are being completely allocated to the rapeseed plant, even though a relevant amount remains within the straw for the succeeding crop on the field. Similarly, this is valid for other crops as well. Comparisons between crops should consider these aspects. Otherwise, we see a methodological discrimination of crops that leave behind high amounts of nutrients on the field via their residues.

Spring pea has lower specific inputs of P₂O₅, K₂O, MgO, agents and diesel, when grown in rotations, compared to modeling in 1-year system boundary. Since legumes perform nitrogen fixation the specific nitrogen input is negative, if modeled in a 1-year system boundary. When considering spring pea as part of a crop rotation, a certain amount of the sum of nitrogen fertilizer of the entire crop rotation is attributed to it. At first glance, it may sound incorrect that legumes carry any burden of nitrogen fertilization. But there should be no doubts that legume will consume as well nitrogen from previous crops. The same concept lowers spring peas' specific inputs of P₂O₅, K₂O and MgO. The significant reduction of specific diesel consumption can be explained via the performance principle: all inputs are allocated between the crop rotations inputs using the nutritional value of its outputs. Compared to the other crops in the rotation, spring pea has relatively low yields. Therefore, a certain part of the P₂O₅, K₂O, MgO, agents and diesel is allocated to the products having greater share of the production potential of the entire rotation.

Comparison of the specific inputs for wheat, barley and rapeseed in the crop rotation *c* (without legume) versus crop rotation *d* (with legume) shows a clear reduction of specific nitrogen inputs. This has two reasons. First, the allocation of a certain part of the nitrogen burden to pea that consumes certain amount of nitrogen from the soil and second, pea performs nitrogen fixation, which reduces the overall amount of nitrogen input of the entire rotation. This helps to reduce the specific nitrogen inputs. A small improvement of MgO inputs could be observed as well.

For the inputs diesel, K₂O, P₂O₅ and agents, higher specific inputs were observed. This is due to the relatively smaller production potential of the pea plant – in terms of yield in combination with the nutritional value of the products. For instance regarding the specific diesel inputs, the diesel consumption is comparable for pea, but the yield of pea is with 2.09 t CU peas / ha considerable lower than for barley 6.73 t CU barley / ha. Since the yields (expressed in Cereal Units = animal nutritional value) are higher for wheat, barley and rapeseeds, they carry more environmental burden than pea.

This performance principle helps to assess entire crop rotations, whilst considering the production function of agricultural systems.

When comparing the specific inputs for the crops, modeled in 1-year system boundaries (no rotation) with the same crops, grown in one of the rotations, lower values can be observed. These reductions might occur for all crops in the rotation. This could be misinterpreted in modeling mistake, but one should bear in mind that modeling of the entire rotation leads to consideration of nutrient shifts between the crop rotation elements via the crop residues on the field. Unless explicit measures

taken, such consideration of nutrient shifts does not take place in LCA modeling without considering crop rotations (Brankatschk and Finkbeiner 2015).

Farmers perform soil tests, nutrition balances and crop rotation planning to achieve optimal nutrient supply for each crop. In this context, stoichiometric calculations are performed and even required for efficient use nutrients and to be in line with good agricultural practice. Since nutrients represent the biggest share of inputs and cause biggest environmental impacts, it can hardly be explained towards farmers, why nutrient shifts between crops shall not be adequately considered in Life Cycle Assessments.

Distinguishing between crops, grown in long crop rotations or 'monoculture' seems to make sense against the background of deviating results, presented in this work. Typically, current LCA results are not accompanied by information about agricultural conditions in which the considered crop was grown. Presented results clearly show differences in LCI results and consequently in environmental performance of crops, whether modeled as part of a rotation or not. In order to make LCA capable of supporting agricultural planning and drawing well-founded decisions towards more sustainable agricultural practices, LCA should be able to measure these differences. Otherwise, environmental performance of crops, grown in versatile rotations could be applied to crops, grown in monoculture. The same might occur in reverse. Environmental advantages and improvements of agricultural practices or crop rotation planning would remain unverifiable by LCA. Identifying climate-smart crop rotations would be hardly feasible. In order to keep in pace with future trends in agriculture and agricultural policies, the consideration of crop rotation effects in LCA seems to be appropriate.

5. Conclusions

In order to meet future needs of food provision and to perform climate action, farmers need advice in identifying appropriate farming practices and crop rotations. Robust and reliably tools are needed to meet this demand and to verify farmers' efforts towards better sustainability in food supply chains.

The outcome of this work shows significant differences in the LCI-results, depending on whether crop rotation effects are being included or not. Modeling crops in 1-year system boundaries apparently leads to different LCA results and does not seem to sufficiently depict agricultural practice. Recent LCA modeling methods are not able to distinguish on product level the environmental burdens of crops, grown in different crop rotations. Furthermore, relevance of straw harvest scenarios towards LCI-results is presented. The need for a better representation of agricultural processes in LCA is quantified.

Considering crop rotation effects, crop residue functions and introducing an agriculture-specific allocation approach might help bringing LCA closer to agricultural reality. The tested approaches may enable LCA to measure environmental benefits of future farming practices and assist the development of environmentally sound crop rotation-strategies. In addition, societal and political requirements will raise the need to assess different types of crop rotation systems.

Within decision-making towards more sustainable processes, farmers will reasonably insist on reliable and robust recommendations that adequately represent underlying agricultural processes and different options. Current modeling practice does not seem to fulfill these criteria; thus, farmers would prefer agriculture-specific LCI-results.

A better representation of different farming practices and crop rotation options is an imperative for future LCA work. Further development of agriculture and recommendations towards more sustainable agriculture will contain changes in agricultural practices and agricultural planning. Reliable tools for assessing different agricultural practices will become necessary. As long as LCA is able to make environmental differences between farming practices visible, it will be a reliable partner towards sustainable agriculture.

Tested approach of modeling crop rotations offers a potential solution towards crop rotation modeling and thus supports farmers in their agricultural planning towards more sustainable agricultural practices. Without inclusion of crop rotation effects, the environmental advantages and the improvements of agricultural practices, enabled through different crop rotations, would remain undetected. In order to keep pace with future trends in agriculture and environmental policies, it is required to comprehensively depict agricultural systems and to consider crop rotations in LCA.

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256. Does collaboration between farms in favoured and less-favoured regions improve environmental performance of dairy production?

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ABSTRACT

Agricultural production in less-favoured areas, such as that in the mountains, is often limited to grassland-based production systems. Farming in such regions is mostly extensive. Thus, environmental impacts per area are rather low, and in many cases the semi-natural farming preserves a high nature value. However, due to the low productivity, environmental impacts per product unit are often higher than those of a similar product from farms in favoured areas. This is also the case for the Swiss mountains. The aim of this study was to test if a division of labour between farms in the Swiss mountains and the more-favoured lowlands reduces environmental impacts of milk. In such a contract-rearing system, dairy cows are kept by lowland farms, and the less-demanding young stock is reared by mountain farms (collaborative system). In a life cycle assessment, this system was compared to a non-collaborative system in which both lowland and mountain dairy farms kept dairy cows and young stock for restocking on the same farm. The functional unit for comparison was one kg of milk. The collaborative system had lower environmental impacts, with significant differences for the impact categories terrestrial ecotoxicity, non-renewable energy demand and use of phosphorus from mineral sources. There were two reasons for the better performance of the collaborative system: (1) each region focussed on the activity for which it had a comparative environmental advantage, and (2) the specialisation induced by collaboration reduced management complexity. We conclude that division of labour between favoured and less-favoured regions can help keeping less-favoured areas in production, while reducing environmental impacts of agricultural products.

Keywords: contract rearing, comparative advantage, mountain farming

1. Introduction

Less-favoured areas face natural constraints that impede competitive agricultural production. Examples of such areas are mountains, areas with poor soil quality, or remote areas with difficult market access. Traditionally, such areas are managed extensively and likewise considered to be of special importance due to their high nature value and the provision of an attractive landscape. As a consequence of the extensive farming practices, environmental impacts per unit of land are lower than those of more-favoured regions (Rudow, 2014). However, due to the lower productivity of the land, the environmental impacts per food product unit are usually larger if production happens on less-favoured areas than if it happens on more-favoured areas (Hörtenhuber et al., 2010; Ripoll-Bosch et al., 2013). Thus, there is conflict between environmentally sound provision of agricultural products and conservation of farming areas with high nature value in less-favoured regions.

To conserve important habitats and attractive landscapes in less-favoured regions and environmentally optimise agricultural production, we propose to shift the focus from absolute environmental advantages to comparable environmental advantages. An absolute advantage would exist if the less-favoured regions could produce an agricultural product with lower environmental impacts per product unit than favoured regions. It is rather unlikely to identify such products due to the lower productivity of less-favoured regions. Comparative environmental advantage, on the other hand, exists when a product can be produced at lower environmental opportunity costs. When a farmer decides to produce one agricultural product instead of another, environmental opportunity costs correspond to the amount of the other product the farmer could have produced with the same environmental impact. In a previous analysis with modelled average dairy farms in Switzerland, lowland farms were found to have a comparative environmental advantage for keeping dairy cows, while mountain farms had one for keeping the young stock needed to restock the dairy herd (Marton et al., 2016). A contract-rearing system between the two regions thus has the potential to reduce the overall environmental impacts of dairy production. In the present study, we performed life cycle assessment (LCA) of commercial farms from mountain and lowland regions that were either participating in the contract-rearing scheme or not. The goal of this study was to test whether the

theoretical advantages of the contract-rearing system can be translated into real benefits for the environment.

2. Methods

The goal of the LCA was to compare collaborative and non-collaborative dairy production. For the present evaluation, collaborative dairy production was defined as the situation in which only the lowland farm kept dairy cows, while the mountain farm was specialised in rearing young stock for the lowland farms. Non-collaborative dairy production was defined as the situation in which both lowland and mountain farms produced milk and reared the young stock needed for restocking on the same farm. For the comparison, we analysed environmental impacts of one kg of fat and protein corrected milk. The system boundaries are displayed in Figure 1. Allocation rules of the LCA guidelines of the International Dairy Federation (IDF, 2010) were applied, i.e. physical-causality allocation between milk and meat, and system expansion through avoided burden for manure applied outside of the dairy system. Data for 2012 from four collaborating and four non-collaborating farms from both regions, the mountains and lowlands, were collected. The analysed farms were selected randomly within each region, and therefore the farms were not actually collaborating with the respective farms from the sample, but with other farms that were not assessed. To simulate possible collaborations, each collaborating lowland farm was thus virtually combined with each collaborating mountain farm, resulting in a set of 16 possible collaborative combinations. Farms were combined based on the lowland farm's annual demand for heifers and the mountain farm's annual heifer production. The lowland farm and a proportion of the mountain farm corresponding to the ratio 'heifers demanded : heifers produced' were combined to form a collaborative system with a closed restocking cycle. Non-collaborative farms from the two regions were combined so that the land-use ratio within the two regions corresponded to that within the collaborative production system.

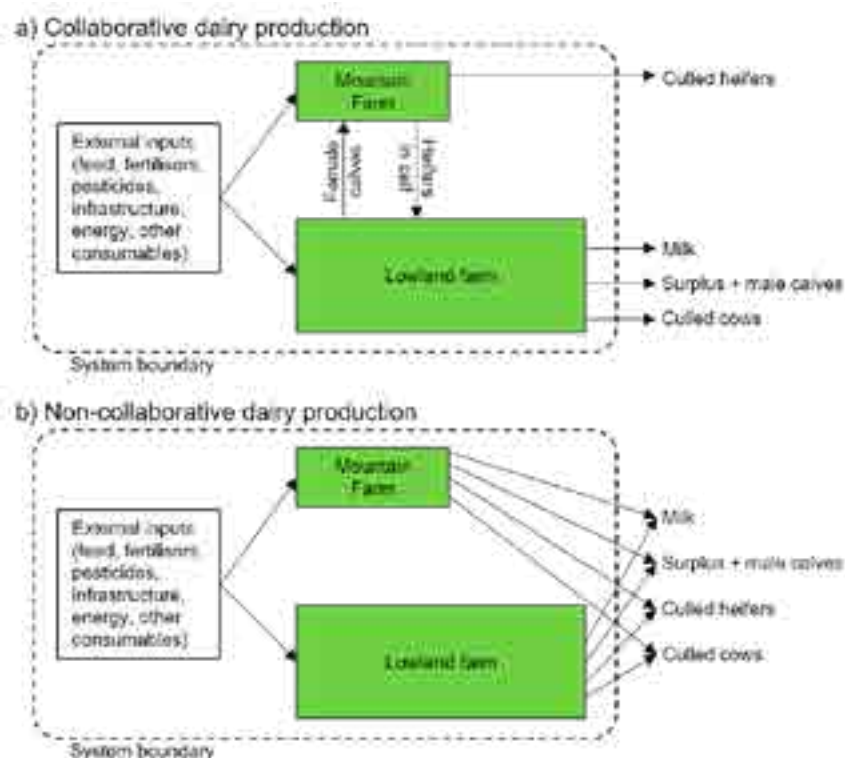


Figure 4: System boundaries of the two dairy production systems compared.

The life cycle inventory was calculated with tools developed for the EU FP7 project CANTOGETHER (Teuscher et al., 2014), and life cycle impact assessment was performed with

SimaPro 7.3. For the present study, we focussed on five impact categories: terrestrial ecotoxicity (EcoTox, CML 2001, Guinée et al., 2001), marine water eutrophication potential due to nitrogen input (EutN, EDIP2003, Hauschild and Potting, 2005), global warming potential over 100 years (GWP, IPCC 2013, Myhre et al., 2013), non-renewable cumulative energy demand from fossil and nuclear sources (nrCED, Frischknecht et al., 2007a), and phosphorus use from mineral sources based on elementary flows from ecoinvent® (ResP, Frischknecht et al., 2007b).

3. Results

For all environmental impact categories studied, the median of the impact of milk produced in the collaborative system was lower than that from milk from the non-collaborative system (Figure 2). This difference was significant for EcoTox, nrCED and ResP, for which the median of the impact with collaboration was lower by 62%, 19% and 55% than the median of the impact with non-collaboration. One main reason for the lower impact was the use of less concentrates on collaborating farms. Collaborating lowland farms used less than half the amount of concentrates per cow than non-collaborating lowland farms, although the milk yield was similar. Young stock were generally fed with low amounts concentrates, regardless of the production system. Collaborating farms, both in lowlands and mountains, also used lower amounts of fossil energy carriers per hectare, which contributed to their comparably low nrCED. For EcoTox and ResP, variability within the collaborative system was rather high: the impact of the farm combination with the highest impact was 8.4 and 3.7 times larger, respectively, than that of the farm combination with the lowest impact. The difference in EcoTox was mainly the result of a feeding strategy, in which feeding of potatoes caused a higher impact. Potatoes were only fed on one of the collaborating lowland farms. The difference in ResP was the result of the fertilisation strategy of one single lowland farm that applied mineral P fertiliser on grassland, while none of the other collaborating or non-collaborating lowland farms used mineral phosphorus fertilisers on grassland.

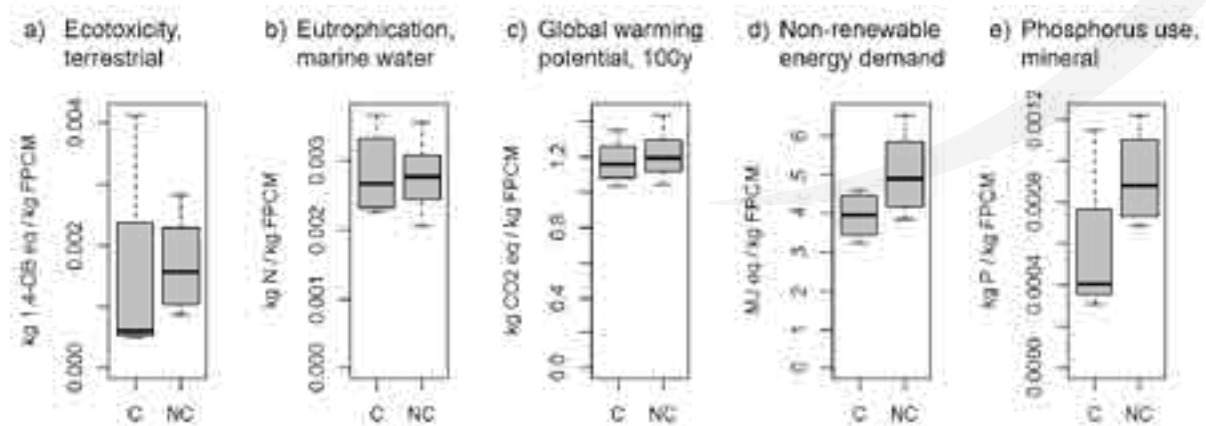


Figure 5: Boxplots comparing the environmental impact per kg of fat and protein corrected milk (FPCM) of the collaborative (C, n=16) and non-collaborative (NC, n=16) dairy production systems for the five impact categories studied.

4. Discussion

Results of the present study confirmed those of a previous study with simulated farms (Marton et al., 2016). Collaboration allowed lowland and mountain farmers to focus on activities in which they had a comparative environmental advantage. Differences between collaborative and non-collaborative systems were more pronounced than those between simulated farms. We assume that, besides the reduction achieved through comparative environmental advantages, a reduced management complexity on collaborating farms has contributed to the advantage of the collaborative system, as both mountain and lowland farms could focus on fewer farming activities. Reduction in management

complexity is often linked to gains in efficiency (Kingwell, 2011). In the present sample, the collaborating lowland farms managed to achieve the same milk yields as the non-collaborating lowland farms, although they used fewer external inputs such as concentrates or energy carriers. The forage types used on collaborating and non-collaborating lowland farms, mainly grass and maize, did not differ much. As the farms were located in the same region (canton of Thurgau), there were no apparent external reasons that explained why collaborating farmers were able to produce more milk based on their forage than the non-collaborating farms. Consequently, reduced management complexity might be the reason for the higher efficiency observed on the collaborating farms.

Like other LCA studies of commercial farms, variability among farms within each group was high. In some cases, variability was influenced by a single issue, like application of triple superphosphate on grassland, which led to a high ResP, or feeding of potatoes, which led to much higher EcoTox. On one hand, such information could be used to further optimise the farming activity. On the other hand, it also illustrates the sensitivity of environmental impacts to special events or unusual practices. Application of mineral phosphorus to grassland is rather uncommon, as the phosphorus requirements of plants are usually covered by manure applications. In this case, triple superphosphate was applied in autumn to a temporary grassland which was ploughed and planted with sugar beets the following spring. Since phosphorus has low mobility in soils and high fixation to the soil matrix (Shen et al., 2011), it is stored in soils for a longer period; thus, it may have been taken up by the sugar beets even though it was applied to grassland. Accordingly, it should have been at least partially attributed to the sugar beets. This would have reduced ResP of dairy production because sugar beets are sold as a cash crop, and not used within the dairy production.

A different situation was observed with the potatoes fed to dairy cows and their effect on EcoTox. Potato production indeed requires high quantities of pesticides. However, potatoes are commonly cultivated for food and not for feed production. Potatoes used as feed are either waste from the food-processing industry or those that did not fulfil quality criteria for human consumption and thus never entered the food chain (Willersinn et al., 2015). From an environmental viewpoint, using these potatoes as feed is reasonable; however, in the present study, the farm that used them was penalised by a higher EcoTox impact. The IDF guidelines do not cover this specific problem. For feed that is a co-product of food processing (e.g. soya bean meal vs. oil), economic allocation is recommended. But for potatoes that never entered the food chain due to quality issues, such an allocation is not feasible. Thus, the full environmental impact of potato production was attributed to the dairy system. From an attributional point of view, this is correct, but it would set incorrect incentives demotivating farmers from using such products.

These two examples show that there is no one-size-fits-all solution in LCA and emphasises the importance of deep analysis of individual results to draw the correct conclusions. In the present study, these special cases influenced the results of individual farms, which led to outliers in the group of collaborating farms, but for both impact categories concerned, advantages of the other collaborating farms over non-collaborating farms were still sufficiently high to generate significant advantages of collaborative over non-collaborative dairy production systems. Different allocation or attribution of the triple superphosphate applied to grassland and of the feed potatoes would thus not have changed the final conclusion.

5. Conclusions

For the example of dairy farming in the Swiss lowlands and mountains, we found that collaboration can reduce environmental impacts of agricultural production. Division of labour in addition can help sustaining agricultural areas in the less-favoured mountains. Further research is needed to test whether division of labour between favoured and less-favoured areas based on comparative environmental advantages can be applied to other agricultural production systems or other geographic regions.

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80. LCI of food products sectors: from field data collection to the big picture

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ABSTRACT

This article deals with the construction of an LCI at the scale of agri-food sectors. It develops the methodology used to model the inventory. The methodology has been applied to two French agri-food sectors: the PDO Beaujolais and Burgundia wine sector and the PGI South West duck foie gras sector. The result of this study will allow sectors stakeholders to get an accurate snapshot of the sector's organization by the analysis of the agri-food sector inventory.

Keywords: inventory model, inventory scaling up, sector's environmental assessment, Beaujolais and Burgundia wine sector, foie gras from the French South-West area sector

1. Introduction

The ACYDU project, co-funded by the French National Research Agency (ANR), investigates the environmental, social, economic and territorial impacts of agri-food sectors. Three sectors have been investigated, in relation with emblematic French products under official quality labels: the foie gras from the South-West area (Protected Geographical Indication) sector, the Beaujolais and Burgundia wine (Protected Designation of Origin) sector and the Comté cheese (Protected Designation of Origin) sector. Various methodologies, either qualitative or quantitative, were applied to address the four types of impacts under study. Environmental impacts were assessed through attributional LCA. The specificity of the use of LCA in this specific context is the look at the agri-food sectors with a broad perspective, i.e. encompassing all steps linked with the products themselves (from agricultural stages to final consumption) but also complementary activities that belong to the sector (research, promotion, etc.). The ACYDU project outcomes are mainly directed at agri-food sectors decision makers in order to help them to identify strategic areas for improvement.

The specific objectives of this research project require to propose methodological adjustments to the product LCA and Organizational Life Cycle Assessment guidelines. Previous methodologies dealing with combining macro-economic sectors data (Input-Output transactions tables tracking purchase flows between sectors) and LCA methodologies have been developed (Lave et al., 1995; Hendrickson et al., 2005). More recently, Schmidt et al. (2007) used the multi-regional environmentally extended supply and use / Input-Output database Exiobase2 to assess the environmental impact of the global food consumption. These approaches referred, as Economic Input-Output LCA (EIO-LCA) or hybrid LCA, rely on aggregate sector-level average data that may not be representative of a specific product system. Commodity sectors in national input-output tables used in EIO-LCA are broad aggregates that cover a large number of products and do not take into account technological product specificities. As these methods did not fit the study's objectives, an ad hoc methodological approach, presented in this paper, has been developed. Challenges related to data availability and representativeness have had to be faced. Practical application is shown based on the work that has been carried out for 2 out of the 3 case studies: the PDO Beaujolais and Burgundia wine sector and the PGI South West duck foie gras sector. A critical review of this study being under process, this present paper does not show the impact assessment for the agri-food sector under study.

2. Methods

A distinctive feature of this study is the use of agri-food sectors boundaries as boundaries of the systems considered for the LCA. These boundaries have been established building on the methodological work conducted in another work package of the ACYDU project that aimed at (i) identifying the life cycle steps and (ii) defining a decision tree to be able to identify which stakeholders belong to the sector (Lempereur, 2015; Assogba, 2015).

The steps considered within the system boundary are: (i) production, transport and processing of raw materials and other inputs required for the production, transport, delivery and consumption of final products; (ii) management of waste generated over the different life cycle steps; (iii) office and travelling activities of stakeholders, including the ones belonging to the “close environment” (i.e. considered within the sector even though not involved in the production chain, such as dedicated research centers, federations, etc.). The construction of buildings and the manufacture of equipment have been excluded from the assessment as previous LCA studies have highlighted that their contribution in terms of potential environmental impacts is negligible for mass-market products.

To build the LCI, a methodology deriving from both product and organizational LCA was developed and applied. According to the ISO 14044 (ISO, 2006), the functional unit is the quantified performance of a product system used as a reference unit. As this terminology does not fit with the life cycle assessment of an organization, the ISO 10072 (ISO, 2014) refers to *product portfolio* as the quantified expression of the studied system. It was chosen to use the wording *quantified expression of the studied system*. For both studied agri-food sectors, the quantified expressions of the studied systems correspond to their global activities over a 1 year period. For the PDO Beaujolais and Burgundia wine, the period considered is the wine year 2011-2012. The reference flow linked to the quantified expression of the studied system is the grape production, the wine making, the packaging, the distribution and the consumption of 237.1Ml of wine. For the PGI foie gras from French South-West sector, the period considered is the 2014 civil year, corresponding to the production of 2'645 t of raw foie gras and 4'988 t of processed foie gras and the following distribution and consumption steps (PALSO, 2014).

The LCI was built using a bottom-up approach in the sense that inventories were first built for sub-systems and then scaled-up to complete the global LCI at the agri-food sector level.

Building-up sub-inventories for each life cycle step

Each agri-food sector is composed of several organization activities. These companies can usually be classified in “organization’s types” regarding their activities (for example farmers, feed manufacturers, retailers, and so on), sometimes spread over several life cycle steps (Figure 1). To deal with organizations which activities cover different life cycle steps, it is necessary to cut the organization system in different subsystems and to identify several intermediate reference flows, corresponding to the outputs of the different life cycle steps.

Primary data have been collected from the different types of organizations belonging to the defined sub-systems through questionnaires. Using annual activity data provided by participating organizations, average data have been established (i.e. average energy consumption, water consumption, amount of waste produced...). These data have been expressed per unit of the output *intermediate reference flow* of the subsystem under study (e.g: number of fattened ducks after the rearing and over-feeding step, volume of wine after the wine making step).

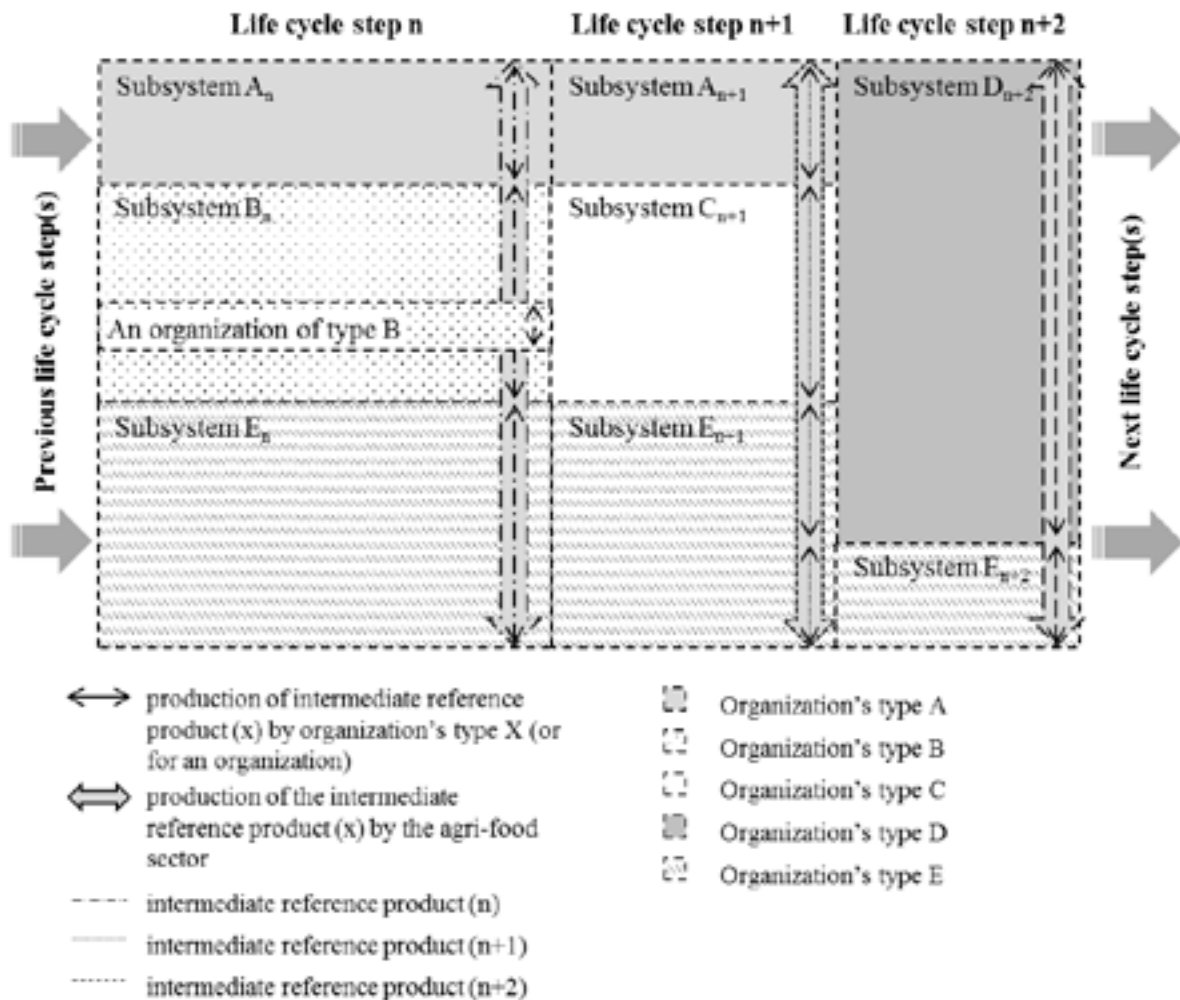


Figure 1: Graphic representation of a theoretical agri-food sector model used to build-up the life cycle inventory at the scale of the studied agri-food sector

Scaling-up the LCI at the agri-food sector level

The intermediate reference flows for the different sub-systems are used to connect the sub-systems together and build-up the inventory at the scale of the studied agri-food sector. To be able to do this scale-up, it is necessary to quantify the different intermediate reference flows for the whole studied agri-food sectors (thick vertical arrows on Figure 1) and for each organization's type (thin vertical arrows on Figure 1). In practical terms, this consists in establishing a mass balance featuring inputs and outputs flows at the sector level for the reference period. These data are then used to put together the global sector model. Primary data sources used for this task are statistics from trade associations and Customs trade data to be the most accurate. Assumptions based on sectors specialists were made to deal with unavoidable data gaps. To obtain inventories of subsystems at the sectors scale, inventories expressed per unit of intermediate reference flow (e.g. 1 fattened duck, 1 hl of wine) are multiplied by the intermediate reference flow productions of the corresponding organization's type (e.g. annual number of fattened ducks for the sector, annual volume of wine as an output of the wine making step for the sector).

3. Results

The outcome is an LCI adapted to the specific scope of the agri-food sectors under study. Figures 2 and 3 below present the life cycle steps studied for each sector, and the intermediate reference flows for each step that are used to scale-up the LCI as explained above.

Each LCI is composed of six steps: five production steps (agriculture, transformation, packaging, distribution and consumption) and one transversal step referred as “close environment” composed by the activities of the organizations that belong to the sector but are not involved in the production chain.

The LCI of the PDO Beaujolais and Burgundia wine sector begins with the production of 312 900 tonnes of wine grape (94% conventional agriculture and 6% organic) that are then transformed during the wine-making and aging step into 2 371 000 hl of wines. Different types of organizations are involved in this step: individual cellars (66% of the volume), cooperative cellars (30%) and vintners (4%). By-products (grape pomace and lees) are valorized by distilleries. Wines are then packaged. Packaging is the historical activity of the vintners who package 52% of the wines in the Beaujolais and Burgundia sector. The other 48% are packaged by cellars. Considering a loss rate of 0.3% of wine during the packaging step (CEEV, 2016), 2 363 800 hl of packaged wine are distributed, 62% in France and 38% abroad. 43% of the wines are distributed by retailers, 35% by direct sale and 22% by the cafés-hotels-restaurants. Then, considering a loss rate of 1% during the distribution and 5% during the consumption step (CEEV, 2016), 2 223 236 hl of wines are consumed (cf. Figure 2).

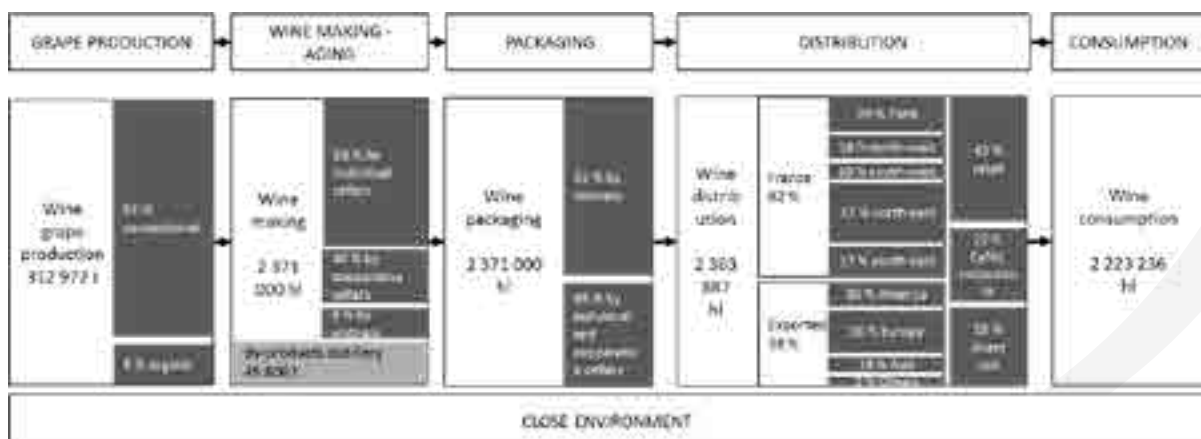


Figure 2: Model of product flows in the PDO Beaujolais and Burgundia wine sector

The LCI of the PGI South West duck foie gras sector starts with the raising of ducklings, followed by the overfeeding step, leading to about 24 616 360 fattened ducks ready for slaughtering. Once the foie gras has been taken out after cutting, raw foie gras is either commercialised raw or transformed into a range of different products. Putting together sector data in order to complete the global sector mass balance pointed out the importance of the product flows that are downgraded as non PGI along the production chain. Indeed, some intermediate products leave the PGI production chain along the process, because of compliance issues regarding the PGI specifications or commercial opportunities. The analysis of available statistics for the PGI South West duck foie gras sector suggests that 8.5% of fattened ducks get out of the PGI production chain before slaughtering. Further downgrading occurs after slaughtering. In the present LCA, the issue has been dealt with by allocating to the PGI foie gras sector all inputs and outputs arising upstream from the downgrading. In the end, 7 633 tonnes of foie gras products (either raw or transformed) are commercialised under the PGI label. In order to take into consideration the potential environmental impacts related to the packaging, it has been necessary to establish the breakdown of the different types of packaging (i.e. glass containers, plastic containers, metal cans...) for the various products. Assumptions were necessary since available sector data did not include such information. Market data for the PGI foie gras sector show that 96% of products are sold in France and 4% exported (PALSO, 2014). A loss rate of 1% over the distribution step has been considered (Figure 3).



Figure 3: Model of main product flows in the PGI South West duck foie gras sector

Both case studies use the same approach to complete the LCI of the different sub-systems for each of the life cycle steps. For the **agricultural** step, LCI from the French AGRIBALYSE database (version 1.3) have been used (Koch et al., 2015; and more recent updates). For the foie gras case study, inventory data for rearing and over-feeding is based on the work on the PGI production system described in Deneufbourg et al., 2016. The **transformation** step (covering wine making and aging on one hand; slaughtering, cutting and transformation on the other hand) considers: transport of raw materials and inputs to the industrial site, energy and water consumption, consumption of ingredients and process inputs, consumption of cleaning products, transport of by-products to the valorization site (and valorization itself for the wine sector), treatment of waste. A lot of effort focused on the collection of primary data for this step. A sample of organizations (17 organizations from wine sector and 14 from foie gras sector) has been selected, based on technological representativeness. The **packaging** step is composed by: upstream transport of transformed products and other inputs, energy and water consumption, consumption of inputs (mainly primary and secondary packaging), consumption of cleaning products, treatment of waste. The **distribution** step includes: transport of packaged products up to the point of sale, storage by the retailer and storage at the point of sale, end of life of the secondary packaging. The **consumption** step is composed of: transport from the sale's outlet, storage at the consumption place, energy, water and inputs consumption for the preparation of the products, end of life of the primary packaging. The "**close environment**" considers impacts linked to business travels and office work (energy and water consumption). The ecoinvent database version 3.1 was used as the main source of background data.

Multifunctionality has been managed in different ways. For the wine case study, no allocation has been used. The sub-sector in charge of wine byproducts valorization (distillation) has been defined as part of the wine sector. System expansion has been used. For the foie gras case study, this question arises in relation with the rearing/overfeeding and slaughtering steps since these steps result in foie gras production but also other meat parts (magret, strips of breast...) and by-products that are valorized in other sectors (feathers, feet, blood...) (Figure 4). Economic allocation has been applied in the reference scenario, based on economic data concerning the sales price at the slaughterhouse after cutting provided by a sample of companies. An allocation factor of 55% has therefore been applied for foie gras.

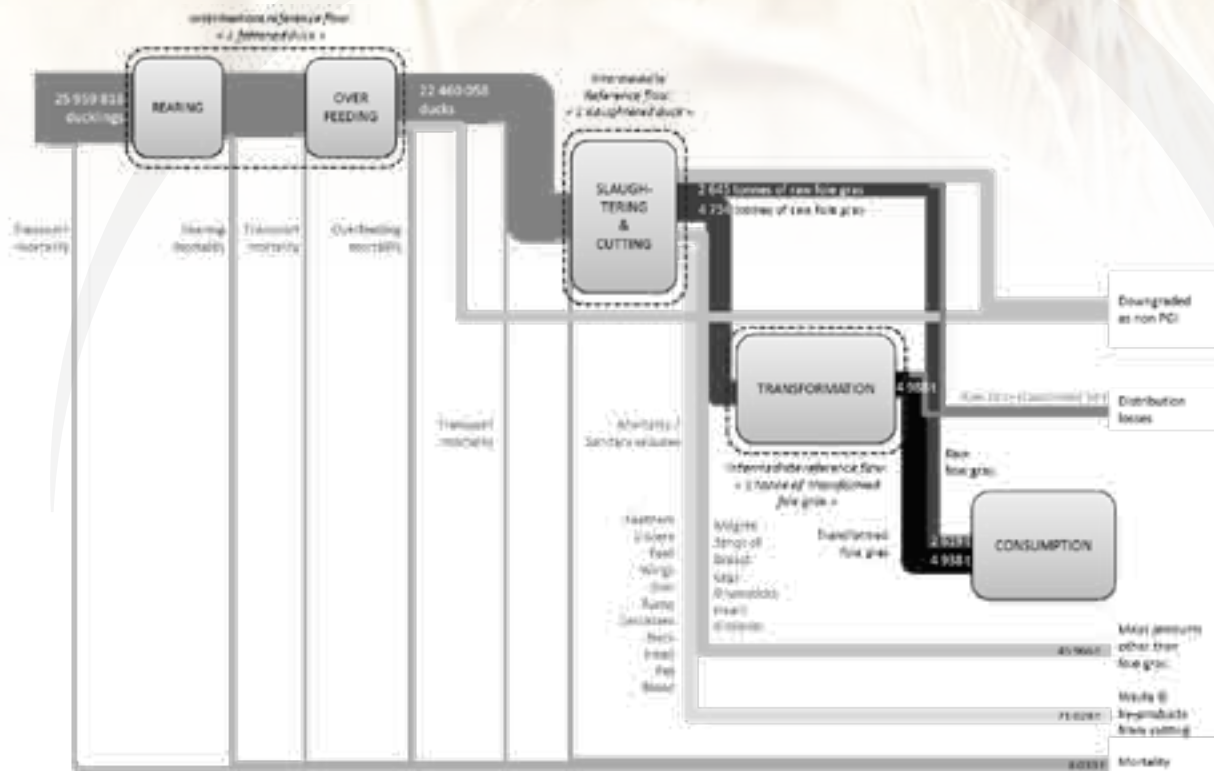


Figure 4: Model of input and output flows in the PGI South West duck foie gras sector featuring byproducts

4. Discussion

Establishing a LCI at a food sector scale starting from primary data collected in individual organizations is a complex task. The quantification of flows, first at the sub-system scale and then at the global scale of the sector for each life cycle step requires substantial research and data compilation work. In addition, despite efforts to ensure representativeness, the data collection for this type of project depends on volunteer organizations that are interested in this exercise. Assessing the representativeness of the different organizations is also a difficult task, especially in agri-food sectors characterized by a large number of small organizations. As a matter of fact, whereas 17 organizations took part to the study for the Beaujolais and Burgundia sector, they only represent 9% of the wine produced. Another issue that had to be faced was that adjustments in the collected primary data were necessary in order to build up the sectors LCI. For instance, in the wine sector, the activities of a given organization often cover several life cycle steps (viticulture, wine making-aging and packaging). Data collected for these organizations (e.g. energy and water consumption) had therefore to be allocated to several steps while only global data were provided.

Several areas for further work can also be identified. For example, additional work could be carried out to explore different ways of addressing downgrading issues for products under registered designation such as PDO or PGI. The methodology that was chosen to deal with the downgrading of PGI foie gras along the production chain implies that the potential environmental impacts of the sector are maximized. Indeed the number of ducks that is taken into account for the upstream steps is as a consequence of the chosen approach well above the number of ducks that would actually be required based on the final quantity of PGI foie gras placed on the market. Allocation issues between products and by-products could also be further studied.

It is also believed that the conducted work has tackled some interesting methodological issues that are not so often investigated in LCA. The presented methodology was developed based on two case studies. It ought to be applied to other food sectors in order to check its repeatability.

5. Conclusions

This work has led to the development of a methodology that can provide a useful framework for LCA practitioners that would like to study the potential environmental impacts of agri-food sectors. However the construction of an LCI that fits the boundaries of an agri-food sector remains a difficult exercise given the extensive data requirements.

The thorough mass balance of input and output product flows at a sector level and the resulting global LCI can be useful to agri-food sectors' decision-makers in order to get an accurate snapshot of the sectors' organization together with an assessment of the potential environmental impacts.

This work can therefore be used to better understand the environmental significance of agri-food sectors and be a basis to explore the consequences of possible changes in the sectors to improve the overall environmental performance.

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ORAL PAPERS

MEAT PRODUCING 2

18. Meat producing (2)

276. A biophysical allocation rule for meat products, based on energy requirements of body-tissue growth

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ABSTRACT

In agricultural life cycle assessment (LCA), the choice of allocation methods to spread out impacts between coproducts is an important issue, as they may induce different conclusions in impact levels. We proposed a biophysical allocation method to dispatch the upstream environmental burdens and the use of raw materials to the body-related coproducts of beef cattle production system at slaughterhouse stage.

The method is designed to build a relationship between coproducts of the beef cattle production system and their associated net energy requirements for body growth. So it doesn't take into account the fate of the different coproducts, but only their building costs (i.e., energy needed for building tissues). A combination of metabolic growth model (Gompertz function) and models of energy cost of tissues was used to estimate the energy requirements for body growth from birth to slaughter age. The allocation factors were calculated based on the energy requirements attributed to build body tissues characterized by their chemical compositions (protein and lipid) with exclusion of waste. Finally, this method was compared with other allocation methods (e.g., physical, economic).

The biophysical allocation reflects a physical and biological relationship between the coproducts as required by ISO standard. It provided a moderate allocation factor for human food due to their chemical characteristics compared to the other physical allocation methods. In addition, the data required is specific to species and less influenced within a predefined system than economic allocation.

This study provides a generic and robust biophysical allocation method to handle the coproducts in beef cattle system. The method can be considered as an original contribution to the international debates on the allocation methods in LCA applied to livestock products, especially among the stakeholders of the meat value chains.

Keywords: Gompertz function, animal co-products, body tissue composition, physiological function

1. Introduction

Allocating the environmental burdens and the use of raw materials among these co-products is an important issue in attributional life cycle assessment (LCA). Several methods exist for doing so, and ISO 14040/14044 (ISO 2006) recommends a hierarchical choice of allocation methods. Ideally, allocation should be avoided by expanding the system to include additional functions of co-products, or by dividing the process into several sub-processes with relative input and output data. When it is not possible to avoid allocation, one should attempt to attribute system inputs and outputs to co-products according to their underlying physical relationships (e.g., mass/energy allocation) under the condition that the co-products have similar characteristics. Only when a physical relationship alone cannot be applied as the base of allocation, should one establish other relationships among co-products (e.g., economic allocation). For agriculture, physical and economic allocations are "classic" methods commonly used in LCA (van der Werf and Nguyen, 2015; ADEME 2010; Cederberg and Stadig 2003). Mass allocation reflects a physical relationship; it sometimes cannot show the causality between animal-related co-products. Protein allocation reflects the function of products from agricultural systems to provide protein to humans or animals (Nguyen et al. 2012); however, it fails to consider multiple characteristics of co-products. Switching among one single characteristic (e.g., protein, lipid) as an allocation indicator may change results, because the functions of co-products vary greatly according to their final uses (e.g., human food or biofuel) (LEAP 2014). Economic allocation, often used in LCA, reflects marketable values of agri-food co-products according to their uses, such as food, feed and biofuel (EPD 2012; PAS 2050 2008). However, it seems insufficiently robust, due to temporal and spatial fluctuations in market prices of co-products, and the values of co-products at production level cannot reflect their values after the transformation (Gac et al. 2012). To overcome the causality and functional problems of the classical physical allocation procedure, a biophysical allocation method was developed based on the energy required to produce co-products (IDF 2010).

Few studies have used biophysical allocation to divide environmental burdens among livestock co-products (Nguyen et al. 2013; Wiedemann et al. 2015). They used allocation factors based on feed energy requirements. These energy requirements result from animal physiological processes (e.g., maintenance, growth, activity, lactation, gestation) to produce milk/wool and live animals in production systems and reflect the underlying biophysical relationship among co-products. However, these studies only considered livestock co-products at the farm gate; none used biophysical allocation for body tissues that emerge from animal processing, which have different destinations (e.g., human food, tanning, composting). This study aims to propose an alternative allocation rule of animal coproduct based on scientific sound biological processes. We developed a new biophysical allocation method to handle body-related co-products in an LCA of a meat-production system at the slaughterhouse gate. Our allocation method combines a generic metabolic growth model, which was widely used to predict body growth (increase in chemical composition) and a model to predict energy requirements as a function of protein and lipid growth for each tissue (Fig. 1), which should reflect a physico-chemical relationship among tissues.

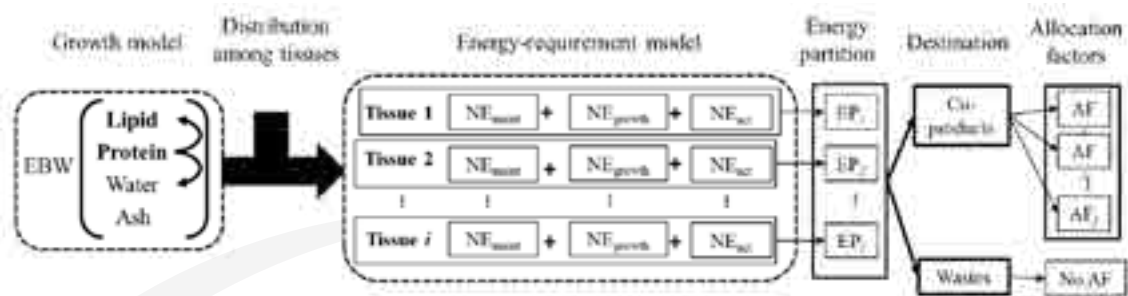


Figure 1: Model design to calculate biophysical allocation factors for body tissues combining a body-growth model and an energy-requirement model (EBW: empty body weight, NE_{maint} : net energy for maintenance, NE_{growth} : net energy for growth and NE_{act} : net energy for activity).

2. Methods (or Goal and Scope)

The biophysical allocation rule is based on 4-step method: the prediction of body growth with a Gompertz function, the calculation of net energy for maintenance, growth and activity, the calculation of energy partition for tissues, and finally the calculation of the allocation factor.

The method first predicts potential body growth rate from birth until maturity using a growth function. We chose the Gompertz function, based on available information from the literature (Emmans 1997; Hoch and Agabriel 2004a; Johnson et al. 2012; van Milgen et al. 2008; Whittemore et al. 1988), to model dynamics of body tissues. The Gompertz function requires few parameters, and its input data are readily accessible. In addition, its parameter values are based on biological characteristics of the animal modeled, rather than being simply mathematically fitted values (Wellock et al. 2004). These studies provide a theoretical and practical basis to calibrate the parameters of the same function adapted to different animal species.

The growth of the protein (kg) after birth is predicted by the Gompertz function as:

$$PROT(t) = PROT_0 \times \exp\left(\frac{\mu \times (1 - \exp(-D \times t))}{D}\right) \quad (1)$$

where μ is the Gompertz coefficient, which indicates the initial rate of protein growth, and $PROT_0$ is protein mass at birth. Gompertz parameter D can be derived as:

$$PROT_m = PROT_0 \times \exp\left(\frac{\mu}{D}\right), \text{ when } t \rightarrow +\infty \quad (2)$$

where $PROT_m$ is protein weight at maturity, which defines the upper limit of the asymptote of protein growth. Solving for D :

$$D = \mu / \ln\left(\frac{PROT_m}{PROT_0}\right) \quad (3)$$

According to Johnson et al. (2012), empty body weight (EBW), meaning body weight minus digestive content weight, includes protein (PROT), lipid (LIP) and water (W) contents, but the ash content is excluded because of its small percentage (less than 2% for cattle). Thus, there is:

$$EBW(t) = PROT(t) + LIP(t) + W(t) \quad (4)$$

Then, relationships between these component and EBW are assumed to follow the equations:

$$W(t) = \lambda * PROT(t) \quad (5)$$

$$LIP(t) = f * EBW(t) \quad (6)$$

$$PROT(t) = \left[\frac{1-f}{1+\lambda} \right] * EBW(t) \quad (7)$$

where λ is a constant ratio of body water to protein, which indicates the linear relationship between protein and water, and f is the normal content (%) of lipid in EBW, which is assumed to increase linearly with EBW from birth to maturity.

Combining equations (5), (6) and (7), lipid growth is described as a function of protein content:

$$LIP(t) = \frac{1}{2a(t)} * (-b(t) - \sqrt{b(t)^2 - 4a(t)c(t)}) \quad (8)$$

in which

$$a(t) = (f_m - f_0) / (EBW_m - EBW_0) \quad (8a)$$

$$b(t) = (f_0 - 1) + a(t) * [2 * PROT(t) * (1 + \lambda) - EBW_0] \quad (8b)$$

$$c(t) = f_0 * PROT(t) * (1 + \lambda) + a(t) * PROT(t) * (1 + \lambda) * [PROT(t) * (1 + \lambda) - EBW_0] \quad (8c)$$

where f_0, f_m is the lipid content (%) in EBW at birth and at maturity, respectively, and EBW_0 is EBW at birth. Johnson et al. (2012) describe the derivation in detail.

Equations 1-8 predict total protein and lipid gain during normal growth. To predict protein and lipid gain of individual tissue, we assumed that the percentage of protein (p_i) and lipid (q_i) in each tissue were constant during the growth period. Thus, there are:

$$PROT_i(t) = PROT(t) * p_i \quad (9)$$

$$LIP_i(t) = LIP(t) * q_i \quad (10)$$

Once protein and lipid contents were known, we used them as variables to calculate the metabolic energy requirements (step 2). Indeed, metabolic energy requirements can be described according to a hierarchical flux of energy: from energy intake (gross energy) to digestible energy, where loss of digestibility occurs (feces), then to metabolizable energy (loss of energy due to methane emission and urine), and finally to net energy (loss of energy due to heat production) (NCR 1998; Noblet and Van Milgen 2004). We used net energy to express requirements because it directly reflects the amount of energy used by an animal's body for biological processes. Body metabolism is based on the biological processes of maintenance, growth and activity. The maintenance function requires energy to maintain normal metabolic functions of the animal ("energy for maintenance"). The growth function requires energy to increase body weight ("energy for growth"). The activity function requires energy to obtain food, water and shelter ("energy for activity") (IPCC 2006). All of this energy consumption can be estimated as a sum of net energy requirements for body tissues. Body metabolism is influenced by multiple factors, such as nutritional, genetic and environmental parameters (Micol et al. 1993). We considered only normal requirements for potential body growth and did not consider these factors. Therefore, we assume that the animal has *ad libitum* access to feed and that energy intake will meet its total metabolic energy requirements.

We used net energy for maintenance (NE_{maint}) to express the energy required for the animal's basic metabolic function, when no body weight is gained or lost (no weight change). Several studies investigated the energy system for animal growth and body composition and assumed that maintenance energy is directly proportional to the protein mass in the body (Emmans 1994). We kept this assumption because it enables allocating NE_{maint} to body tissues that have different protein

contents. We used the following equation to calculate NE_{maint} , which equals the energy required to synthesize protein minus the energy lost as heat during protein degradation (Johnson et al. 2012):

$$NE_{\text{maint}}(t) = \left(\frac{1}{Y_{P,S}} - Y_{P,D} \right) * k_{P,D} * \varepsilon_P * PROT(t) \quad (11)$$

where $Y_{P,S}$ is the efficiency of protein synthesis, $Y_{P,D}$ is the efficiency of protein degradation, ε_P is protein energy content, and $k_{P,D}$ is the protein degradation coefficient, which can be determined given the fractional synthesis rate ($k_{P,S}$) and the net accretion of proteins. Thus, there is a balance between synthesized proteins and degraded proteins:

$$PROT * k_{P,D} = PROT * k_{P,S} - \text{Net accretion of proteins} \quad (11a)$$

According to Loblely et al. (1980), the fractional synthesis rate varies among tissues:

$$k_{P,S} \text{ for whole body} * \text{total protein} = \sum k_{P,S} \text{ for tissue} * \text{tissue protein} \quad (11b)$$

By definition, net energy requirements for growth (NE_{growth}) indicate the energy for protein and lipid retention in the body; so, the rate of NE_{growth} can be described by the growth rate of protein and lipid in individual tissues, following this equation:

$$\frac{dNE_{\text{growth}}}{dt} = \frac{\varepsilon_P}{Y_{P,S}} * \frac{dPROT}{dt} + \frac{\varepsilon_L}{Y_{L,S}} * \frac{dLIP}{dt} \quad (12)$$

where $Y_{L,S}$ is the efficiency of lipid synthesis, ε_L is the lipid energy content, and $dPROT/dt$ ($dLIP/dt$) is the rate of protein (lipid) retention in potential growth, which can be estimated according to the model of body weight prediction mentioned above. Therefore, NE_{growth} for a given day can be expressed as:

$$NE_{\text{growth}}(t) = \frac{\varepsilon_P}{Y_{P,S}} * [PROT(t) - PROT(t-1)] + \frac{\varepsilon_L}{Y_{L,S}} * [LIP(t) - LIP(t-1)] \quad (13)$$

As for net energy for activity (NE_{act}), Johnson et al. (2012) assumed that NE_{act} was a function of EBW with a constant coefficient. They argued that the coefficient for activity energy depends on rearing conditions (e.g. stall, pasture), and the activity costs based on this assumption correspond to the empirical curve response in the Australian Feeding Standards (SCA 1990). However, to obtain a more generic model for calculating NE_{act} than this country-specific model, we applied the method of IPCC (2006):

$$NE_{\text{act}}(t) = C_{\text{act}} * NE_{\text{maint}}(t) \quad (14)$$

where the coefficient (C_{act}) corresponds to the animal's feeding situation. We assumed that energy requirements for activity are modest ($C_{\text{act}} = 0.17$ for cattle, Table 10.5 of IPCC (2006)).

The third step allows the calculation of energy partition for tissues. Since the net energies for maintenance and growth are directly related to the protein and lipid contents in the body, we assumed that both were attributed to tissues according to their protein and lipid percentages:

$$NE_{\text{maint},i}(t) = \left(\frac{1}{Y_{P,S}} - Y_{P,D} \right) * k_{P,D} * \varepsilon_P * PROT(t) * p_i \quad (15)$$

$$NE_{\text{growth},i}(t) = \frac{\varepsilon_P}{Y_{P,S}} * [\Delta PROT(t)] * p_i + \frac{\varepsilon_L}{Y_{L,S}} * [\Delta LIP(t)] * q_i \quad (16)$$

$$NE_{\text{act},i}(t) = C_{\text{act}} * NE_{\text{maint},i}(t) \quad (17)$$

where $\Delta PROT(t)$ ($\Delta LIP(t)$) is calculated as the difference in protein (lipid) weight between the initial (at birth) and final protein (lipid) contents on day t , and p_i and q_i are the percentages of protein and lipid of tissue i out of the total protein and lipid contents of the body, respectively. We assumed that the percentages of protein and lipid in each tissue were constant during growth and that $\sum p_i = \sum q_i = 1$.

The last step (4) allows calculating the allocation factor according to ISO 14040/14044 rule i.e. all the inputs/output are allocating to the co-products excluding de facto tissues considered as waste from the allocation procedure :

$$F_{\text{alloc},j} = \frac{EP_j}{\sum_j EP_j} \quad (18)$$

The method was tested on beef cows. We assumed that growth starts at birth, with 50 kg of EBW₀ composed of 12 kg of protein and 3 kg of lipid until slaughter age at 495 EBW₁. The mature weight (EBW_m) was assumed to be 600 kg, with 105 kg of protein and 180 kg of lipid. The categories of beef products and co-products were defined according to CMWG (2015): human food (edible tissues such as muscles or the liver), category 1/2 (C1/C2) by-products (tissues considered as waste), spreading/compost (e.g., digestive contents) and four C3 co-products - processed animal protein (e.g., blood), gelatin (e.g., bone), fat and greaves, hide for tanning.

3. Results (or LCI)

The simulation ran from birth until maturity (1041 days), and we calculated allocation factors at slaughter age (509 days). Metabolic energy requirements were calculated for each tissue. Although both protein and lipid increased over time, growth rates differed for carcass and non-carcass tissues (Fig. 2). The deposition rate of carcass protein is higher than that for non-carcass protein due to its increasing proportion during fattening. Likewise, lipid deposition in carcass tissues is slightly higher than that in non-carcass tissues.

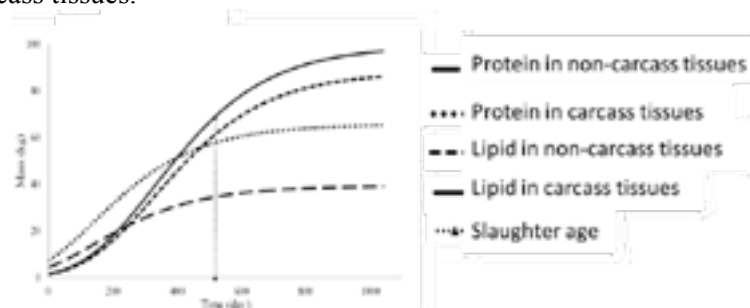


Figure 2: 1Growth curves of protein and lipid mass in carcass and non-carcass tissues as functions of time (per day) for beef cow from birth to maturity (the vertical line indicates slaughter age).

Total net metabolic energy requirements at slaughter age, calculated as cumulative energy requirements during the growth period, were 31,539 MJ, of which 56% was energy for maintenance, 34% was energy for growth and 10% was energy for activity. The rates of energy for maintenance and activity increased with body growth, while the rate of energy for growth decreased, since protein and lipid growth rates decreased. Metabolic energy requirements were attributed differently to carcass and non-carcass tissues; at slaughter age, carcass tissues required about 43% of total net energy, while non-carcass tissues required the remaining 57% (29% for the GIT, 5% for the liver, and 23% for the others). According to equation (11), energy requirements for maintenance depend on both the protein content of a tissue and its protein degradation rate. Therefore, the GIT and carcass tissues required a high percentage of total energy for maintenance, while the liver and other non-carcass tissues required little energy for their maintenance. Conversely, metabolic energy requirements for growth are a function of protein and lipid growth rates; so, carcass tissues had higher metabolic energy requirements than non-carcass tissues (Fig. 3)

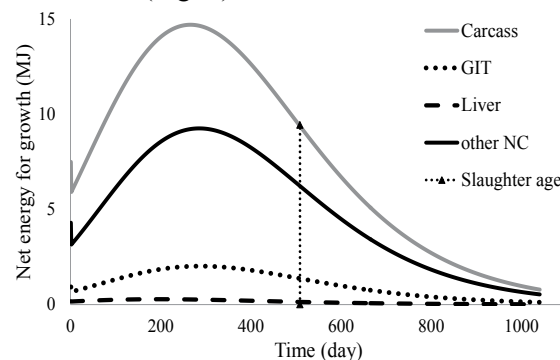


Figure 3: Net energy requirements for growth of different tissues: carcass, gastrointestinal tract (GIT), liver and other non-carcass tissues (other NC) as a function of time (day) for beef cow from birth to maturity (the vertical line indicates slaughter age).

Finally, when comparing allocation methods according to destinations of body tissues (Fig. 4), the biophysical allocation method induced smaller level of allocation for human food (50%) than mass (56%) and protein (62%) based methods, and higher level than dry matter (38%) based method. Economic allocation was significantly different from the other methods. Since economic allocation is based on economic values of co-products, the edible co-products as human food under economic allocation had an allocation factor of 95%, compared to 38-62% for the other four allocation methods. However, economic allocation method induced smaller allocation factors for C3 co-products than other physical allocation methods. For co-products destined to processed animal protein, the biophysical allocation method induced higher level of allocation (17%) than other methods, because some high energy-required GIT tissues were classified as pet food. Gelatin had similar allocation factors (6-10%) among methods (except economic allocation). Fat and greaves together had a higher allocation factor using the DM allocation method (36%) than the others (1-19%) due to a larger percentage of adipose tissues (e.g., tallow and fat). Hide for tanning had higher allocation factors under protein allocation (12%) than others (4-7%), due to its higher protein contents.

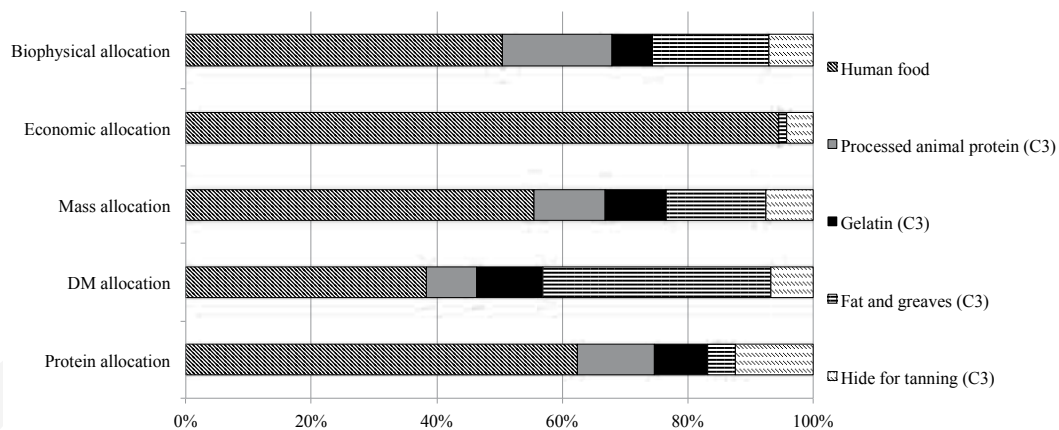


Figure 4: Allocation factors for different product destinations of beef cow at slaughter age according to biophysical, economic, mass, dry matter (DM) and protein allocation methods C3: category 3 according to CMWG (2015))

4. Discussion

The first submodel of simulation predicts protein and lipid growth based on the Gompertz function. It indicates that lipid increases quadratically and protein increases more linearly (Fig. 10), which was confirmed by the literature review of Owens et al. (1995). Although we applied the Gompertz function for genetically standardized animal growth, it can be adjusted easily with observed data to consider variability among animal types, species and environmental conditions, because only 3 parameters are required (i.e., Gompertz coefficient μ , Gompertz parameter D and protein mass at birth $PROT_0$).

The second submodel calculates metabolic energy requirements of different tissues. The common view is that the NE_{maint} is a function of average metabolic body weight (i.e., $BW^{0.75}$), which estimates the average metabolic energy requirements (per day) of an animal (IPCC 2006). However, such an exponential function cannot make a direct link between body tissues and metabolic energy requirements. Therefore, it cannot reflect that certain visceral organs have higher maintenance requirements than muscle tissues (Ortigues and Doreau 1995). We used a linear equation to calculate NE_{maint} as a function of protein content in individual body tissues. The energy-related parameters $Y_{P,d}$, Y_P , ε_P were assumed to be constant, and their values were those commonly used for cattle (Emmans 1997; Johnson et al. 2012; Roux 2014). The protein degradation rate k_P varies among cattle tissues, which indicates high requirements for NE_{maint} of several visceral organs, especially GIT organs. However, such detailed data may not be available for other animal gender, breeds or species. In such situations, we suggest using a single value of k_P for each tissue as well as for the entire body. In this way, NE_{maint} is determined only by tissue protein content. Future research could estimate metabolic energy requirements for each tissue to obtain additional data for k_P .

The biophysical allocation method is in accordance with the ISO (2006) standard, since it reflects mechanisms underlying a physical relationship (i.e., metabolic energy requirements) among co-products. Biophysical allocation considers multiple characteristics (i.e., protein and lipid contents) of co-products, which may help to decrease differences among stakeholders (e.g., meat producers vs. leather producers) points of view, who may prefer different allocation rules. Unlike the protein allocation method, in which co-product impacts are driven only by their protein contents, biophysical allocation reflects the energy cost of building tissues upstream, regardless of their fates downstream. It also reveals the cause-effect relationship between tissues according to the energy required to maintain their physiological functions. It is a change in perspective as it proposes a rule based on building the co-products, while the other methods are based on outputs characteristics (weight, price, chemical composition), and therefore on their destination and use. These destination and use of the co-products are typically under the concern of the different stakeholders with different points of view difficult to conciliate. The biophysical method taking into account the composition of tissues and their metabolic role induces a higher allocation factor on inner organs compared to the other methods. This is particularly sensitive for tissues classified in C1-C2 by-products. Although energy partition indicates that tissues as C1-C2 by-products had large energy requirement ratios, their allocation factors were zero according to ISO standard. It is a specific characteristic in Europe, for cattle, where these products are not targeted to human food, but waste. It would not be the same for other species (e.g. pig), or other cultural context where these products have more diverse uses. Although biophysical allocation factors varied over time, once the growing time is fixed (by setting the final EBW), the parameters are not influenced, unlike in economic allocation. Moreover, the biophysical allocation method can be applied to dairy farming systems to allocate impacts to milk, calves and live animals at the farm gate, and then to allocate impacts of live animals to their body tissues when sent to slaughter.

Although biophysical allocation factors varied over time, once the growing time is fixed (by setting the final EBW), the parameters are not influenced, unlike in economic allocation. Moreover, the biophysical allocation method can be applied to dairy farming systems to allocate impacts to milk, calves and live animals at the farm gate, and then to allocate impacts of live animals to their body tissues when sent to slaughter. For example, net energy for lactation should be attributed to milk production (Thoma et al. 2013), and the IPCC (2006) provides a method to calculate this net energy as a function of milk production and its fat content. Therefore, this new method is robust and flexible enough for application to different animal types (e.g., dairy or beef cows). Our study focused on development of the new method for calculating allocation factors, without using it to estimate environmental impacts. Thus, future research should include a complete LCA in which the potential impacts resulting from different allocation methods are compared.

5. Conclusions

Biophysical allocation follows the hierarchical rule of the ISO standard and can differentiate characteristics of livestock co-products. This method does not consider the fate of co-products but considers only the cost of building them. This approach can be considered an original contribution to international debates on allocation methods applied to livestock products in LCA. It should be considered and discussed by stakeholders in livestock-production industries.

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62. How important is the region in determining the nutritional strategy applied to broiler systems in order to reduce environmental impact?

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ABSTRACT

The environmental impacts associated with broiler production arise mainly from the production and consumption of feed. The aim was to apply a LCA method to develop a formulation tool for broiler diets in two regions (the UK and US). This was designed to target and reduce specific environmental impact categories individually. Using linear programming, least cost broiler diets were formulated for breeds commonly reared in each region. The environmental impact of the systems was defined using 6 metrics: global warming potential (GWP), fresh water eutrophication potential (FWEP), marine eutrophication potential (MEP), terrestrial acidification potential (TAP), non-renewable energy use (NREU) and agricultural land use (ALU). Diets were then formulated for each region to minimise each impact category, without compromising bird performance. In most cases the diets formulated for environmental impact objectives increased the cost of the diets close to the maximum limit allowed (30%), with the exception of the least NREU diet in the UK. The results suggested there was potential to minimise one impact category, whilst simultaneously reducing the impact of other categories compared to least cost, through diet formulation in the UK. In the US, there was no way to minimise one impact category through diet formulation without increasing the value of other impact categories. Deciding how to trade-off between different impact categories, or reduce simultaneously all impact categories considered in the tool, may require the development of a multi-criteria decision making approach to diet formulation.

Key words: Poultry, Broilers, Diet Formulation, Life Cycle Assessment, Environmental Impact Mitigation

1. Introduction

Global poultry meat production grew by 104% between 1990 and 2012 (FAO, 2016) and is predicted to become soon the world's most consumed form of animal protein (OECD/FAO, 2014). Much of the environmental impact associated with broiler chicken production arises from the production and consumption of feed (up to 75% of their environmental impact); therefore it is logical to focus on diet formulation and feed ingredient choice in order to mitigate this (Pelletier, 2008, Leinonen et al., 2012). For broiler systems, focusing only on global warming potential (GWP) would not be sufficient, as due to their reliance on high protein diets, broiler chicken production is associated with high eutrophication (EP) and acidification potentials (AP), as well as relatively high non-renewable energy use (NREU) (Leinonen et al., 2012). The majority of AP and EP caused by broiler production is due to emissions during manure storage and application, as a direct result of the excretion of N and P by the birds (Leinonen et al., 2012).

In this study a novel methodology was developed and applied to formulate diets for reduced impact in specific environmental categories, whilst not penalising bird growth, by applying a Life Cycle Assessment (LCA) approach integrated into a mechanistic diet formulation tool. The consequences of formulating diets for least impact in one environmental category on the other environmental impact categories and costs were investigated. Birds are fed diets based on very different dietary ingredients in the EU and North America, either because of legislation, trade agreements or climatic conditions, so the opportunities for reduction in specific environmental impact categories may be expected to differ between the two regions (Kebreab et al., 2016). Here the UK and US broiler systems represent broiler production in the two regions.

2. Method

2.1. Model structure

A LCA methodology was applied to a diet formulation tool to demonstrate the potential for reducing the environmental impacts associated with the production of broiler chicken meat via changes in their poultry diet in the UK and US. The system considered was

conventional broiler production, which is the predominant one in Europe and North America. The functional unit was the growth of one metric tonne of broiler live weight. In the UK poultry systems, the average broiler was a Ross 308 raised to a slaughter weight of 2.2 kg (Defra, 2014b); this broiler strain is used widely in Europe. In the US poultry systems, the average broiler was a Ross 708 raised to a slaughter weight of 2.8 kg (National Chicken Council, 2016). Three stages of broiler production were modelled for the UK and four were modelled for the US, in accordance with the nutritional specification requirements for each breed (Aviagen, 2014a, 2014b): the starter phase (hatching - day 10); the grower phase (day 11 - 24); the finisher phase (day 25 – 39 or slaughter, i.e. in the UK); and the withdrawal phase, from day 39 until slaughter (US only). The main compartment of material flow in the life cycle inventory consisted of the production of feed ingredients. The ingredients available to be incorporated into poultry diets in each region were based on input data from literature, national inventory reports, databases (e.g. FAOSTAT) and expert knowledge.

2.2. Impact assessment

The metrics used to quantify the environmental impacts of the different diet formulations were: GWP, EP, terrestrial AP (TAP), agricultural land use (ALU) and NREU. GWP was quantified as CO₂ equivalent (CO₂ eq) with a 100 year timescale. Under these conditions, 1 kg of CH₄ and N₂O emitted are the equivalent of 25 and 298 kg of CO₂ respectively (IPCC, 2006). The CO₂ eq. released due to land-use change was included and followed the PAS2050:2012-1 methodology (BSI, 2012). The EP impacts were separated into marine EP (MEP) for N-based emissions and fresh water EP (FEP) for P emissions, using the ReCiPe midpoint method (Goedkoop et al., 2009). Calculations of EP and AP followed the method of the Institute of Environmental Sciences at Leiden University (<http://cml.leiden.edu/research/industrialecology/>). Non-renewable energy use was calculated in accordance with the IMPACT 2002+ method (Joliet et al., 2003).

2.3. Diets formulated

Nutritional values for all ingredients available to poultry diets in each region were taken from Premier Nutrition (2014) and placed into a matrix. The prices of region-specific ingredient were obtained from grey literature and personal communication with industry (i.e. USDA, Evonik, Aviagen, Defra, IndexMundi). Using the linear programming tool “Solver” (Mason, 2012), least cost broiler diets were formulated for each growth phase in each region that met the broiler energy and nutrient specifications (Aviagen, 2014a, Aviagen, 2014b). Since these requirements were met in every diet formulated, it was assumed that growth rate per kg of feed was unaffected. An inventory of feed ingredients specific to each region was compiled in Simapro and this software was used to conduct the LCA calculations. Maximum and minimum inclusion rates for each ingredient were developed by referring to diet composition manuals and expert feedback, so that palatability, inhibition of digestibility or variability in specific ingredients did not adversely affect bird performance (Leinonen et al., 2013). The minimum crude protein requirement of each breed, as was defined by industry for each phase, was at least met by each diet; it was allowed to fluctuate above this level which enabled for more or less synthetic amino acid inclusion.

Fossil fuel inputs to fertilizer production, emissions resulting from the spreading of fertilizers, energy inputs to processing (drying, grinding etc.) and transport, all contribute heavily to the impacts associated with feed production. The average GWP, FWEP, MEP, TAP, NREU and ALU per kg of each ingredient were added to the list of ingredient properties in the diet formulation tool. Where system separation was not possible, coproduct allocation within the feed supply chain was conducted using economic allocation, in accordance with the method recommended by the FAO (2015). A sum of the environmental impacts of feed ingredient production and litter management (see section 2.4) provided the total environmental impact associated with the diet formulation. Therefore, by using linear

programming, a diet was formulated to minimise each impact category. Each diet was compared to the least cost diet, which would most closely represent a contemporary commercial broiler feed composition. All diets formulated for environmental impact objectives were subject to a 30% maximum cost increase in comparison to the least cost diet (Mackenzie et al., 2016).

2.4. Manure model

The nitrogen (N), phosphorous (P) and potassium (K) content of the poultry manure was calculated using the mass balance principle; the nutrients retained in the broiler's body (McGahan and Tucker, 2003) were subtracted from the total N, P and K supplied by the diet. A value for each impact category was calculated based on the excretion of one kg of each nutrient. The manure model estimated the emissions of ammonia (NH₃), nitrous oxide (N₂O) and nitrogen oxides (NO_x) and phosphate (PO₄) that occurred during housing, storage, and application to field. The emissions were accounted for in accordance with the methodologies for calculating emissions from managed soils, livestock and manure management and storage, outlined by the IPCC (2006). After removal from the house, manure was assumed to be stored in covered field heaps for 6 months prior to spreading on the land (Gates et al., 2008, Defra, 2011). The total N₂O was assumed to equate to the same value as NO_x as was assumed in the Velthof et al. (2012) model. Due to the limited emissions data available for the US, and to keep the methodologies used consistent, the emissions arising from housing and storage in the two systems were assumed to be equitable, as a percentage of the nutrients released in the manure.

Broadcast field spreading, followed by incorporation through tillage (within 24 hours), was assumed for both regions due to manure management statistics and local codes of practice (USDA, 2009, Defra, 2014a). It was assumed that only 1.6% of K was lost before it reached the field, whilst the loss of P before it reached the field was negligible (Defra, 2011). Phosphate emissions at field were calculated based on emissions factors reported by Struijs et al. (2011). N₂O and NO₃ emissions at the field were calculated based on IPCC (2006) emission factors which were adapted to the climatic conditions of each region. NH₃ emissions at field were based on Webb and Misselbrook (2004) and Moore et al. (2011). The nutrients incorporated into the soil were assumed to replace equivalent nutrients, which would have otherwise been delivered in the form of synthetic fertilizers, by 70% (N), 80% (P) and 100% (K) to account for over-application (Williams et al., 2006): predominantly ammonium nitrate, potassium chloride, potassium sulphate and di-ammonium phosphate. Offsetting the need to apply as much synthetic fertilizer can be credited to the poultry production system, as is standard in LCAs (Leinonen et al., 2012).

2.5. Uncertainty

A Monte Carlo approach was applied to the model to quantify the potential uncertainties in the study (e.g. measurement errors, variation in production data due to differences in crop yield, feed intake, bird mortality etc.) in order to make it possible to evaluate differences between the least cost diet and the diets formulated for environmental impact objectives. During each simulation, the model was run in parallel 1000 times and, during each run, a value of each input variable was randomly selected from a predetermined distribution for said variable; the method is described comprehensively in Mackenzie et al. (2015). Environmental impact levels were reported as significantly different where one diet had a greater impact than the other in more than 95% of the parallel simulations of the LCA model ($p < 0.05$).

3. Results

3.1. Diet formulations - UK

In the UK a standard least cost diet, across all three stages, was composed of 483 g/kg wheat, 66.8 g/kg rapeseed, 241 g/kg soymeal and 124 g/kg field peas, plus oil and specialist ingredients (Table 1). In the least GWP diet, soymeal was reduced in favour of maize gluten meal, rapeseed meal and sunflower meal, which were incorporated at inclusions of 48.3, 34.2 and 88.6 g/kg respectively; wheat was also reduced, when compared to the least cost diet, at 453 g/kg, but whole rapeseed remained the same. In the least FWEP diet, wheat inclusion was increased but rapeseed was removed completely. In the least MEP and TAP diets maize usurped wheat as the primary energy ingredient (577 and 630 g/kg respectively) and had an increased soy oil content relative to the least cost and least GWP diets. The NREU diet had a greater inclusion of wheat and soymeal when compared to the least cost diet. Like the least MEP and TAP diets, the least ALU diet was primarily maize based, but also contained 66.3 g/kg of whole rapeseed.

The production of the functional unit on the least cost diet had a GWP, FWEP, MEP, TAP, NREU and ALU impact value of 3060 kg CO₂ eq., 0.6657 kg P eq., 27.38 kg N eq., 69.61 kg SO₂ eq., 16.63 GJ and 4675 m² respectively. All least environmental impact diets significantly increased costs by between 16 and 30% when compared to the least cost diet, except for the NREU diet which significantly increased cost by just under 4% (Figure 1). The least GWP diet decreased the GWP by 37%, but increased NREU by 31% and TAP by 8.2%; although every other impact category showed a potential increase when broilers were fed this diet, these were not significant. The Least FWEP diet decreased the values of all impact categories, when compared to the least cost diet, with the exception of TAP which increased by 0.07% and the NREU which was not significantly different. The least MEP and TAP diets showed a similar trend in the reduction of environmental impacts; however every impact value was lower in the least TAP diet except for MEP. The least NREU diet was the only diet which had a lower NREU value than the least cost diet. The least ALU diet reduced the GWP, FWEP and MEP significantly compared to the least cost diet, but resulted in a small increase in TAP (0.62%) and a 53% increase in NREU.

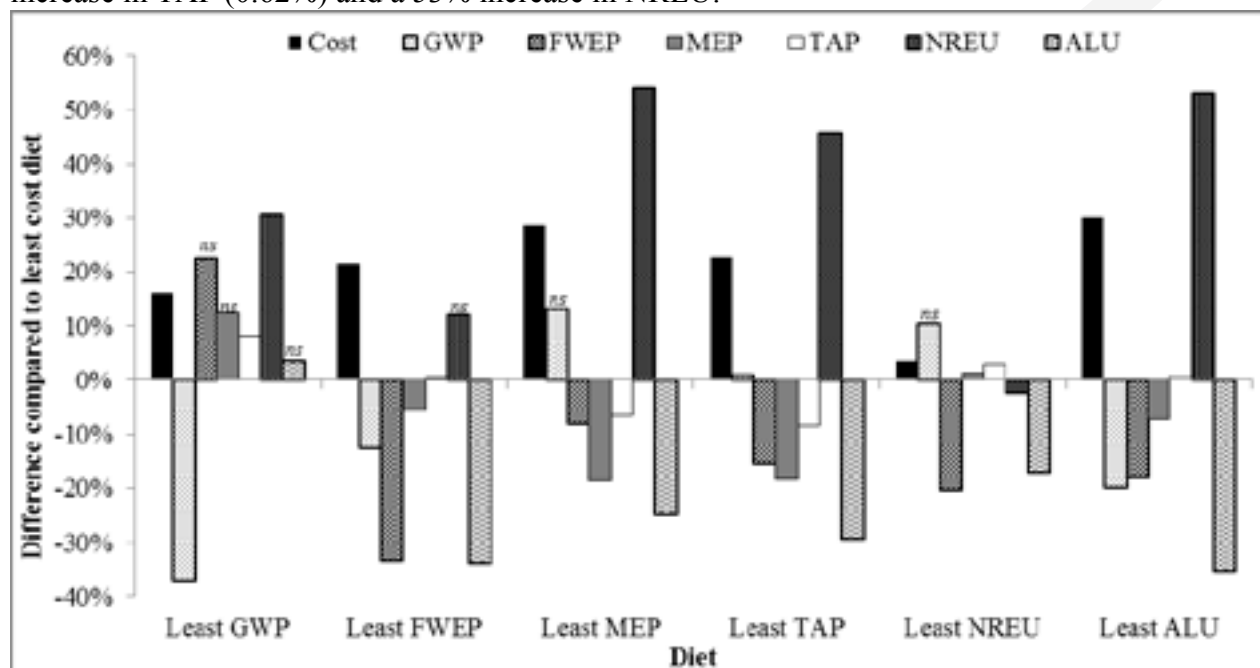


Figure 1: Environmental impacts of different UK broiler diets, each formulated to reduce a specific environmental impact category, as compared to a least cost formulation baseline. The impact categories tested include GWP (kg CO₂ eq.), FWEP (kg P eq.), MEP (kg N eq.), TAP (kg SO₂ eq.), NREU (MJ) and ALU (m²). All impact category values were significantly different ($p < 0.05$) from their corresponding value produced by the least cost diet unless otherwise stated (*ns*).

3.2. Diet formulations – US

In the US, a standard least cost diet was composed of 611 g/kg maize and 208 g/kg soymeal plus oil, animal coproducts and additives (Table 1). In contrast to the UK diets, the US diets consisted of a higher percentage of soymeal in the starter phase, and lower percentage inclusions in the later phases. In the least GWP diet maize incorporation was reduced dramatically (307 g/kg) when compared to the least cost baseline and instead barley was included as an additional energy source (262 g/kg). Ingredients derived from soybean increased, which was the opposite of what happened in the UK least GWP diet. In the least FWEP diet wheat usurped maize as the primary energy ingredient and was included at a rate of 664 g/kg. The incorporation of maize and fishmeal was high in the least MEP and TAP diets when compared to other diet formulations. The least NREU incorporated 277 g/kg of maize and 262 g/kg of barley, much like the least GWP diet, but contained more soy bean (106 g/kg) and slightly less soymeal (228 g/kg) than that diet. The least ALU contained the least soybean and its derivatives compared to all other US diet formulations and the highest incorporation of specialist ingredients.

The production of the functional unit on the least cost diet had a GWP, FWEP, MEP, TAP, NREU and ALU impact value of 917.7 kg CO₂ eq., 0.4154 kg P eq., 20.66 kg N eq., 63.16 kg SO₂ eq., 12.24 GJ and 2775 m² respectively. All least environmental impact diets had increased costs of between 23% (least TAP) and 30% (least FWEP) when compared to the least cost diet (Figure 2). The least GWP diet decreased GWP by 6.7% and NREU by 15%, but increased every other impact category significantly compared to the least cost diet. The least FWEP diet caused an 18% decrease in MEP, but increased every other impact category when compared to the least cost diet. The FWEP and NREU both increased in the least MEP diet, with insignificant reductions seen in the GWP. In the least TAP diet only MEP and TAP were reduced compared to the least cost diet. The least NREU had a significantly reduced GWP and NREU when compared to the least cost diet, but significantly increased FWEP, MEP, TAP and ALU. The least ALU diet only significantly reduced the ALU (by 18%) compared to the least cost diet.

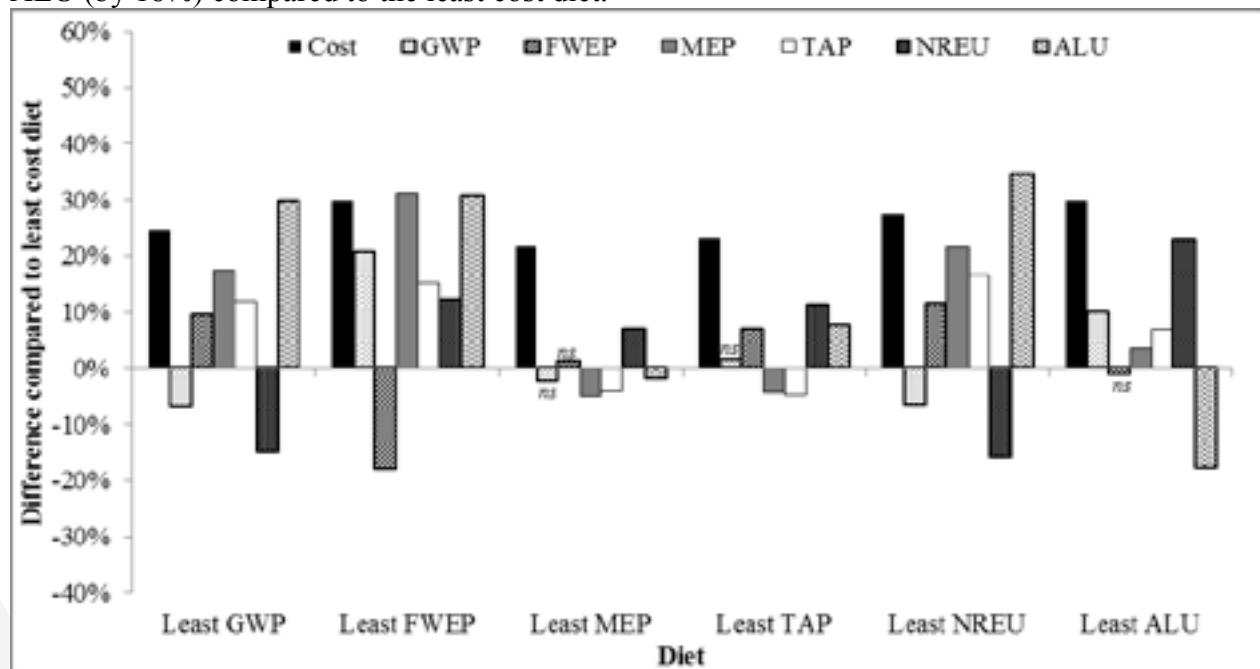


Figure 2: Environmental impacts of different US broiler diets, each formulated to reduce a specific environmental impact category, as compared to a least cost formulation baseline. The impact categories tested include GWP (kg CO₂ eqv.), FWEP (kg P eq.), MEP (kg N eq.), TAP (kg SO₂ eq.), NREU (MJ) and ALU (m²). All impact category values were significantly

different ($p < 0.05$) from their corresponding value produced by the least cost diet unless otherwise stated (*ns*).

Table 1: Percentage inclusion of the main ingredients included in the diet formulations (Not including synthetic amino acids and other specialist ingredients, e.g. enzymes and premix components).

Ingredient	Least Cost		Least GWP		Least FWEP		Least MEP		Least TAP		Least NREU		Least ALU	
	UK	U S	UK	US	U K	US	UK	US	UK	US	UK	US	UK	US
Wheat	48	0	45	0	63	66	0	0	0	0	55	0	0	0
Maize	0	61	0	31	0	0	56	66	63	65	0	28	59	61
Maize gluten meal	0.3 3	2.4	4.8	0	4.8	4.9	0	0	0	0	0.18	0	4.8	4.9
Rapeseed Whole Rapeseed meal	6.7 0	0	6.7 3.4	0	0	0	0	0	0	0	6.7	0	6.6	0
Barley	0	0	0	26	0	0	0	0	0	0	0	26	0	0
Sunflower r meal	0	-	8.9	-	0	-	0	-	0	-	0	-	0	-
Soybeans soy meal	0 24	0 21	0 5.9	4.4 25	0 20	0 15	0 27	0 21	0 26	0 24	0 30	11 23	0 21	0 14
Field peas	12	-	12	-	0	-	0	-	0	-	0	-	0	-
Vegetable Oil	0	0	4.6	0	0	0	0	0	0	0	0	0	0	0
Soy Oil	4.2	2.2	6	4.7	3.7	2.8	4.8	2.0	3.0	2.8	4.2	4.1	0.5	0.9
Fishmeal	0.4	0	5	4	5.0	2.9	5.0	5.0	5.0	5.0	0.42	0	5.0	5.0
Meat and Bone meal	-	2.7	-	2.7	-	2.7	-	0	-	0	-	2.7	-	2.7
Poultry Offal	-	3.7	-	3.7	-	3.7	-	3.7	-	0	-	3.7	-	3.7
DDGS (Maize)	-	4.7	-	0	-	0	-	0	-	0	-	0	-	5.0

4. Discussion

A diet formulation model was developed which utilised LCA methodology to formulate broiler diets for environmental impact objectives in the UK and the US for the first time. The least environmental impact diets increased the cost of the diets greatly with the exception of the least NREU diet in the UK. In most cases the increase was close to the limit of 30%. Although the limit was set arbitrarily it would be unrealistic to consider higher increases in diet costs (Mackenzie et al., 2016). As is consistent with industrial practices, the least cost diet in the UK was based on wheat and soya and in the US it was based on maize and soya. The least environmental impact diets forced the inclusion of some alternative cereals, namely the use of maize in the UK least MEP and TAP diets, and barley and wheat for the US least GWP and FWEP diets respectively. In the UK, the GWP associated with broiler feed production could be reduced considerably by incorporating protein sources which

had a lower embedded CO₂ eq. burden associated with them than soya, namely rapeseed meal and sunflower meal. This is because soya is mainly imported from South America in the UK broiler systems and is in some cases associated with land-use change (Leinonen et al., 2012), which is not the case in US broiler production. GWP was minimised in the US by including barley, which has a low GWP, but high MEP when compared to maize, and by removing DDGS maize. Wheat was used as the primary energy crop in the least FWEP diet in both the UK and US due to its lower associated P emissions compared to maize. The least FWEP US diet had an increased MEP due to wheat's higher impact value for this impact category.

The MEP and TAP diets were highly sensitive to N emissions and as such ingredients derived from N fixing crops which require no nitrogen fertilizer, for example soya, were readily incorporated. Maize was favoured over wheat in both instances, due to wheat having a greater associated MEP impact value and despite wheat having a slightly lower TAP value than maize. If the diet were to be formulated deprived of maize, wheat would be used as the primary energy ingredient, less soymeal would be incorporated and more soy oil would be used to satisfy the bird's nutritional specifications; soy oil is more than twice as environmentally impacting in every impact category than soymeal. The change in the impact categories in the UK and US least ALU diets were similar, showed reduced use of oils and incorporated as many specialist ingredients and as much fishmeal as possible. Rapeseed and more soymeal had to be included in the UK diets, due to this region's system having no access to alternative protein sources, such as meat and bone meal.

In the US, there was no way to minimise one impact category through diet formulation without increasing other impact categories, however the UK showed more potential. For instance, in the UK it would be possible to reduce several impact categories without simultaneously increasing others significantly and this could be achieved by constraining the maximum TAP increase compared to the least cost diet to zero when formulating a least FWEP diet. This diet would be 21% more expensive than the least cost formulation, but would reduce the GWP (by 0.13%), FWEP (by 33%), MEP (by 5.6%) and ALU (by 44%) compared to the least cost diet. This diet would have an unchanged TAP value and an insignificantly increased NREU value compared to the least cost diet. Similarly, if the UK least NREU diet was formulated, whilst the MEP and TAP were constrained so that they may not increase above the levels they were at in the least cost diet, a diet could be formulated that would decrease the FWEP (by 22%), TAP (by 2.2%) and ALU (by 19%) compared to the least cost diet; the GWP would be insignificantly increased. This diet would cost 2.1% more than the least cost diet.

In most cases, where NREU was not targeted specifically, the resulting least environmental impact diets resulted in increased NREU; the exception to this was the least GWP diet in the US. Minimising GWP through diet formulation in both regions' systems was disadvantageous in many respects, causing significantly greater TAP and NREU in the UK and significantly increasing MEP, TAP and ALU in the US compared to their corresponding least cost diets. It is important to acknowledge this, as GWP is often the impact category to which stakeholders pay most attention when modelling the environmental impact of livestock systems. Further development of the diet formulation model to integrate a multiple criteria decision making approach for formulating broiler diets would enable multiple environmental impact objectives to be considered to help resolve this issue.

5. Conclusion

Methodologies such as the one applied here, in which a cradle to farm gate LCA model was integrated into a diet formulation tool, can allow nutritionists and livestock producers to integrate environmental objectives into diet formulation, facilitating sustainable feeding strategies and management choices. For instance, it is clear that there is potential to

reduce every environmental impact category through diet formulation in the UK. For the results presented here, there was no way to minimise the impact of feed production for one impact category without affecting another through diet formulation in the US, therefore it might be reasonable to suggest a multifaceted approach that targets more than one impact category at a time. Depending on environmental impact objectives, consideration of the effect of diets beyond GWP might be something to take into account. For non-ruminant production systems there is increasing concern regarding the associated EP and AP impacts (LEAP, 2015). This study emphasises clearly that targeting only GWP is not necessarily a sustainable solution to mitigating the environmental impact of the poultry industry.

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137. How much animal-source food can we produce while avoiding feed-food competition?

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ABSTRACT

Livestock directly contribute to food supply by providing essential nutrients to humans, and indirectly support cultivation of food crops by providing manure and draft power. Livestock, however, also consume human-edible food or graze on land suitable for cultivation of food crops. As we face the challenge of feeding 9.7 billion people by 2050, preferably without expanding the amount of agricultural land, there is an increasing need to avoid competition for land between animals and humans. We performed a review on studies that provide insight into the amount of animal-source food (ASF) produced without feed-food competition. So called default livestock are only fed with co-products, food-waste, crop-residues, or biomass from grazing land. Results showed, that between 7 g and 27 g of animal source protein per person per day can be produced from default livestock. On average, it is recommended to consume about 57 g of protein from ASF or plant-origin per person per day. Although ASF from default livestock does not fulfil the current global animal protein consumption of 32 g per person per day, about one third of the protein each person needs can be produced without competition for land between feed and food production. Default livestock, therefore, can have an important contribution to the future nutrition supply.

Keywords: land use, feed-food competition, livestock, food-waste, marginal land

1. Introduction

Livestock directly contribute to food supply by providing essential nutrients to humans, and indirectly support cultivation of food crops by providing manure and draft power. Livestock, however, also consume human-edible food or graze on land suitable for cultivation of food crops. As we face feeding 9.7 billion people by 2050, preferably without expanding the amount of agricultural land, there is an increasing need to avoid competition for land between animals and humans.

Current life cycle assessments of animal-source food products, however, do not provide insight into the competition for land between humans and animals. Van Zanten et al. (2016) developed the so-called land use ratio (LUR), which offers identification of livestock systems that contribute to global food supply and was developed to deal with feed-food competition. The LUR compares the amount of human digestible protein (HDP) in one kg of ASF with the maximum amount of plant based (HDP) that can be derived from food-crops grown on all land used to cultivate feed required to produce that one kilogram ASF. The LUR is calculated using the following formula:

$$LUR = \frac{\sum_{i=1}^n \sum_{j=1}^m (LO_{ij} \times HDP_j)}{HDP \text{ per kg ASF}}$$

where LO_{ij} is the land area occupied (m^2 year) to cultivate the amount of feed ingredient i ($i=1,n$) in country j ($j=1,m$) needed to produce 1 kg ASF, and HDP_j is the maximum amount of HDP that can be produced per m^2 year by direct cultivation of food crops in country j . The denominator contains the amount of HDP of 1 kg ASF.

A LUR <1.0 implies that livestock produce more HDP per m^2 than crops produce. Van Zanten et al. (2016) applied the LUR to three case studies: dairy cows on peat soil, dairy cows on sandy soil, and laying hens. The LUR's for the case of laying hens and dairy cows on sandy soil equaled about 2.1, implying that all land required to produce 1 kg HDP from laying hens or cows on sandy soil

could yield about twice the amount of HDP from human food-crops. For dairy cows on peat the LUR was 0.67. The LUR for cows on peat was lower than for cows on sandy soil because land used to grow grass and grass silage for cows on peat was unsuitable for production of food-crops. Results of the LUR showed that livestock production systems using mainly co-products, food-waste, and biomass from marginal land, can produce HDP more efficiently than crop production systems do.

Only when the LUR is zero, which can occur by feeding only co-products and food-waste, for example, or by cows grazing only on marginal land, is feed-food competition completely avoided. The availability of those leftover streams is, however, limited and, therefore, the amount of ASF produced based only on leftover streams is also limited. Ratios, such as the LUR or the protein and energy conversion ratio as developed by Wilkinson (2010), therefore, do not provide insight into the absolute amount of animal-source food (ASF) we can produce without avoiding feed-food competition.

Our aim was, therefore, to calculate the amount of ASF available per person in 2050, while avoiding the competition for land between humans and animals.

2. Methods

We performed a review on studies that provide insight into the amount of animal-source food (ASF) produced without feed-food competition. The criteria of including a peer reviewed study was that livestock should only be fed with co-products, food-waste, crop-residues, or biomass from grazing land (altogether referred to as leftover streams). Arable land is not used or used only minimally to produce feed ($LUR < 1.0$), only products that humans cannot or do not want to eat are fed to livestock, and biomass from marginal land is used to feed ruminants (Garnett, 2009; Rööß et al., 2016). Several recent studies have concluded that using leftover streams is important to reduce the environmental impact of animal source food (e.g. Schader et al., 2015; Rööß et al., 2016), but only five had an approach that focused solely on leftover streams: Elferink et al., 2008; Smil, 2014; Schader et al., 2015; Van Kernebeek et al., 2016; Van Zanten et al., 2016. Those five studies were, therefore, included in this review.

3. Results

Results showed, that between 7 g and 27 g of animal source protein per person per day can be produced from livestock fed on only products humans cannot or do not want to eat (table 1).

Elferink et al. (2008) concluded that about 27 g protein originating from pig meat can currently be consumed per person per day. Their calculation considered only available co-products, and did not consider food-waste and biomass from marginal land. Availability of co-products was based on average Dutch consumption of three crops: sugar beets, soybeans, and potatoes, which represent approximately 60 % of the co-products produced from the food industry in the Netherlands. They then calculated that Dutch person consumes on average 43 kg sugar, 18 kg soy oil, and 97 kg potatoes per year. Furthermore, they corrected for the total share of co-products produced in the Netherlands.

Smil (2014) concluded that in total about 200 million tons of meat (carcass weight) can be produced currently, resulting in about 9 g of protein per person per day. He based his calculation on the amount of available co-products, crop-residues, and biomass from grazing land, but he did not include food-waste. He assumed that globally 40 Mt meat can be produced from ruminants feeding on crop-residues, 40 Mt pig meat and 70 Mt chicken meat can be produced from monogastrics feeding on co-products, and 40 Mt meat can be produced from ruminants grazing on grasslands.

Schader et al. (2015) concluded that in 2050 about 26 g of meat, 2 g eggs, and 138 g milk can be consumed per person per day, resulting in protein supply of 9 g per person per day. Their calculation was based on the amount of available co-products and biomass from grazing land, but did not include food-waste. Bottom-up mass flows were used for the calculations, based on data from the Food and Agricultural Organisation (FAOSTAT, 2013).

Van Kernebeek et al. (2016) concluded that land use was most efficient if people (up-to a human population of 35 mln) would consume about 7 g of protein from ASF (mainly milk) derived from livestock fed mainly on co-products. Their calculation was mainly based on co-products and marginal land, and hardly on food-waste. They used linear programming to determine minimum land use required to feed the Dutch population.

Van Zanten et al. (2016) concluded that by feeding only those products that are not or cannot be consumed by humans to livestock, we could produce about 21 g protein per person per day. The calculation was based on the assumption that a **balanced, healthy** vegan diet (**based on peer reviewed articles**) was consumed, resulting in production of co-products not used by humans. Second, it was assumed that 10% of food waste was inevitable, and, available as livestock feed. Last, it was assumed that global grasslands were valued by cattle. By feeding co-products and food-waste to pigs about 14 g protein per person per day can be consumed. By using all grazing land about 7 g of protein from ASF per person per day. If only marginal grasslands are used 3 g of protein per person per day can be produced. So, of this 21 g, about 17 g was produced without competition between feed and food-crops for arable land (4 g less if grassland with potential for crop production was excluded).

TABLE 1. Different estimations of the protein production from animal source food from livestock production systems that only use feed products that are not in competition with humans: food waste, co-products, marginal ground and crop-residues.

	g protein per capita per day	food waste	co-products	marginal land	crop-residues	Products
Elferink et al. (2008) ¹	27		x			meat
Smil (2014) ²	9		x	x	x	meat
Schader et al. (2015) ³	9		x	x		meat, milk, eggs
Van Kernebeek et al. (2016) ⁴	7		x			meat, milk
Van Zanten et al. (2016) ⁵	21	x	x	x		meat and milk

¹ in: Journal of Cleaner Production, ² in: Global Food Security ³in: Journal of The Royal Society Interface, ⁴ in: international Journal of Life Cycle Assessment, ⁵ In: Animal

4. Discussion

The amount of protein from ASF per person per day calculated by Smil (2014), Schader et al. (2015), and Van Kernebeek et al. (2015) was relatively lower than Elferink et al. (2008) and Van Zanten et al (2016). Results of Elferink et al. (2008) were relatively high, even though, only food co-products were considered. These relatively high results were mainly caused by two assumptions: 1) Elferink et al. (2008) based his assumption on food intake in the Netherlands (which is above the global average food intake) resulting in a relatively high amount of co-products that become available during the processing of the food. 2) Looking at the co-products that become available soybean meal (SBM) has a high share in it. SBM has a high nutritional value for livestock and, therefore, growth performance of livestock can be relatively high.

The amount of ASF produced in Van Zanten et al. (2016) was higher because it included both food-waste and feed-food crops. The importance of food-waste as livestock feed was also recognized by Zu Ermgassen et al. (2016), who concluded that feeding heat-treated food-waste to livestock can reduce the land use impact of pork production within the EU by 20% (about 1.8 billion hectares of agricultural land). Feeding food-waste to livestock is currently not allowed, and some people question the legal status of food-waste (Zu Ermgassen et al., 2016). Zu Ermgassen et al. (2016) state that feeding food-waste to livestock can be a safe alternative if food-waste is heat-treated. Such practices are applied commonly in Japan and South Korea, where about 35% of the food-waste is fed to livestock (Zu Ergassen et al., 2016).

Furthermore, Van Zanten et al. (2016) **considered** feed-food crops by selecting those food ingredients in the vegan diet whose co-products **also** had a high nutritional value for livestock. Oil

production, for example, originated solely from soy cultivation, which resulted in the co-product Soybean meal (SBM). As mentioned above in Elferink et al. (2008) also all human required oil was obtained from soy cultivation; they concluded that about 27 g of protein per person per day could be consumed.

Land use efficiency to fulfil human nutritional requirements, therefore, can be achieved by maximising the human food output per hectare. Both the direct outputs (human edible crop product) and the indirect outputs (animal source food produced on crop co-product) should be considered. Selecting feed-food crops, therefore, can be a new strategy to increase land use efficiency. To analyse feed-food crops to increase land use efficiency, both direct food production via crop cultivation and indirect food production via animal source food production, needs to be assessed.

The results of this review, therefore, show the importance of considering food-waste and feed-food crops when aiming to reduce the environmental impact of the food system as they increases the amount of ASF that can be produced without feed-food competition. These mitigation strategies are particularly of interest as they are not currently applied.

Another mitigation strategy suggested, is the use of crop residues for animal production, which was only included in the study of Smil (2014). He estimated that crop residues fed to beef cattle would result in 2 grams protein per person per day, about 20% of all animal sourced protein produced on co-products in his study. The use of crop residues for livestock production, however, is a rather controversial strategy, as crop residues, are currently left on the field which might play an important role in maintaining soil fertility (Blanco-Canqui & Lal, 2008). Other studies, such as Van Zanten et al. (2016), therefore, excluded crop residues.

Although using different assumptions, each studies concluded that consuming a small or moderate amount of ASF (7 to 27 grams per person per day) by humans is most land use efficient. Currently, however, the average global consumption of animal protein is about 32 g per person per day. To avoid feed-food competition completely, the total world-wide consumption of ASF must, therefore, be reduced. We did not intend to calculate the amount of ASF people should eat. The results, however, show that livestock can contribute to sustainable nutrition supply by using food-waste and valuing crop co-products especially of feed-food crops.

5. Conclusions

Between 7 g and 27 g of animal source protein per person per day can be produced from livestock fed on only products humans cannot or do not want to eat. Within current livestock systems, co-products and biomass from marginal land are already used. Feeding food-waste and considering feed-food crops, however, are examples of mitigation strategies that currently can be implemented to reduce further the environmental impact.

On average, it is recommended to consume about 57 g of protein from ASF or plant-origin per person per day. Although ASF from default livestock does not fulfil the current global animal protein consumption of 32 g per person per day, about one third of the protein each person needs can be produced without competition for land between feed and food production. Livestock, therefore, does have an important contribution to the future nutrition supply.

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146. The effect of feeding strategy on environmental impacts of pig production depends on the context of production: evaluation through life cycle assessment

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ABSTRACT

LCA has been used in many studies to evaluate the effect of feeding strategy on the environmental impact of pig production. However, because most studies have been conducted in Europe the question of the possible interactions with the context of production is still in debate. The objective of this study was then to evaluate these effects in two contrasted geographic contexts of production, Brazil and France. The LCA considered the process of pig fattening, including production and transport of feed ingredients, feed production and transport to the farm, raising of fattening pigs, and manure storage, transport and spreading. Impacts were calculated at the farm gate and the functional unit considered was one kg of body weight gain over the fattening period. Performances of pigs were simulated for each scenario using InraPorc® population model (2000 pigs for each scenario considering between animal variability). The LCA calculations were performed for each pig according to its own performance and excretion, and the results were subjected to variance analysis. The results indicate that for some impacts there are clear interactions between effects of feeding program, origin of soybean and location of production. For climate change, the interest of phase feeding and incorporation of feed-use amino acids (FU-AA) is very limited and even counterproductive in Brazil with soybean from South, whereas it appears to be an efficient strategy with soybean from Center West, especially in Europe. Rather similar effects are observed for cumulative energy demand.. Conversely, eutrophication and acidification potential impacts are reduced by phase feeding and FU-AA addition in a rather similar way in all situations. In all situations, precision feeding, the only strategy that takes account for the between animal variability, is the most effective approach for reducing life cycle impact of pig fattening, whereas the potential of phase feeding program and FU-AA is dependent of soybean origin and geographical context of pig production, contrasting with many published results.

Keywords: pig, fattening, feeding strategy, environment, life cycle assessment

1. Introduction

The environmental impacts of pig production have come under increased debate in recent years, resulting in greater focus on identifying and mitigating the environmental degradation they may cause. The better adjustment of nutrient supply to the requirements of animals (Dourmad and Jondreville, 2007) may be a key factor in improving efficiency of nutrient retention, reducing excretion and consequently increasing the sustainability of pig production. In recent years, life cycle assessment (LCA) has been widely used in agriculture (Guinée et al., 2002) and several studies of swine production chains have been conducted (Nguyen et al., 2010, Garcia-Launay et al., 2014). Most of LCA studies about the effect of pig feeding strategies do not usually compare results in various geographical contexts and the question may be raised about possible interactions with the context of production. The differences in the human and natural resources between Brazil and France have led to the establishment of different supply chains, which may result in different environmental impacts of pig production. This offers contrasted situations in terms of raising animals, climatic conditions, diet formulation and type and origin of feed resources, which can be used to evaluate whether some feeding strategies may be environmentally friendly to one situation but not to another.

The aim of this study was thus to evaluate, using LCA, the effects of different feed formulation and feed distribution strategies on the environmental impacts of pig fattening in two contrasted contexts of pig production in South Brazil and West France.

2. Methods

System, goal and scope definition

The definition of system and subsystem boundaries was derived from Nguyen et al. (2010) and Garcia-Launay et al. (2014). The LCA considered the activity of pig fattening in two different geographical contexts (France and Brazil), including crop production, grain drying and processing, production and transport of feed ingredients, feed production at the feed factory, transport of the feed to the farm, growing to finishing pig production, and manure storage, transport and spreading. The pig

production system considered was typical of conventional growing-finishing pig farms located in Brittany and South Brazil. The environmental consequences of manure utilization were evaluated using system expansion as described by Nguyen et al. (2010). Thus, manure produced was assumed to substitute a certain amount of mineral fertilizers as described by Nguyen et al. (2010).

The assessment considered the growing-finishing pig production system, with four different feeding programs: two phases (2P), four phases (4P), daily multiphase (MP) and precision feeding (PR). These strategies were combined with three formulation scenarios built with least-cost formulated feeds: (i) feeds without feed-use amino acids (FU-AA) allowed (noAA), (ii) feeds with FU-AA (withAA) and fixed crude protein content corresponding to the usual local practical recommendations, and (iii) feeds with FU-AA without any minimum crude protein constraint (lowCP). For each scenario, two types of protein sources were considered: soybean meal only (SOY) or a mix of different protein sources (MIX) including soybean meal and meat and bone meal in Brazil and soybean meal, rapeseed meal and pea in France. Two hypotheses were also considered for the soybean origin: Centre West (CW) and South (SO) Brazil, which are contrasted in terms of recent deforestation. This resulted in a total of 96 scenarios: four feeding strategies tested in 12 situations of feed formulation, in two contexts of pig production.

Performance data from experimental studies in Brazil and France were used to adjust average animal profile parameters for growth and feed intake using InraPorc® software. These profiles were used to calculate, according to InraPorc®, the average nutritional requirement curves for each sex (females and castrated males), these requirements being used for diet formulation. Because variability between animals is known to affect the response of pig populations, we used the population version of InraPorc® (Brossard et al., 2014) to evaluate the animals' response to the feeding strategies. Simulations for 2000 pigs (50% females, 50% castrated males) were performed for each feeding scenario in each country, in order to determine animal performance, nutrient balance and excretion according to InraPorc®. A total of 192 000 pigs, i.e. 96 scenarios x 2000 pigs, were simulated on a daily basis and a database was built from these simulations.

Life cycle inventory

Resource use and emissions associated with the production and delivery of inputs for crop production (fertilizers, pesticides, tractor fuel and agricultural machinery) came from the Ecoinvent database version 3 (SimaPro LCA software 8.0, PRé Consultants). Energy use in the building for light, heating and ventilation was considered, but not the emissions and resources used for the construction of buildings. Veterinary medicines and hygiene products were also not included.

It was assumed that soybean was produced in Brazil for both geographical contexts, because most of the soybean meal imported to France comes from Brazil. For Brazilian crops, LCI came from de Alvarenga et al. (2012) for maize, and from Prudêncio da Silva et al. (2010) for soybean, taking into account the effect of land-use change on carbon release due to conversion of Brazilian forest to cropland for soybean (PAS2050, 2011). For French crops, LCA came from a national database developed by French research institutes with data on the environmental impacts of all main ingredients used in animal feeds (Wilfart et al., 2015). For the feed ingredients that are co-products, i.e. soybean meal, soybean oil, rapeseed meal, rapeseed oil, and wheat bran, the resource use and emissions were economically allocated. Data for FU-AA, phytase, salt, phosphate, sodium bicarbonate and limestone used in the diet came from Wilfart et al. (2015). Meat and bone meal was assumed to come from poultry slaughter. Impacts associated with broiler production were based on Prudêncio da Silva et al. (2014) and those associated with processing were based on Wilfart et al. (2015).

Emissions to air during swine production were estimated step-by-step for NH₃, N₂O, NO_x and CH₄. The CH₄ emissions from enteric fermentations were calculated according to feed digestible fiber content using equations from Rigolot et al. (2010). The CH₄ emissions from manure storage were calculated according to IPCC (2006) and Rigolot et al. (2010b) considering the average ambient temperature in each region (22°C for South Brazil and 13°C for West France). The NH₃ emissions from the building and during manure storage were calculated according to emission factors proposed by Rigolot et al. (2010), considering the effect of temperature. The N₂O and NO_x emissions from slurry storage were calculated according to IPCC (2006) and Dämmgen and Hutchings (2008), respectively. The amounts of nitrogen, P, Cu, Zn, K and organic matter excreted by the pigs were obtained from InraPorc® simulations. These data were used to calculate the amount of each element

available at field application. During field application, NH₃ emissions were based on Andersen et al. (2001), N₂O emissions on IPCC (2006) and NO_x emissions on Nemecek and Kägi (2007). The potential of NO₃ and PO₄ leaching came from Nguyen et al. (2011).

Characterization factors and functional unit

We based our analysis on the CML 2001 (baseline) method V3.02 as implemented in Simapro software, version 8.05 (PRé Consultant, 2014) and added the following categories: land occupation from CML 2001 (all categories) version 2.04 and total cumulative energy demand version 1.8 (non-renewable fossil + nuclear). Thus, we considered the potential impacts of pig production on climate change (CC, kg CO₂-eq.), eutrophication potential (EP, g PO₄-eq.), acidification potential (AP, g SO₂-eq.), terrestrial ecotoxicity (TE, g 1,4-DCB-eq.), cumulative energy demand (CED, MJ), and land occupation (LO, m².year). The CC was calculated according to the 100-year global warming potential factors in kg CO₂-eq. Impacts were calculated at the farm gate and the functional unit considered was one kg of body weight gain (BWG) over the fattening period.

Calculations and statistical analyzes

The LCA calculations were performed for each pig according to its individual performance and excretion from 70 days (with 30 kg BW on average) to an average BW of 115 kg at slaughter. These calculations were performed using a calculation model developed with SAS software (SAS Inst. Inc., Cary, NC). Performance responses and environmental impacts were subjected to variance analysis using the GLM procedure (SAS Inst. Inc., Cary, NC). The statistical model included effects of country, protein source, feeding phases and amino acid inclusion. Differences were considered significant if $P < 0.05$. Means were compared using the Tukey test. For LCA data, we also performed a variance analysis taking into account the effect of interactions between country and the other factors, in order to evaluate the behavior of environmental impacts among scenarios. All analyses were performed using SAS version 9.2 (SAS Inst. Inc., Cary, NC).

3. Results

Climate change

With soybean from SO, the average values for CC ranged between the feeding programs from 2.31 to 2.45 kg CO₂-eq. per kg BWG in Brazil and from 2.28 to 2.35 kg CO₂-eq. per kg BWG in France (Fig. 1A). When soybean meal from CW was used, CC values increased up to 2.75 to 2.96 kg CO₂-eq. per kg of BWG in Brazil and up to 2.61 to 2.89 kg CO₂-eq. per kg BWG in France. Depending on the feeding program, the lowest CC impact was reached for PR, both for soybean from SO and from CW. With soybean meal from SO, the highest impacts among the AA inclusion scenarios were observed for lowCP in Brazil and for withAA in France. Conversely, with soybean meal from CW, the highest impacts were observed for noAA in both countries. Independently of soybean origin and geographical context of pig production, SOY showed higher impacts than MIX scenarios.

The variation of CC impacts among scenarios was highest for noAA, intermediate for withAA and lowest for lowCP. Scenarios based on soybean meal from CW showed higher CC impacts than scenarios based on soybean from SO (Fig. 1A). Differences between protein sources (i.e. SOY and MIX) were less pronounced for soybean from SO compared to CW and were reduced when AA inclusion increased. There was a clear interaction between the soybean origin and AA inclusion scenario, in both countries. With soybean from CW, CC impact decreased when the incorporation of AA increased, the effect being more marked for SOY than for MIX protein source, whereas no effect or even the opposite was observed with soybean from SO. The effect of the feeding program on CC was mainly affected by the soybean origin and the level of AA inclusion. With soybean from CW, increasing the number of feeding phases and precision feeding reduced CC impact in all situations; however the magnitude of the effect decreased when AA inclusion increased. Precision feeding resulted in a reduced impact in all scenarios.

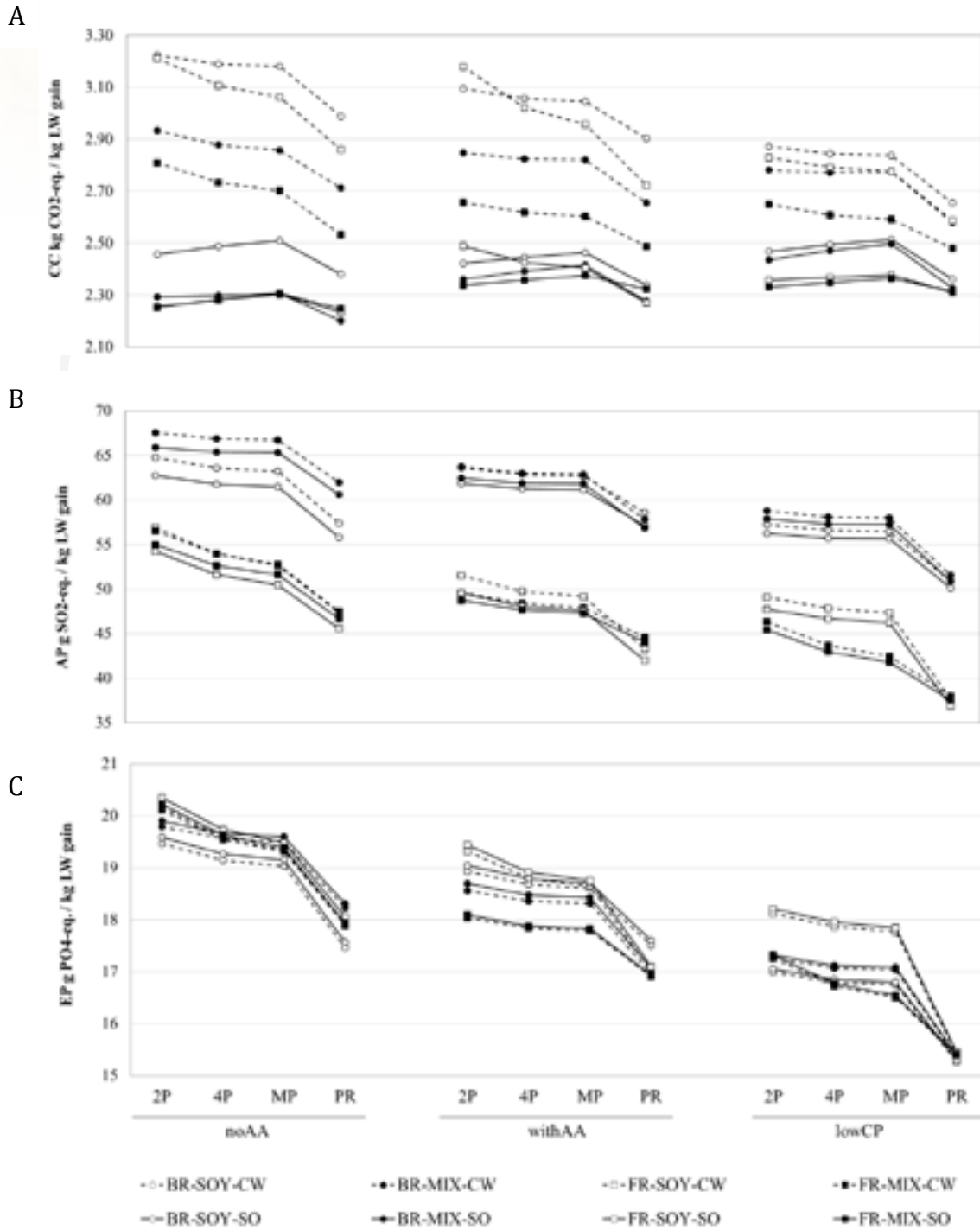


Figure 1. Interactions between effects of the feeding program, the use of amino acids and the soybean origin on environmental impact of climate change (A), acidification potential (B) and eutrophication potential (C) for Brazilian (BR) and French (FR) context of pig production. 2P, 2-phase, 4P, 4-phase, MP, multi-phase and PR, precision feeding programs. noAA, no amino acids, withAA, with amino acids, lowCP, without constraints in the crude protein content. SO, soybean from South region; CW, soybean from Centre-West Brazil. SOY, soybean meal as sole protein source, MIX, diversified protein sources.

Acidification potential

With soybean from SO, the values for AP ranged between the feeding programs from 55.3 to 61.2 g SO₂-eq. per kg BWG in Brazil and from 42.1 to 50.1 g SO₂-eq. per kg BWG in France (Fig. 1B). With soybean from CW the values were only slightly increased up to 56.4 to 62.6 g SO₂-eq. per kg BWG in Brazil and up to 43.1 to 51.7 g SO₂-eq. per kg BWG in France. The lowest AP impact was reached for PR, as well for soybean from SO and CW. The highest AP impacts among the AA inclusion scenarios were observed for noAA in both countries, and the lowest for lowCP. For the French context, SOY showed 1.7% higher impacts than MIX protein source. Conversely, for the Brazilian context MIX showed 2.9% higher AP impact than SOY. The variation of AP impact among scenarios (Fig. 1B) was highest for noAA, intermediate for lowCP and lowest for withAA. In all scenarios of AA inclusion and whatever the feeding program or country, scenarios based on soybean meal from CW showed higher AP impacts than scenarios based on soybean from SO (Fig. 1B). Whatever the soybean origin, AP impact decreased when the incorporation of AA increased. The effect of the feeding program on AP was affected by the level of inclusion of AA. Increasing the number of feeding phases and precision feeding reduced the AP impact in all situations; however the magnitude of the effect decreased when AA inclusion increased. In all scenarios of feed formulation, whatever the soybean meal origin and the protein source, increasing the number of feeding phases from 2P to PR, decreased the AP impact by about 10% in Brazil and 16% in France.

Eutrophication potential

The values for EP ranged between the feeding programs from 16.3 to 18.1 g PO₄-eq. per kg BWG in Brazil and from 16.3 to 18.4 g PO₄-eq. per kg of BWG in France, with similar results for soybean from SO and CW (Fig. 1C). According to the feeding program, the lowest EP impact was obtained for PR in both countries (mean of 16.3 g PO₄-eq. per kg BWG) and the highest for 2P (mean of 18.2 g PO₄-eq. per kg BWG). The highest EP impacts among AA inclusion scenarios were observed for noAA scenarios, and the lowest for lowCP scenarios, in both countries. Independently of soybean origin, SOY showed higher EP impacts than MIX in France, whereas the opposite was found in Brazil. Differences between protein sources (i.e. SOY and MIX) increased when AA inclusion increased and were less pronounced in Brazil than in France. With both protein sources, EP impact decreased when the incorporation of AA increased, the effect being more marked for France than for Brazil. In the Brazilian context, diets based on SOY showed lower impacts than those based on MIX in noAA and lowCP scenarios, whereas the opposite was observed in the withAA scenario. Increasing the number of feeding phases and precision feeding reduced the EP impact in all situations, mainly when moving from MP to PR program. In all scenarios of feed formulation, whatever the soybean origin and the protein source, increasing the number of feeding phases, from 2P to PR, reduced the EP impact by about 8% in Brazil and 11% in France.

Cumulative energy demand

With soybean from SO, the values for CED ranged between the feeding programs from 12.8 to 13.4 MJ-eq. per kg of BWG in Brazil and from 11.7 to 12.5 MJ-eq. per kg of BWG in France (Fig. 2A). With soybean from CW, the values increased up to 15.0 to 16.1 MJ-eq. per kg BWG in Brazil and up to 12.9 to 14.4 MJ-eq. per kg of BWG in France. The lowest CED impact was reached for PR, both for soybean from SO and CW. On average, the highest CED impacts among the AA inclusion scenarios were observed for lowCP in both countries, and the lowest for noAA. On average, SOY scenarios showed higher CED impacts than MIX scenarios of protein source. In all scenarios of AA inclusion and whatever the feeding program or country, scenarios based on soybean meal from CW showed higher CED impacts than scenarios based on soybean from SO (Fig. 2A). The effect of the feeding program on CED was affected by the soybean origin and the level of AA inclusion. With soybean meal from CW, increasing the number of feeding phases and precision feeding reduced the CED impact by about 8.6% in all AA inclusions scenarios. Conversely, no effect or even a slight increase was observed with soybean from SO.

Terrestrial ecotoxicity

The values for TE ranged between the feeding programs from 8.45 to 9.19 g 1,4-DCB-eq. per kg BWG in Brazil, and from 13.1 to 14.2 g 1,4-DCB-eq per kg BWG in France (Fig 2B). According to the feeding program, the lowest TE impact was reached for PR, whereas the effect of phase feeding

was very limited. In Brazil variability of TE was reduced when AA inclusion increased, without difference in mean values, whereas in France TE tended to decrease when AA inclusion increased. However the effects were rather limited. Independently of soybean origin and geographical context of pig production, SOY showed higher impacts than MIX scenarios but differences between protein sources were much higher in the French context compared to the Brazilian one.

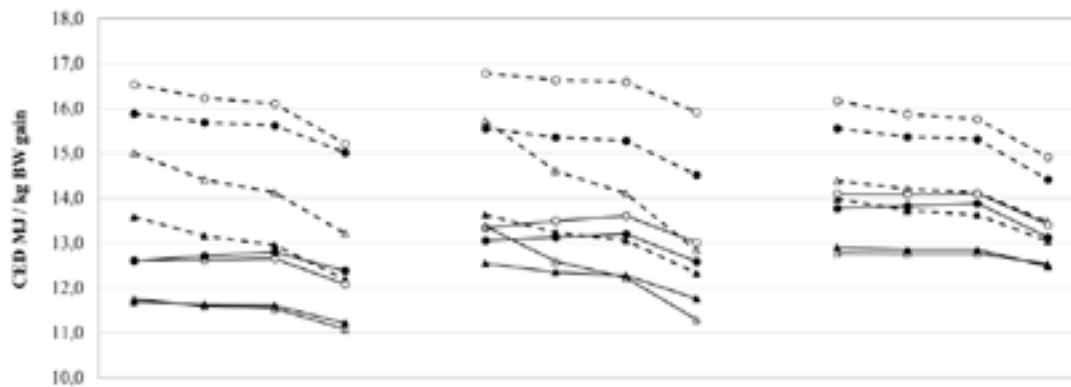


Figure 2. Interactions between effects of the feeding program, the use of amino acids and the soybean origin on environmental impact of cumulative energy demand (A), terrestrial ecotoxicity (B) and land occupation (C) for Brazilian (BR) and French (FR) context of pig production. 2P, 2-phase, 4P, 4-phase, MP, multi-phase and PR precision feeding programs. noAA, no amino acids, withAA, with amino acids, lowCP, without constraints in the crude protein content. SO, soybean from South region; CW, soybean from Centre-West Brazil. SOY, soybean meal as sole protein source, MIX, diversified protein sources.

There was a clear interaction between country and AA inclusion scenarios for TE. In the French context, TE impact decreased by 23% when the incorporation of AA increased, whatever the soybean origin and protein source, whereas no effect or even the opposite was observed in the Brazilian context. For France, increasing the number of feeding phases and precision feeding reduced the TE impact in all AA inclusion scenarios; the magnitude of the effect decreasing when AA inclusion increased. Conversely, for Brazil, increasing the number of feeding phases tended to slightly increase the TE impact in all situations. Precision feeding resulted in a reduced TE impact in all scenarios of feed formulation, whatever the protein source and soybean origin. Compared to 2P, PR decreased the TE impact by about 21% in France and 8% in Brazil.

Land occupation

The values for LO ranged between the feeding programs from 2.33 to 2.52 m².year per kg BWG in Brazil and from 3.89 to 4.05 m².year per kg BWG in France, with similar results for both origins of soybean (Fig. 2C). According to the feeding program, the lowest LO impact was reached for PR, both for soybean from SO and from CW. The highest impacts among the AA inclusion scenarios were observed for the noAA inclusion scenario in both countries, whatever the origin of soybean meal. Independently of soybean origin and geographical context of pig production, SOY showed higher LO impacts than MIX scenarios. There was an interaction between the protein source and AA inclusion scenario in the French context. In the noAA scenario, SOY showed 2.1% lower LO impact than MIX, whereas when the incorporation of AA increased, SOY showed 5.7% higher impact than MIX, whatever the origin of the soybean meal. Precision feeding resulted in a reduced impact in all scenarios.

4. Discussion

With the hypothesis that 70% of the soybean was from CW and 30% from SO, Garcia-Launay et al. (2014) calculated lower CC impact in France for an MP program compared to a 2P feeding

program. This was also the case in our study, in both countries, when soybean meal came from CW, whereas when it came from SO, the MP program resulted in a higher CC impact than 2P. This indicated that the effect of phase feeding on CC may depend on the origin of the soybean. In our study and in the studies by Eriksson et al. (2005) and Garcia-Launay et al. (2014), diets based on soybean meal only showed higher impact than diets based on more diversified protein sources. The lower CC impact obtained with these diets was related in Brazil to the use of meat and bone meal, a co-product with low CC impact. For French conditions, rapeseed meal and pea were not associated with any deforestation process, since this process occurred many centuries ago and, thus, was not taken into account in the LCA evaluation.

Variation of CC impact among scenarios was clearly reduced with the increased inclusion of FU-AA, indicating that the effect of feeding programs on CC was more pronounced when no FU-AA were included. A more pronounced effect of feeding program was also observed when soybean was from CW and especially when it was the sole protein source. In this situation, because of the high CC impact of CW soybean meal, increasing the number of feeding phases and AA inclusion was very efficient in reducing CO₂-eq. emissions. This was not the case when the soybean was from SO. In Brazil with soybean from SO, CC impact tended even to rise when the number of feeding sequences increased and it decreased only in the case of precision feeding, whereas in France it always decreased. The possibility of reducing CED impact by increasing the number of feeding phases was confirmed for diets based on soybean meal from CW, but not for soybean from SO. Precision feeding only resulted in reduced CED impact in that situation.

Since both nitrogen and P contribute to eutrophication, and nitrogen contributes to acidification by ammonia emissions (Guinée et al., 2002), the AP and EP impacts were reduced in both countries by increasing the number of feeding phases and with the incorporation of FU-AA. This was not surprising because both strategies reduced nitrogen and P excretion and, consequently, also reduced the NH₃ emissions from animal housing and manure management and field application.

Feeding strategies affected TE impacts only when high impact feed ingredients were used. For this reason, the incorporation of FU-AA, the increase in the number of feeding phases and precision feeding reduced TE impact in the French context, but not in Brazil. The lower TE impact in Brazil was associated with the low TE impact of maize production, which represented more than 70% of feed composition in Brazilian formulations.

5. Conclusions

The results of this study indicate that precision feeding would be the most effective approach for reducing the life cycle impact of pig fattening, whereas the potential of multi-phase feeding programs depends on the impact considered, soybean origin and the geographical context of pig production. The interest of phase feeding and incorporation of FU-AA for reducing CC impact is limited in South America with soybean from South Brazil, whereas it appears to be an efficient strategy with soybean from Center West, especially in Europe. Conversely, potential eutrophication and acidification impacts are largely reduced by phase feeding and FU-AA addition in a rather similar way in all situations.

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159. Integrating Pigs and Crops for Environmental Benefit - A Swedish Case Study

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ABSTRACT

Meat consumption is still increasing on a global scale. This calls for environmentally sound animal production, as the agricultural phase in the life cycle of meat is by far the most important. Hence, the question arises how an agricultural production system has to be designed to meet these expectations. The objective of this study was to analyse an alternative pig production relying on locally produced protein feed with a sustained meat output and simultaneously not compromising the overall output for human consumption. Finally, we aimed at drawing comparisons to commonly applied pig production in a mixed farming context with regard to the environmental impacts.

For our case study, we modelled two farming scenarios situated in the county V stra G taland, in southwestern Sweden. The baseline scenario S0 consisted of a pig-fattening farm combined with cereal cropping, relying heavily on purchased protein feeds, i.e. soybean meal from Brazil (current practice in Sweden). The alternative scenario (S1) was designed by enlarging the farm area by a factor of 1.5 taken from fallow land to maintain the pig output and producing the majority of the protein feeds (i.e. horse bean, oilseed rape) on-farm respecting a sound crop rotation. To assess the environmental impacts, life cycle methodology developed within the framework of EU FP7 project CANTOGETHER was applied. The system boundary was set at farm gate. We assessed two functional units: i) the livelihood preservation function, and ii) the productive function. Detailed analyses of global warming potential, aquatic eutrophication with nitrogen, and terrestrial ecotoxicity (pesticides) revealed a reduction of the environmental burden per ha farmland. The overall assessment showed that there was a reduction of the environmental impact for all impact categories with respect to both, the livelihood preservation and productive function.

The targeted environmental effects were achieved due to locally produced protein feed and improved crop rotation with reduced pesticide use. However, an important factor for the results was the fact that the alternative scenario comprised an area 1.5 times larger than in the baseline scenario. This had a "dilution effect" on environmental impacts per ha farmland. As the output for human consumption (in MJ digestible energy) increased disproportionately, the impacts with regard to the productive function were even more reduced.

In conclusion, the substitution of imported soybean meal with home-grown protein sources on a mixed pig-fattening farm reduced the environmental impacts of the farm itself and of the food produced on this farm. However, we used a farm modelling approach, which relied on certain preconditions, like the availability of additional land for protein crop production or a change regarding the distribution of the regionally grown crops and their area of cultivation. The applicability of this approach on real pig-fattening units therefore needs to be evaluated taking into consideration the locally given conditions.

Keywords: Life cycle assessment, pig production, mixed farming, environmental impacts, meat, European grain legumes.

1. Introduction

In agriculture, there is a rising political and social expectation of environmentally sound production. Especially meat consumption and along with it animal production is among consumers increasingly seen not only in the light of animal welfare, but also with respect to its environmental impact. However, meat demand is still high, globally with an annual growth rate of 3% for the period 1990 until 2009 (Henchion et al., 2014 based on FAO, 2014). This calls for agricultural production systems with an environmentally improved production of meat. But what options do we have?

Several publications have shown the importance of the agricultural phase in the life cycle of meat production and here feedstuffs are key (e.g. de Vries & de Boer 2010; Gerber et al 2013). With regard to feeding, soybean meal is an attractive feeding component especially for monogastric animals such as pigs and poultry due to its favourable composition of essential amino acids, but also due to the favourable ratio between protein and energy content. Since most production of soybeans for the world market occurs in Brazil and Argentina, animal fattening farmers rely on imports of soybean meal stemming from South America. This has several consequences: First, a less diverse crop rotation with a stronger focus on cereals and hence more specialised farming in Europe with drawbacks regarding pests and plant diseases. Second, a specialised soybean cropping in South America along with land use change and deforestation. This leads to a loss of biodiversity and a release of previously stocked carbon generating effects on climate change. On the other hand, locally produced protein crops when maintaining the animal production at the current level require suitable areas for cropping and have to comply with crop rotation rules (e.g. oilseed rape only every fourth year).

Within this setting, the goal of this study was to analyse an alternative pig production relying on locally produced protein feed with a sustained meat output as well as not compromising the overall output for human consumption. We aimed at drawing comparisons with regard to the environmental impacts between this alternative and commonly applied pig production in a mixed farming context.

2. Methods

For our case study, we modelled two farming scenarios situated in the county Västra Götaland, in southwestern Sweden. It is an important region for pig fattening in Sweden, but also suited for arable cropping either in combination with pig fattening or on specialised crop farms. The main cereals grown in the county of Västra Götaland in 2012 were oats, spring barley, and winter and spring wheat. The other main uses of arable land in 2012 were temporary grassland, fallow land, oilseed rape, peas and horse beans. There is a cereal surplus in the area and an availability of cropland currently out of production. Actual production patterns for pig fattening consist of high reliance on imported protein feed (i.e. soybean meal) and regionally produced cereals.

The baseline scenario (S0) was a pig-fattening farm combined with cereal cropping, relying heavily on purchased protein feeds, i.e. soybean meal from Brazil (current practice in Sweden). The crops, of which 80% were sold, were wheat, barley, and oat. The alternative scenario (S1) consisted of enlarging the farm area by a factor of 1.5 taken from fallow land to maintain the pig output and to grow the majority of the protein feeds (i.e. horse bean, oilseed rape) on-farm respecting a sound crop rotation. The surpluses of crops were sold, i.e. mostly cereals. Oilseed rape was sold for extraction into oil and meal. The scenarios, i.e. baseline scenario S0 and alternative scenario S1, were developed based on different sources of Swedish statistical data (e.g. Statistics Sweden, 2013; Pigwin, 2015; County Board of Västra Götaland, 2010; Swedish Board of Agriculture, 2015).

The system boundary was set at farm gate. Activities beyond farm gate (downstream) were outside of the study's scope. The temporal system boundary is one calendar year for permanent grassland, animals, and for all production means. For arable crops and ley, the temporal boundary is from the harvest of the previous main crop until the harvest of the actual main crop.

With regard to the goal of the study, we applied two functional units: i) the function of land cultivation expressed in ha utilized agricultural area and year (ha UAA*a), and ii) the productive function expressed in megajoule digestible energy (MJ DE). Here at farm level this means that all agricultural outputs (food and feed products) were converted into energy, which is digestible by humans. Arable crops, which were sold for animal production (e.g. barley, oat, wheat), were also considered. This was done using standardised production scenarios and conversion factors yielding values of digestible energy from the animals fed by the sold arable crops.

The production inventories were calculated with farm LCA tools developed for the EU FP7 project CANTOGETHER (Teuscher et al., 2014). Life cycle impact assessment was performed with SimaPro 7.3 (PRé Consultants, Amersfoort, The Netherlands, 2016). The life cycle inventories employed in this study originate from the ecoinvent data base v2.2 (ecoinvent Centre, 2010) and from the SALCA database (Nemecek & Kägi, 2007).

Table 1 shows the assessed impact categories and resource-use.

Table 1: List of the assessed impact categories and resource use with indication of the corresponding methods.

Impact category/ Resource-use	Method
<i>Resources related impacts</i>	
Non-renewable energy demand	ecoinvent, Hirschier et al., 2010
Global warming potential (100a)	IPCC, 2007
Ozone formation (Human)	EDIP03, Hauschild & Potting, 2005
Ozone formation (Vegetation)	EDIP03, Hauschild & Potting, 2005
Land competition	CML01, Guinée et al., 2001
Deforestation	at life cycle inventory stage
Use of phosphorus resources	at life cycle inventory stage
Use of potassium resources	at life cycle inventory stage
Total water use (blue water)	at life cycle inventory stage
<i>Nutrients related impacts</i>	

Aquatic eutrophication N	EDIP03, Hauschild & Potting, 2005
Terrestrial eutrophication	EDIP03, Hauschild & Potting, 2005
Acidification	EDIP03, Hauschild & Potting, 2005
Pollutants related impacts	
Aquatic ecotoxicity (100a), CML	CML01, Guinée et al., 2001
Terrestrial ecotoxicity, pesticides (100a), CML	CML01, Guinée et al., 2001
Human toxicity (100a), CML	CML01, Guinée et al., 2001

In order to identify the main contributors to each environmental impact caused by agricultural production, we thematically grouped the different inputs and emissions into eleven input groups allowing a more detailed analysis: ‘Buildings and equipment’, ‘Machinery’, ‘Energy carrier’, ‘Fertilizers and field emissions’, ‘Pesticides’, ‘Purchased seeds’, ‘Feedstuffs, concentrates (purchased)’, ‘Feedstuffs, roughage (purchased)’, ‘Purchased animals’, ‘Animal husbandry’, and ‘Other inputs’. This grouping was chosen to allow a generic farm LCA tool in the frame of the CANTOGETHER project (Teuscher et al., 2014).

Our assessment concept consisted of two steps: First, we assessed three impact categories more closely from three different areas of impact with respect to hectare utilised agricultural area to give information on the environmental performance of each of the respective areas. For the resource related impacts, we analysed the global warming potential (GWP), ii) the nutrients related impacts are represented by aquatic eutrophication N, and iii) terrestrial ecotoxicity (pesticides) sheds a light on the area of pollutants related impacts. We chose the pesticide part of terrestrial ecotoxicity because one of the targets for the alternative scenario S1 was reducing the impacts of pesticides through a more diversified crop rotation. Second, we performed an overall appraisal of all other assessed impact categories per hectare utilised agricultural area and per megajoule digestible energy. The latter allowed identifying possible trade-offs between the two functions, i.e. livelihood preservation and productive function.

When assessing scenarios on farm level, we need to evaluate significance and relevance of the differences, as in LCA calculations there are various sources of uncertainty. The uncertainty of many of the parameters is not known, hence it is not feasible to perform a full analysis of statistical significance (Nemecek et al., 2011). In consequence, the assessment of the results was done in classes based on the statistical variance of environmental impact indicators from a cropping system experiment and expert knowledge (see Table 2; and Nemecek et al., 2005).

Table 2: Assessment classes for the differences between two scenarios. Each value is expressed as the percentage of a reference. Here it is the baseline scenario S0.

Assessment class	Resources related impacts	Nutrients related impacts	Pollutants related impacts
Very favourable	<77%	<63%	<53%
Favourable	77-91%	63-83%	53-77%
Similar	91-110%	83-120%	77-130%
Unfavourable	110-130%	120-160%	130-190%
Very unfavourable	>130%	>160%	>190%

3. Results

Global warming potential (GWP): In the alternative scenario (S1), the global warming potential was reduced by 23% (Figure 1) as compared to the baseline scenario (S0) per hectare utilised agricultural area, which is considered to be a very favourable effect on this environmental impact.

The most important input groups contributing to the GWP were ‘Fertilisers and field emissions’ and ‘Animal husbandry’ in both, the baseline and the alternative scenario. In the baseline scenario, the input group ‘Feedstuffs, concentrates’ was also of importance.

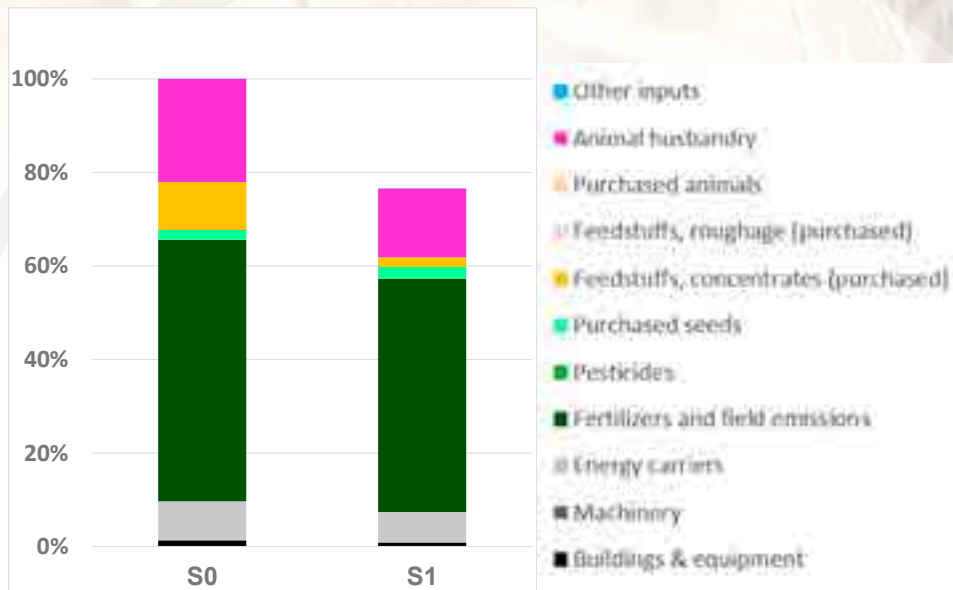


Figure 1: GWP 100a (kg CO₂ eq.) per ha utilised agricultural area (UAA) * year divided into the contributions of the different input groups. S0 = Baseline scenario; S1 = Alternative scenario. S1 is expressed in % of S0 (normalised at 100%).

The contribution of the input group ‘Feedstuff concentrates’ decreased by 8 percentage points in the alternative scenario as compared to the baseline scenario (Figure 1). This decrease is related to the targeted novelty in the alternative scenario, the replacement of imported soybean meal with locally produced protein feeds. In the baseline scenario, GHG emissions from purchased feedstuffs were related to the transport of soybeans from Brazil and soybean production itself including the effects from deforestation and land change on GWP. Replacing most of the soybean meal with locally produced feed therefore resulted in a reduction of GHG emissions.

The impact of the input group ‘Animal husbandry’ was also reduced by 33%, because in the alternative scenario, the same numbers of pigs were produced on an area, which was 1.5 times larger than in the baseline scenario. This had a “dilution effect” on the impact of ‘Animal husbandry’ per ha farmland.

Finally, the contribution of the input group ‘Fertilizers and field emissions’ decreased by 11% in the alternative scenario. While the mineral N-fertiliser rates were equal in the baseline and the alternative scenario, the amount of pig slurry applied per ha was reduced by about 50% in the alternative scenario (19.8 m³ in S0 vs. 10.6 kg in S1). This decrease in pig slurry application explained the reduced impact of the input group ‘Fertilizers and field emissions’ on GWP and was owed again to the extension of the farm area and the resulting “dilution effect”.

Aquatic eutrophication N: In the alternative scenario (S1), the aquatic eutrophication N was reduced by 21% as compared to the baseline scenario (S0) per hectare utilised agricultural area (Figure 2), which is considered a favourable effect on this environmental impact.

The most important input groups contributing to aquatic eutrophication N were the same in both scenarios, namely ‘Fertilisers and field emissions’, ‘Purchased seeds’ (main contributor wheat grain seeds: 43% in S0 and 33% in S1), ‘Feedstuffs, concentrates’ and ‘Animal husbandry’.

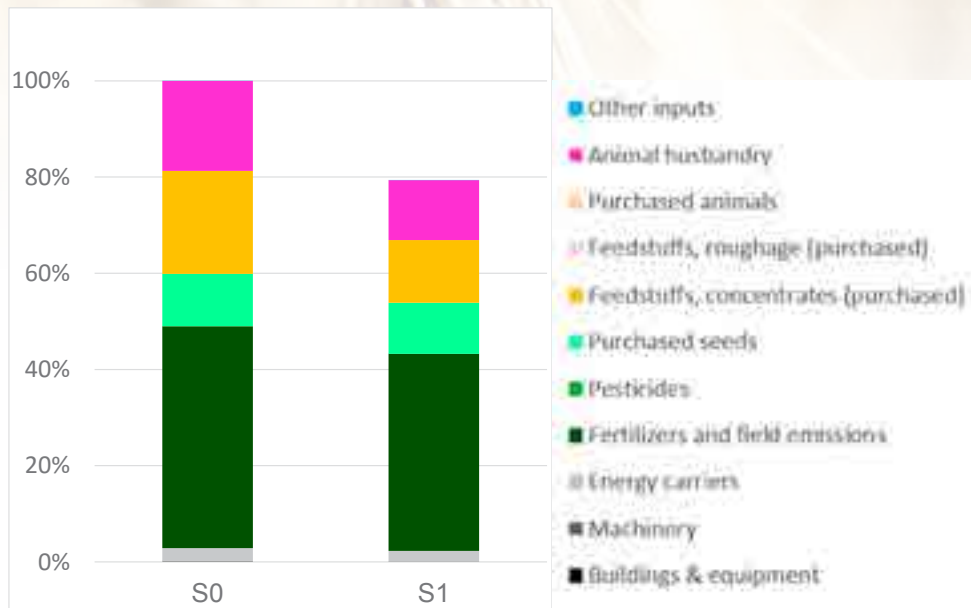


Figure 2: Aquatic eutrophication N (kg N) per ha utilised agricultural area (UAA) * year divided into the contributions of the different input groups. S0 = Baseline scenario; S1 = Alternative scenario. S1 is expressed in % of S0 (normalised at 100%).

As for GWP, the most important decrease in aquatic eutrophication N was related to the reduced feedstuff import and the reduction for the input group ‘Animal husbandry’ in the alternative scenario. The contribution of the input group ‘Feedstuff concentrates’ decreased by 39%. For ‘Animal husbandry’, with a reduction of 33%, the same is valid as for the results for GWP: A “dilution effect” is seen, due to the enlargement of the farm area in S1.

Terrestrial ecotoxicity (pesticides): In the alternative scenario (S1) terrestrial ecotoxicity was reduced by 24% as compared to the baseline scenario (S0) per hectare utilised agricultural area (Figure 3), which is considered a favourable effect on this environmental impact.

In both scenarios, the most important input groups contributing to terrestrial ecotoxicity (pesticides) were ‘Pesticides’, ‘Purchased seeds’ and ‘Feedstuffs, concentrates’.

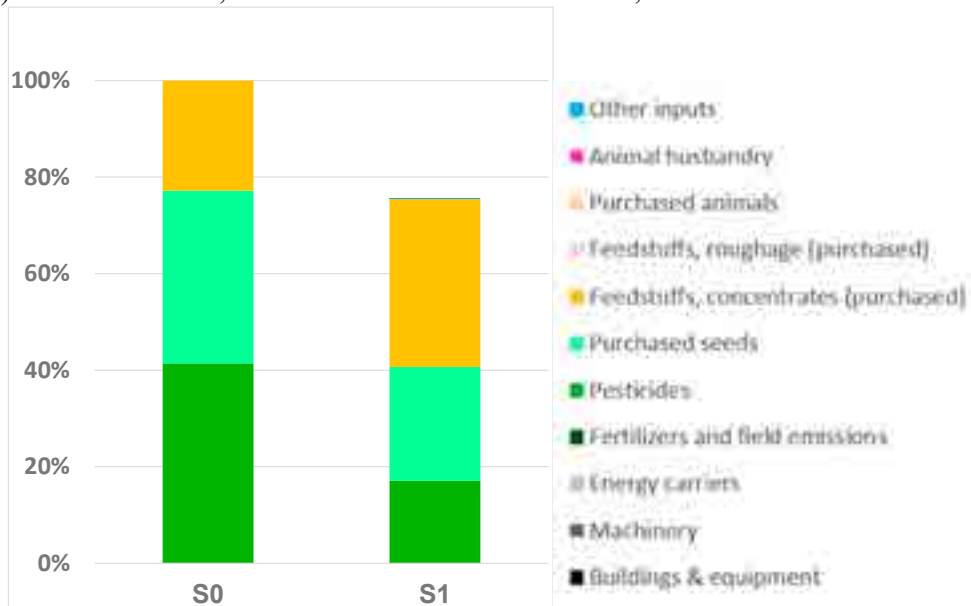


Figure 3: Terrestrial ecotoxicity, pesticides (kg 1,4-DB-eq.) per ha utilised agricultural area (UAA) * year divided into the contributions of the different input groups. S0 = Baseline scenario; S1 = Alternative scenario. S1 is expressed in % of S0 (normalised at 100%).

For terrestrial ecotoxicity (pesticides), there was a shift in the relative contributions of the different input groups from the baseline to the alternative scenario. The contribution of ‘Pesticides’ (-59%) and ‘Purchased seeds’ (-34%) decreased whereas the contribution of ‘Feedstuffs, concentrates’ increased (+53%) in the alternative scenario.

The decreased contribution of ‘Pesticides’ applied on the farm is explained by the reduced use of the active ingredient Diflufenican (a herbicide) in the alternative scenario. As aimed by the strategy of the case study, the improved crop rotation resulted in lower pesticide use.

The increased contribution of ‘Feedstuffs, concentrates’ was related to the different feeds that were imported in the baseline and alternative scenario and the pesticides applied on these crops. For soybean meal, the main feed imported in the baseline scenario, other pesticides were used than for wheat bran, which was the main feed imported in the alternative scenario. Isoproturon (a herbicide) used in wheat production had a high impact on terrestrial ecotoxicity (pesticides), which explains the increased contribution of the input group ‘Feedstuffs, concentrates’.

The comparison of the alternative scenario (S1) with the baseline scenario (S0) for the livelihood preservation function, i.e. per hectare utilised agricultural area, shows that all assessed impact categories had a reduction of the environmental burden, which is categorised as favourable or even very favourable with the exception of land competition and aquatic ecotoxicity judged as similar (Table 3).

Table 3: Summary of the LCA results of the case study Västra Götaland for all calculated impact categories. Results are expressed as S1 in percent of S0 for two functional units, ha Utilised Agricultural Area*year (ha UAA*a) and MJ Digestible Energy (MJ DE). ++ is very favourable, + favourable, o similar, - unfavourable, and -- very unfavourable, according to Table 2.

Impact categories	Units	S1 (ha UAA*a)	S1 (MJ DE)
<i>Resources related impacts</i>			
Non-renewable energy demand, fossil and nuclear	MJ eq	83% +	55% ++
Global warming potential (100a)	kg CO ₂ eq	77% ++	51% ++
Ozone formation (Human)	Person*ppm*h	73% ++	48% ++
Ozone formation (Vegetation)	m ² *ppm*h	74% ++	49% ++
Land competition	m ² *a	92% o	61% ++
Deforestation	m ²	5% ++	3% ++
Use of phosphorus resources	kg	68% ++	45% ++
Use of potassium resources	kg	48% ++	32% ++
Total water use (blue water)	m ³	77% ++	51% ++
<i>Nutrients related impacts</i>			
Aquatic eutrophication N	kg N	79% +	53% ++
Terrestrial eutrophication	m ²	77% +	51% ++
Acidification, GLO	m ²	78% +	51% ++
<i>Pollutants related impacts</i>			
Aquatic ecotoxicity (100a), CML, total	kg 1,4-DB eq	79% o	53% +
Terrestrial ecotoxicity, pesticides (100a), CML	kg 1,4-DB eq	76% +	50% ++
Human toxicity (100a), CML	kg 1,4-DB eq	49% ++	32% ++

With regard to the productive function, i.e. per MJ digestible energy for human consumption, all assessed environmental impacts were reduced in the alternative scenario compared to the baseline scenario (Table 3). The majority of results was categorised as very favourable for S1 compared to S0. Only in the case of aquatic ecotoxicity the classification of the result was favourable.

4. Discussion

In this case study, action within animal production was taken at feed production: purchased soybean meal from South America was replaced to a very large part with grain legumes and rapeseed

meal from oil seed rape cultivated on the own farmland. This led to the expected and desired reduced environmental effects with regard to global warming potential, aquatic eutrophication N, and terrestrial ecotoxicity (pesticides). However, due to its high content of protein and energy, replacing soybean meal with other protein feeds, here horse beans and rapeseed meal, leads to changes in the composition of the pig feed ratio, most important here, a higher share of wheat bran in the alternative scenario (S1).

This is an internalisation strategy with regard to the protein feed. It proved to be very successful in terms of reducing the environmental impact without having trade-offs as to the preservation of livelihood.

The “dilution effect” from producing the same quantity of pigs on a 1.5 times larger area, would normally lead to reverted results for the analysis of the productive function, i.e. per MJ digestible energy for human consumption. However, owing to the additional production of crops in the alternative scenario, which increased disproportionate (+126 percentage points) to the utilised agricultural area (+50 percentage points), we also observed a clear improvement with regard to the MJ digestible energy, which was even more pronounced than the improvement for the livelihood preservation function.

Still, it has to be pointed out that these favourable results were only possible due to the extension of the farm area, a condition, which in many regions of Europe might be difficult to achieve. Such an approach is feasible if there is available arable land currently out of production (e.g. due to set aside policy). Alternatively, a change in arable production resulting in less cereal production for sale, i.e. the export market and using the freed area for the production of protein and oil seed crops would also allow to apply this strategy. However, the latter case would mean a shift in the global equilibrium resulting in a lack of the reduced crops and a surplus of the increased crops on a global scale.

In the present study, we used a farm modelling approach, which relied on certain preconditions, like the availability of additional land for protein crop production. The applicability of this approach on real pig-fattening units therefore needs to be evaluated taking into consideration the locally given conditions.

5. Conclusions

In this case study, we found that the combination of adjusting the crop rotation by introducing a grain legume crop, i.e. horse bean, and an oil crop, i.e. oilseed rape, resulted in the desired environmental effects for the livelihood preservation function. Moreover, opting for a strategy with an increase of the cropping area, where the additional land serves to cultivate protein feed – allowing a sound crop rotation – as well as crops for direct human consumption led to environmentally positive effects, also with respect to the productive function.

Overall, this internalisation strategy resulted in convincing environmental improvements with respect to both, the livelihood preservation and the productive function.

In conclusion, this approach can be recommended to lessen the environmental burden of pig fattening. However, it is only practical in two conditions: i) where set aside arable land can be taken into crop production or ii) where on a regional or national level changes with respect to the choice of the cultivated crops, e.g. reducing the cereals’ cropping while increasing the area of the protein crops, are executed. The latter however, has an impact on the global equilibrium of both, the reduced and increased crops.

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28. Environmental footprint of milk and meat from the French cattle sector: improvements since 1990 and future trends until 2035

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ABSTRACT

French cattle farming is committing to take part to national and European greenhouse gases (GHG) mitigation targets. This study combines macro-economics and life cycle assessment to estimate the climate contributions of the different methods of producing milk and meat at national level and to propose improvement strategies for the future. Analyses of former production systems, livestock population and agricultural practices allowed us to specify GHG emissions, energy consumption and corresponding footprints from 1990 to 2010. Various coherent, plausible and contrasted economic scenarios have been chosen to explore possible evolutions up to the year 2035. From 1990 to 2010, the cattle sector reduced its emissions and energy consumptions (respectively by -10.6% and -22%). Origin of such a reduction mainly is the decrease in cattle population: improvements in dairy productivity were followed by a decrease in the number of dairy cows, only partially balanced by an increase in the suckler cows' population. Farmers' progress in fertilization management and energy savings also contributed to the overall reduction. This bred a 20%-reduction of the carbon footprint (CF) of milk but a 5%-increase in the CF of meat, due to changes in animal products (less dairy cows, more animals from suckler herd, with allocation of the impacts). On the basis of an underlying projection of milk and meat productions in 2035, the future trend would be a stabilization of GHG emissions (+0.5%) and a decrease in energy consumption (-13%) between 2010 and 2035. The CF of milk would reach 0.94 kg CO₂eq/kg FPCM and CF of beef 14 kg CO₂eq/kg LW. Adoption of additional mitigation techniques would lead to improve both CF of milk and meat by -5% and -13%. Other scenarios explore contrasted situations on production level, as well as on the ways to produce milk and meat. The results show that mitigation strategies do exist to optimize milk and beef footprints at farm gate. However, the national level of GHG emissions and energy consumption will be mainly directed by economic context and food demand.

Keywords: climate change, energy, cattle, national level, prospective

1. Introduction

With a contribution of 12.6% to the French greenhouse gas (GHG) balance (Dollé et al., 2015), cattle farming is committing to take part to national and European mitigation objectives. This observation is related to the importance of this sector at national level: 1st producer of beef and 2nd producer of milk in Europe, cattle farms directly use about 40% of the French agricultural area (Perrot et al., 2013) and provide 433 000 direct and indirect employments (Lang et al., 2015).

The French Low Carbon National Strategy launched by the government at the end of 2015, targets a 12%-reduction of the agricultural sector GHG emissions in 2028 in comparison with 2013 and a 50%-reduction in 2050 compared to 1990. In this context, a recent analysis of different scenarios combining GHG mitigation options showed that French agriculture and forestry could reduce their GHG by 20% in 2035 (Martin et al., 2015). The scenario approach is a widely used method to explore a highly uncertain future for agriculture (Abildtrup et al., 2006; Audsley et al., 2006; Mandryk et al., 2012) by describing coherent and plausible future states of the world. In the bovine sector, GHG emissions in the next 20 years will depend on numerous factors including the use of GHG mitigation technics, but also other technological improvements, production organizations, global demand, consumers' behavior and public policy.

The Gesebov project investigated the joint evolution of the dairy and beef cattle sectors up to the year 2035, through contrasted prospective scenarios, and its associated level of GHG emissions and energy consumption in a life cycle perspective, both at individual farms and national levels. Since GHG emissions of the bovine sector are firstly explained by the bovine inventory (Casey and Holden, 2006) and secondly by the way meat and milk are produced (Monteny et al., 2006; Johnson et al., 2007), Gesebov scenarios were specifically elaborated to be contrasted in terms of volume of milk and beef produced and technology of bovine production.

This paper focuses on the national level, providing the state of national impacts from 1990 to 2035 and evolution of environmental efficiency at product level (footprints). The aim is to imagine how beef and milk could be produced in the future, in different economic contexts, and to assess how far the simulated scenarios are coping with climate change mitigation objectives, food demand and consumers' expectations regarding environment performance.

2. Methods

This study combines economics, including prospective, and a life cycle assessment approach.

Various coherent, plausible and contrasted economic scenarios have been chosen to explore a range of possible futures up to the year 2035. The trend scenario (S1) is considered as the most probable one (from the 2014 perspective). It has been elaborated considering past trends and the most likely evolution of technology and markets. Assumptions comply with the previous prospective for French beef and dairy production in 2020 (European Commission, 2013, IDELE, 2014). Alternative scenarios (S2 to S4) have been built to explore other plausible futures, considering that new driving forces could be strengthened in the future (such as a huge growth of world food market, or changes in European social demand for food quality and environmental respect). Those scenarios have been built by expert groups gathering people working in the beef and dairy sectors and researchers. In addition, environmental scenarios have been framed, such as variations of S1 and S2, called S1Bis and S2Bis. Those two ones include high investments in GHG and energy mitigation techniques, most of them selected in Pellerin et al. (2013). Changes deal with herd management (age at first calving, mortality), feeding (use of lipids or nitrates, total crude protein intake management), crop and grassland management (legume fodder, mineral fertilization, simplified cropping practices), manure management especially biogas production, energy consumption reduction, hedges and agroforestry development to enhance carbon storage.

Finally, 6 scenarios are available:

- S1: Trend evolution (raise of milk production due to the increasing global demand, stabilization of beef production)
- S1Bis: Trend evolution and strengthened environmental strategy (*idem* and improvement of practices, including GHG mitigation options)
- S2: Answer to a high global demand (huge increase of milk and beef production to satisfy a highly raising global demand)
- S2Bis: Answer to a high global demand and strengthened environmental strategy (*idem* and improvement of practices, including GHG mitigation options)
- S3: Fold on an internal demand which goes upmarket (production reaches first national consumers' demand, wanting products from French and grass based origin, and secondly a decreasing global demand but still for "French quality" products)
- S4: Large drop in consumption and strengthened environmental strategy (decreasing French cattle production because of pressure from citizen and policy makers to reach the GHG mitigation objectives).

All scenarios are mainly driven by the level and nature of the demand for milk and beef (Table 1): consumption of milk and beef per inhabitant, imports and exports, determine the volumes to be produced, and then the number of animals needed; concerns of consumers and policy makers also influence the type of production systems and practices (degree of efficiency and of intensification per animal and area, use of inputs, etc.).

Description of the bovine French farm in 1990 and 2010 is based on national census data (animal heads, areas), on technical references from Inosys-Réseaux d'élevage (national network of about 2,000 breeders), completed when necessary by experts' opinions (diets, fertilizing practices, etc.). The number of dairy cows is distributed through 8 categories described by Ballot et al. (2010) in terms of forage systems, diet and production level; dairy heifers are classified in 3 classes, depending on age at first calving; suckler cows are distributed in 3 forage systems (with a gradient in the place of grass and maize); for the 12 other classes of animals dedicated to produce meat (weaners, young bulls, etc.), an average diet is determined.

For each class of animals and each year, or scenario, the diet and indicators such as milk productivity, weight, mortality, duration of fattening, etc. are described. According to diets, we assessed the need for grass area, maize silage, grain, etc. with a coherence control in 1990 and 2010 to match census data about areas in cattle farms. The need for concentrates purchased is then also established. The time spent in grazing pasture and the type of building allowed to determine the manure management system (slurry or farmyard manure) and the amount available for organic fertilization. Data from national census and UNIFA (national union of fertilizing producers) provided the quantity of mineral fertilizers used in 1990 and 2010. For 2035, they are determined from plant needs, and nitrogen inputs by legumes and effluents. In parallel, simulations at farm scale with the Orfée model (Mosnier et al., 2015) allowed to validate these assumptions and trajectories.

Table 1: Main characteristics of the French bovine production in 1990, 2010 and 2035, through the Gesebov scenarios

	1990	2010	S1	S1Bis	S2	S2Bis	S3	S4
Milk consumption (<i>kg/capita</i>)	351.7	311.8	268.2	268.2	295.0	295.0	241.4	214.6
Total milk consumption (<i>Mt</i>)	19.9	19.6	18.4	18.4	20.2	20.2	16.5	14.7
Milk import (<i>Mt</i>)	2.19	5.18	5.00	5.00	6.50	6.50	2.00	1.00
Milk export (<i>Mt</i>)	6.46	9.13	18.67	18.67	24.00	24.00	7.50	5.00
Milk production (<i>Mt</i>)	24.19	23.60	32.04	32.04	37.71	37.71	22.03	18.70
Evolution since 1990	-	1.1%	37.2%	37.2%	61.5%	61.5%	-5.6%	-19.9%
Milk yield (<i>kg/dairy cow</i>)	4 676	6 956	9 093	9 093	9 725	9 725	7 164	8 591
Dairy cows (<i>1000 heads</i>)	5 303	3 716	3 635	3 635	4 020	4 020	3 227	2 290
Beef consumption (<i>kg cwe/capita</i>)	29.8	26.0	21.9	21.9	24.1	24.1	19.7	17.54
Total beef consumption (<i>Mt cwe</i>)	1 500	1 641	1502	1502	1652	1652	1351	1201
Beef import (<i>Mt cwe</i>)	459	411	400	400	500	500	230	200
Beef export (<i>Mt cwe</i>)	845	541	650	650	992	992	493	306
Beef production (<i>Mt cwe</i>)	1 903	1 764	1 751	1 751	2 143	2 143	1 615	1 307
Evolution since 1990	-	-7.3%	-0.7%	-0.7%	21.5%	21.5%	-8.5%	-25.9%
Suckler cows (<i>1000 heads</i>)	3 708	4 179	3 747	3 177	4 640	3 970	2 810	2 510
Stocking rate (<i>heads/ha AA</i>)	1.86	1.91	1.92	1.95	1.89	1.91	1.82	1.80

cwe: carcass weight equivalent; AA: agricultural area

The ClimAgri® tool was used to assess the GHG emissions and energy consumption of the French bovine farm. It is the national tool to make climate and energy diagnosis for the agriculture and forestry sectors at territory level, taking all inputs and direct environmental fluxes into account, in a life cycle perspective (Doublet, 2011). This tool also provide other helpful indicators such as variation in direct soil organic carbon (under permanent grasslands and hedges) and its potential to compensate climate change (in % of the GHG emissions), direct ammonia emissions (kg N-NH₃/ha of agricultural area, AA) and need of external AA to produce feed purchased (in ha AA imported/ha AA of the French farm).

For the Gesebov project, ClimAgri® was refined. For the animal classes mentioned above, considering their diet and productivity, their enteric emissions and nitrogen excretion were specifically calculated instead of using the default factors proposed by the tool.

The LCA impacts assessed correspond to climate change expressed in kg CO₂eq, with IPCC 2013, and energy consumption, as Non renewable, fossil expressed in MJ, with the Cumulative Energy Demand 1.8 method. For climate change, emissions of GHG and carbon storage are accounted separately, according to ISO 14067:2013 (ISO, 2013).

Functional units to express impacts here are one kg of FPCM (fat and protein corrected milk) produced and one kg of LW (live weight) of beef produced by the French bovine farm.

A biophysical allocation, similar to the one applied in the French AGRIBALYSE® program (Koch and Salou, 2015), is here used to share the environmental impacts between milk and meat: the burdens associated to dairy cows are calculated in separate ClimAgri® files and attributed to milk production, while the burdens of all the other bovine animals (including dairy heifers) are attributed to meat production.

3. Results

Between 1990 and 2010, the cattle sector already reduced its GHG emissions and energy consumptions, respectively by -10.6% and -22% (Table 2). Those reductions mainly are the result of the change in cattle population: improvements in dairy productivity were followed by a decrease in the number of dairy cows, only partially balanced by an increase in the suckler cows' population. Farmers' progress in fertilization management and energy savings also contributed to the overall reduction. This bred a reduction of carbon footprint (CF) of milk from 1.44 to 1.15 kg CO₂eq./kg FPCM and a slight increase in CF of meat from 13.96 to 14.7 kg CO₂eq./kg LW (Table 4), due to changes in animal products (less meat from the dairy herd, more from the suckler herd, with longer cycles of production). The intensification of bovine production (increase in productivity and stocking

rate, Table 1) also led to decrease the areas of permanent grasslands and the subsequent carbon storage (-6%, Table 3).

The future trend for 2035 (S1) would be a total stabilization in GHG emissions and an extra decrease in energy consumption compared to 2010 (respectively +0.5% and -13%). Gains in productivity, efficiency and improvements of practices are still possible. The CF would decrease for milk (-18.6% since 2010) as for meat (-4.6%). Adoption of additional mitigation techniques in the scenario S1Bis would lead to improve both CF of milk and meat (respectively -23.8% and -17.4%), as well as the national GHG and energy balance of the cattle sector. However, in those scenarios, attention must be paid to the risk of further decrease the carbon storage (-20% and -13% for S1 and S1Bis). At the same time, the sector would become less self-sufficient in concentrates.

In the S2 scenario, the high increase in production induces more GHG for the French cattle farm, but a similar energy consumption. Thanks to gains in productivity, footprints would be reduced. The S2Bis leads to a higher environmental efficiency, almost in the same proportion as between S1 and S1Bis. The loss of carbon storage, due to conversion of grasslands to crops, is the weak point of this scenario (Table 3).

Decrease in production and more extensive grass-based systems in S3, would lead to decrease GHG and energy consumption significantly. From the products points of view, footprints are still improved. However, for milk, reductions in footprints are here the weakest. Nevertheless, this does not consider the higher carbon storage potential, which recovers the 1990 rate (Table 3). Ammonia emissions are here significantly lowered, due to a decrease in the stocking rate (Table 1) and a higher use of pasture (less manure in buildings). Feed self-sufficiency is significantly improved.

In S4, the highest decrease in national GHG emissions and fossil energy use is directly linked to the fall of milk and beef production. Gains in footprints are quite high (close to S1Bis or S2 for milk and to S3 for meat), thanks to a mix of extensive grass-based systems and very efficient herd management, both in beef and dairy productions. It allows ammonia reduction and feed self-sufficiency. Grasslands, with their capacity to compensate GHG, are here preserved (compensation increases by 27%).

The higher decrease in CF are obtained thank to the Bis scenarios.

Table 2: Total GHG emissions and energy consumptions of the French cattle sector between 1990 and 2035 through Gesebov scenarios

	1990	2010	S1	S1Bis	S2	S2Bis	S3	S4
GHG (Mt CO ₂ eq.)	83.15	74.33	74.67	66.75	87.89	77.97	60.94	47.71
Energy (GJ)	193 846	150 575	131 194	105 080	149 561	115 914	90 150	71 127

Table 3: Additional direct environmental indicators of the French cattle sector between 1990 and 2035 through Gesebov scenarios

	1990	2010	S1	S1Bis	S2	S2Bis	S3	S4
C compensation (%)	16%	15%	12%	13%	10%	12%	16%	19%
kg N-NH ₃ /ha	35	33	34	32	35	32	28	27
Imported/French AA (%)	32%	31%	33%	34%	26%	27%	24%	24%

Table 4: Environmental impact of milk and beef in France between 1990 and 2035 through Gesebov scenarios

	1990	2010	S1	S1Bis	S2	S2Bis	S3	S4
kg CO ₂ eq./kg FPCM	1.44	1.15	0.94	0.88	0.89	0.82	0.98	0.89
MJ/kg FPCM	5.25	3.66	2.17	1.94	1.92	1.65	2.16	1.99
kg CO ₂ eq./kg LW	13.96	14.70	14.03	12.14	13.92	12.12	13.37	13.07
MJ/kg LW	19.34	20.05	19.33	13.52	19.82	13.82	14.50	14.24

FPCM: Fat and Protein Corrected Milk; LW: live weight

4. Discussion

The Gesebov prospective scenarios are voluntarily contrasted to allow the cattle sector to think about the different possible futures and to decide the actions they want to invest in. One has to keep in mind that reality will be somewhere in between.

The environmental impacts obtained for 2010 are slightly higher than those from the AGRIBALYSE® French LCI database (ADEME, 2015) for the average French milk (0.89 kg CO₂eq/kg FPCM, 2.17 MJ/kg FPCM) and the average French beef cattle (11.93 kg CO₂eq/kg LW, 19.60 MJ/kg LW) at farm gate. For AGRIBALYSE®, optimized systems, described in case studies, were used (Koch and Salou, 2015), while data used here allowed to draw a more realistic picture of the French cattle farm, leading to higher footprints. Carbon footprints are in the range of those met in the bibliography for milk, in France (Dollé et al., 2013: 0.89 kg CO₂eq/kg FPCM), Italy (Guerci et al., 2013: 1.3 kg CO₂eq/kg FPCM) or New Zealand (Basset-Mens et al., 2009: 0.93 kg CO₂eq/kg FPCM), as well as for beef, in France (Moreau et al., 2013 and Veysset et al., 2014: 12.8 to 14.5 kg CO₂eq/kg LW) or USA (Pelletier et al., 2010: 14.8 to 19.2 kg CO₂eq/kg LW).

The detailed use of ClimAgri® (instead of using default values per head for enteric methane or nitrogen excretion) gave a valuable precision, needed when considering a sector with a high contribution to the national GHG balance. It also allowed sensitivity of the results through scenarios. One important point of the ClimAgri® that should be improved in the future, is the accounting for soil carbon dynamics. Even if the method provides accounting for carbon below permanent grasslands and hedges, improvements are also needed to consider temporary grasslands and crops. It would help considering the whole picture of the contribution of the cattle sector to climate change and to identify strengths and weaknesses, especially among scenarios with various use of soils (grains and forages vs grass). Ongoing programs, focusing on how to account for soil carbon in LCA, should also help to improve this point in the future.

Considering the national targets for GHG mitigation in the future, S3 and S1Bis scenarios would be close to the official French objectives in the mid-term (-12% between 2013 and 2028). Only the S4 scenario would be able to reach the long-term target (-50% between 1990 and 2050). This should bring awareness of possible split-over effects. What would the other environmental impacts of the sector be (such as effects on biodiversity, water quality, etc.)? If cattle activity declines, what would the available areas become (agricultural production or not, which practices, which carbon loss)? If global demand raises but French cattle activity declines, reducing its export, production should be transferred to other countries in the world, with lower environmental efficiency sometimes. Then, how would global GHG balance evolve? Last, but not least, social and economic impacts of those scenarios, especially as far as employment is concerned, were not considered here, which opens a wide and interesting field of study and should help considering futures for the cattle sector, not only from a carbon footprint point of view.

5. Conclusions

Analyses of French cattle production systems, livestock population and agricultural practices allowed us to specify GHG emissions, energy consumption and corresponding footprints, between 1990 and 2035. Results show that the sector already enhanced its environmental efficiency and that improvement strategies are still available for the future. Producing milk and beef can absolutely comply with climate and energy savings, as long as mitigation strategies are integrated in the systems.

One has to keep in mind that scenarios are prospective and not predictive ones. The results should now help the sector to make coming stakes its own, with always more questions and strong expectations for livestock regarding climate change.

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ORAL PAPERS

NOVEL FOODS

19. Novel processing and approaches to food

29. LCA of laboratory-scale production of mannosylerythritol lipides

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ABSTRACT

We present LCA results of the production of mannosyl erythritol lipides (MEL). These are biogenic molecules that can act as surfactants, emulsifiers and antimicrobial agents in various consumer products such as detergents and skin moisturizers. The study is part of the ongoing project “Biotensides” that aims to identify, and produce specific MEL types, as well as demonstrate their use in selected end user applications.

The MEL under scrutiny are produced by fungi in a cascade of batch processes. Raw materials are sugars and fatty acids; in our case glucose and soybean oil. We represent the laboratory-scale process chain from raw materials to partially purified MEL (“raw extract”) in GaBi, including several cultivation steps, various sterilization/cleaning processes, as well as the (partial) purification.

We obtained LCA results for the impact categories Global Warming Potential, Acidification Potential, Eutrophication Potential, Abiotic Depletion Potential, as well as for the inventory quantity fossil primary energy demand. One key finding is that the impacts are dominated by the various energy-demanding processes. Two scenarios are included for a simple sensitivity analysis: “35d” with a long fermentation time, and “10d” with a shorter fermentation time.

Assessment of the laboratory-scale production is the first step in optimizing the process chain towards higher volumes. The dominant process is the fermentation, which is also the main step in the production of MEL. It involves an agitated reactor running for several weeks. The 10d scenario demonstrated that there is considerable improvement potential in optimizing the fermentation time. We are planning to amend the study once results for the up-scaled process chain are obtained. For continued environmental optimization it is relevant to know at which point the impacts become more dependent on raw materials than on energy consumption. The assessment also allows LCA researchers to learn about efficiency potentials in biotechnological process chains.

Keywords: surfactants, renewable raw materials, fermentation

1. Introduction

Surfactants are a class of substances that is used most commonly as a component in cleaning products. In this or other roles, they are used in the textile, cosmetics, pharma, food, and plastics industries. They owe their flexible use to their molecular structure: Surfactants are amphiphilic molecules, composed of a hydrophilic and a lipophilic part, which enable them to arrange themselves at the interface of polar (e.g. water) and nonpolar (e.g. oil) media. Thus, they perform as solubilizers, detergents, and emulsifiers (Soberón-Chavez and Maier 2011). Environmental problems caused by surfactants and the finite nature of fossil resources have led to an increasing development of partly or completely bio-based and/or biodegradable surfactants, referred to as biosurfactants in this article (Mann and Bidwell 2001, Grand View Research 2014, Eskuchen and Nitsche 1997). Although most conventional biosurfactants are artificially synthesized, they can also be generated in entirely biological ways by microorganisms, i.e. by fermentation. Microbial surfactants have not yet achieved huge economic importance in the surfactant sector, mostly due to high production costs (Grand View Research 2014, Zion Research 2015). But several features of biosurfactants offer advantages over their conventional counterparts: they are entirely biodegradable, entirely bio-based, less toxic than alternative surfactants, and microbial production allows the generation of a great variety of molecular structures (Soberón-Chavez and Maier 2011). The range of suitable renewable substrates is quite broad, including residuals from the food, agricultural, and forestry industries, such as molasses and lignocellulose (Arutchelvi and Doble 2011, Faria et al. 2014, Banat et al. 2010, Nitschke et al. 2005). Promising candidates for microbial biosurfactants are glycolipids such as mannosylerythritol lipides (MEL) – the topic of this article – and non-ionic surfactants synthesized by various yeast strains (Arutchelvi and Doble 2011).

Depending on the substrate, MEL-producing microorganisms always produce a mixture of different MEL molecules. All MEL molecules are comprised of the sugar moiety mannosylerythritol as hydrophilic group and different types of one or more fatty acid chains and/or acetyl as hydrophobic moiety (Banat et al. 2010, Arutchelvi and Doble 2011). Chemical properties such as polarity of the MEL mixture vary according to the composition of the hydrophobic part. Possible substrates for MEL synthesis are oils like soybean oil, safflower oil, olive oil, castor oil, palm oil, and coconut oil. Suitable carbohydrate sources are e.g. glucose and sucrose (Arutchelvi and Doble 2011, Morita et al. 2015). MEL have been applied as ceramide-like agent in a skin care products, and their suitability for bioremediation of oil has been tested (Morita et al. 2013, Mulligan 2009). A useful instrument to assess environmental performance of these new microbial surfactants is life cycle assessment (LCA).

2. Goal and Scope

Goal: This LCA study is carried out to detect production steps and inputs with major influence on the environmental impact of the MEL life cycle. The MEL mixture in question is not yet produced in relevant quantities, so this study at such an early development stage gives the possibility to improve the production process before it is established in a higher production scale. According to the LCA results proposals will be made for measures which can help to optimize the production and thus decrease the environmental impact of MEL. This LCA study examines the environmental issues connected to the production and disposal of a specific MEL mixture on a laboratory scale. The **functional unit** is 1 kg MEL raw extract.

Production: The production of MEL is referred to as the foreground system. It is divided into two major parts: first, the biotechnological synthesis of MEL through fermentation, and second, the product separation using a solvent. To gain a suitable amount of MEL-producing cells for fermentation, the yeast strain is first cultivated in a petri dish on agar medium. Cells from the petri dish are then used to inoculate a liquid pre-culture which itself serves as inoculum for the fermentation in a bioreactor. During the fermentation, first glucose is used as a substrate to promote cell growth, which is then complimented by soybean oil to start the MEL production. The microorganisms produce a mixture of MEL. Since this mixture is processed and used as a whole, the specific composition is not relevant for the LCA study. After synthesis, MEL molecules have to be separated from the fermentation broth. To decrease the solubility of MEL in the aqueous medium, acid is added to the broth. The next step is a centrifugation to isolate MEL, cellular biomass and residual substrate from the aqueous phase, forming a honey-like pellet in the centrifugation container. The MEL are then extracted from the pellet with ethanol. A second centrifugation allows the separation of the MEL-ethanol phase from the residual aqueous phase. The MEL-ethanol mixture is distilled and the ethanol evaporated, which gives the so called “MEL raw extract”. The raw extract still contains about 10 % of soybean residue. The developers estimate that these residuals probably do not disturb the use of MEL, so no further purification is assessed. Hence the functional unit of 1 kg raw extract rather than pure MEL.

Use: The use phase is not included in this study. Several varying options for the use of these biosurfactants exist and it is not clear at this point which will be the most promising one. Also, it is plausible to assume that MEL will not have any environmental impact during the use phase. They don't need to be heated or stirred, and exposure to users is assessed separately.

End-of-life: The disposal of MEL is included in this study. Whether the end use is in cosmetics or detergents, MEL will end up in waste water. We assume that waste water is treated in a waste water treatment plant, including stabilization and incineration of the sewage sludge.

System boundaries

The MEL life cycle starts with the generation of all inputs needed for the production. This includes extraction and processing of energy carriers, production of the substrates and other medium ingredients, production of ethanol, as well as other additives and means of production. Next step is the production of the aforementioned inputs themselves, followed by MEL synthesis, MEL raw extract separation and Mel disposal via waste water.

Not part of this LCA are: yeast strain development and storage, production and disposal of glassware, machines and other lab equipment such as gloves, pipettes or paper towels. The electricity and water consumption of lab equipment and machines such as autoclaves or centrifuges is included.

The MEL production takes place in Stuttgart, Germany. All transport distances are calculated on this basis, and the German electricity grid mix is assumed for all foreground processes.

Scenarios: Two scenarios of MEL raw extract production are considered: one with 35 days fermentation time and one with 10 days. Fermentation time under research conditions has been extended to 35 days for additional tests, but practically all substrate is fermented within 10 days, according to the developers.

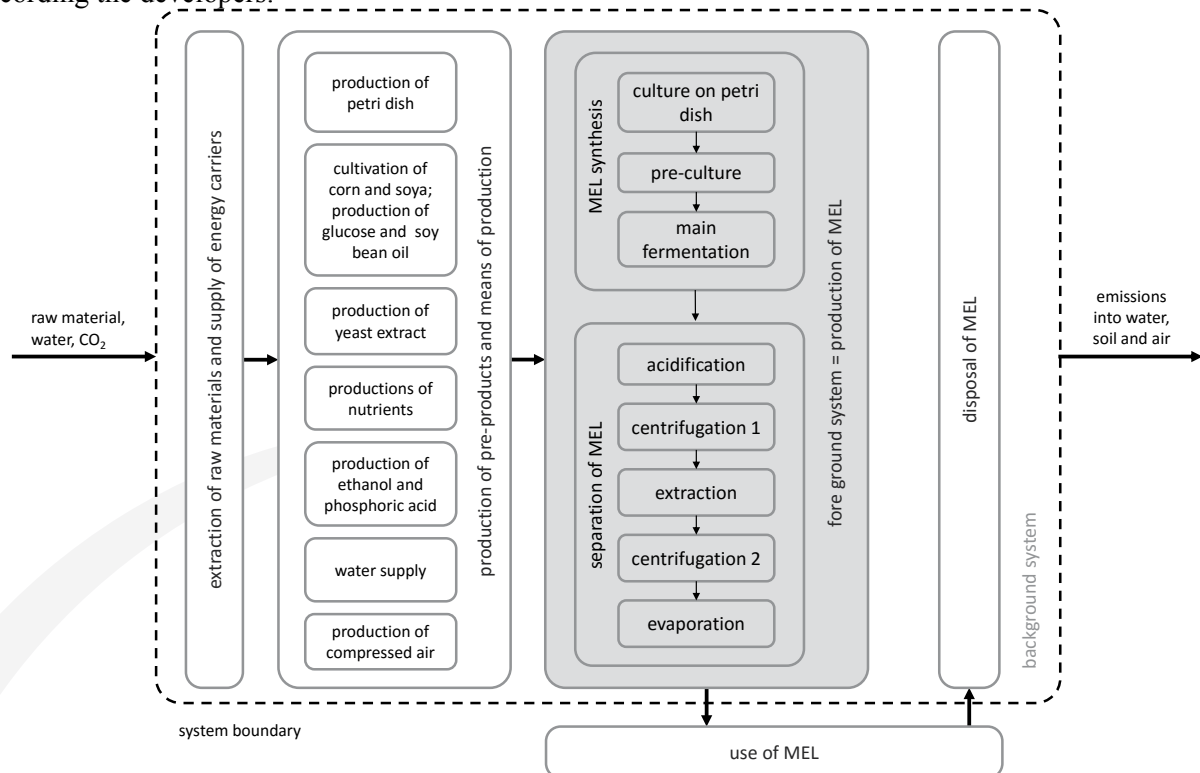


Figure 1: System boundaries of MEL LCA on lab scale. grey: foreground system with MEL synthesis and separation of MEL. white: background system including extraction of raw materials, supply of energy carriers, production of pre-products and means of production as well as MEL disposal. The use phase of MEL is outside the system boundaries.

3. Inventory

Data collection: For MEL production, primary data have been collected from the Fraunhofer Institute for Interfacial Engineering and Biotechnology. These data include medium composition, MEL yield for all production steps, and equipment settings. Electricity and/or water consumption data for the machines were obtained from the respective manuals. For the autoclave and the bioreactor, consumption data were obtained from the manufacturer, as these could not be calculated from the manuals and given information from lab protocols.

Data for background processes like electricity generation and production of precursors and means of production are taken from the *GaBi professional* database supplied by *thinkstep AG*. *GaBi* software (version 6.115) was used for modelling. A summary of data sources is compiled in Table 1.

Cultivation on petri dish: This step accounts for far less than 1% of ingredients and environmental impact of the entire product system, and is not described in detail.

Pre-culture: The pre-culture is divided into two steps. The first pre-culture is a set of 5 flasks, each containing 20 ml of liquid culture. The second pre-culture, inoculated with the first one, is a set

of 9 flasks with 100 ml medium each. In total, 1 L culture suspension is provided, of which 100 ml is disposed and 900 ml are used to start the main fermentation. Each pre-culture is incubated for 2 days at 30 °C on a rotary shaker at 110 rpm. The medium contains glucose, nutrients (monopotassium phosphate, ammonium nitrate, yeast extract) and deionized water. It is sterilized in an autoclave. The pre-cultures are set up and inoculated in a laminar flow cabinet (LFC).

Main fermentation: The main fermentation takes place in a stirred and aerated bioreactor at 27 °C over a 35 day period. It is conducted in fed-batch mode and the final fermentation volume is 26 L. The fermentation starts with an initial amount of starting medium (18 L) containing glucose as a carbon source for cell growth. Later, glucose and soybean oil are added, as the MEL production is based on soybean oil as the carbon source. The origin of both substrates is unknown and it is assumed that glucose is produced from corn in Germany and soybean oil is produced in northern Italy (where most European soybean oil comes from, see Perseus BVBA 2012).

The final medium is composed of the following ingredients: glucose and soybean oil as carbon sources, nutrients (monopotassium phosphate, sodium nitrate, yeast extract, magnesium sulphate) and deionized water.

The bioreactor is aerated with compressed air and stirred to maintain proper oxygen supply to the cells. The stirrer disperses the air into fine gas bubbles, thus creating a large bubble surface for adequate oxygen transmission from the gaseous phase of the bubbles into the liquid phase of the fermentation broth. It also homogenizes the medium. The so-called foam kill is a rotating structure similar to a fan which destroys foam that is formed at the surface and causes the fermenter to overflow. It thus prevents the loss of fermentation broth. To keep the temperature at 27 °C, the fermentation medium can be warmed up by an electric heater and cooled by cooling water running through the bioreactor wall. The use of heating and cooling water depends on the heat production of the cells and it is not documented how long and to what extent heating and cooling have been used. This is why for electricity and water consumption, data supplied by the bioreactor manufacturer have been used.

Once the starting medium is in the bioreactor, it is sterilized using the steam which is created from the medium during heating. During sterilization the temperature in the fermenter is increased up to 120 °C for 20 min by an electric heater and then cooled down by cooling water. The various feeds are sterilized separately in an autoclave. After fermentation, the bioreactor containing the fermentation broth with MEL is again sterilized to stop any microbial activity.

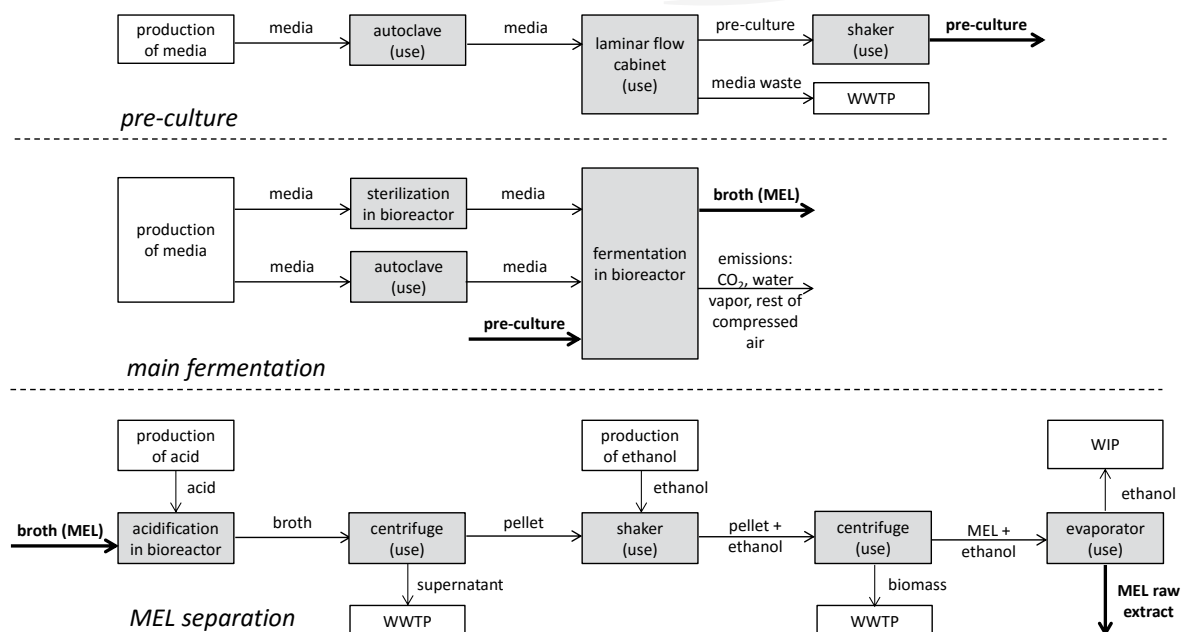


Figure 2: Overview of MEL production and product separation. grey: processes with use of laboratory equipment. white: processes of the database. WWTP = Waste water treatment plant. WIP = waste incineration plant. Cultivation on petri dish and cleaning of glassware are not shown.

Acidification: The whole fermentation broth is acidified by ca. 260 ml phosphoric acid.

Centrifugation 1: The fermentation broth is then centrifuged for half an hour at 17,700 g. The supernatant is decanted and disposed while the MEL-biomass pellet is transferred to extraction.

Extraction: The pellet is supplemented with 5 times the pellet-volume of ethanol and incubated for 1 h in an incubator at 30 °C on a rotary shaker at 120 rpm.

Centrifugation 2: The extraction mixture is centrifuged at the same conditions as before. The MEL and the ethanol are separated from the biomass, which is disposed.

Evaporation: The ethanol is evaporated in a rotary evaporator within 48 h. The evaporator is cooled with water and propylene glycol (50/50). At the end of the product separation 3.4 kg MEL raw extract is obtained.

Table 1: Inputs for MEL production and separation (foreground system). Fermentation duration is 35 d. Values are given per functional unit.

Inputs	Pre-culture	Main fermentation	MEL separation
<i>medium</i>			
glucose	9.4*10 ⁻³ kg	0.52 kg	-
soybean oil	-	1.04 kg	-
nutrients	6.7*10 ⁻⁴ kg	0.062 kg	-
water (deionized)	0,3 kg	6.2 kg	-
compressed air	-	126 m ³	-
<i>machinery</i>			
electricity	69.1 MJ	1789 MJ	65.3 MJ
cooling water	1.9*10 ⁻³ kg	11.0 kg	0.04 kg
tap water	17.2 kg	8.6 kg	15.0 kg
water (deionized)	5.8 kg	14.6 kg	0.39 kg
<i>others</i>			
phosphoric acid	-	-	0.08 kg
ethanol	-	-	6.79 kg
propylene glycol	-	-	0.39 kg
diesel fuel	-	1.4*10 ⁻² kg	-

4. Impact assessment

The following impact categories have been chosen for the life cycle impact assessment: Global Warming Potential 100 years (GWP), Eutrophication Potential (EP), Acidification Potential (AP) and Abiotic Depletion Potential (ADP). For all categories impact models from the Institute of Environmental Sciences (CML) at Leiden University are used. Additionally, consumption of fossil primary energy has been analyzed (PE fossil).

Figure 3 shows the impact of the respective steps of the MEL raw extract life cycle in the all impact categories. The bars are divided into the following life cycle steps: Production of glucose and soy bean oil for all steps of MEL synthesis; electricity generation for the bioreactor; contribution of other inputs of the main fermentation, e.g. medium provision and electricity generation for aeration. Regarding MEL product separation, the most contributing steps extraction and evaporation are shown separately, while the rest of the product separation is summed up. The last phase of the life cycle is the end-of-life phase with MEL disposal in a waste water treatment plant.

GWP: The production and disposal of MEL in lab scale emits 341 kg CO₂ equivalent of greenhouse gases in the 35d-scenario. The electricity generation for the fermenter contributes most to the environmental impacts with 82 % and the entire main fermentation contributes a GWP share of

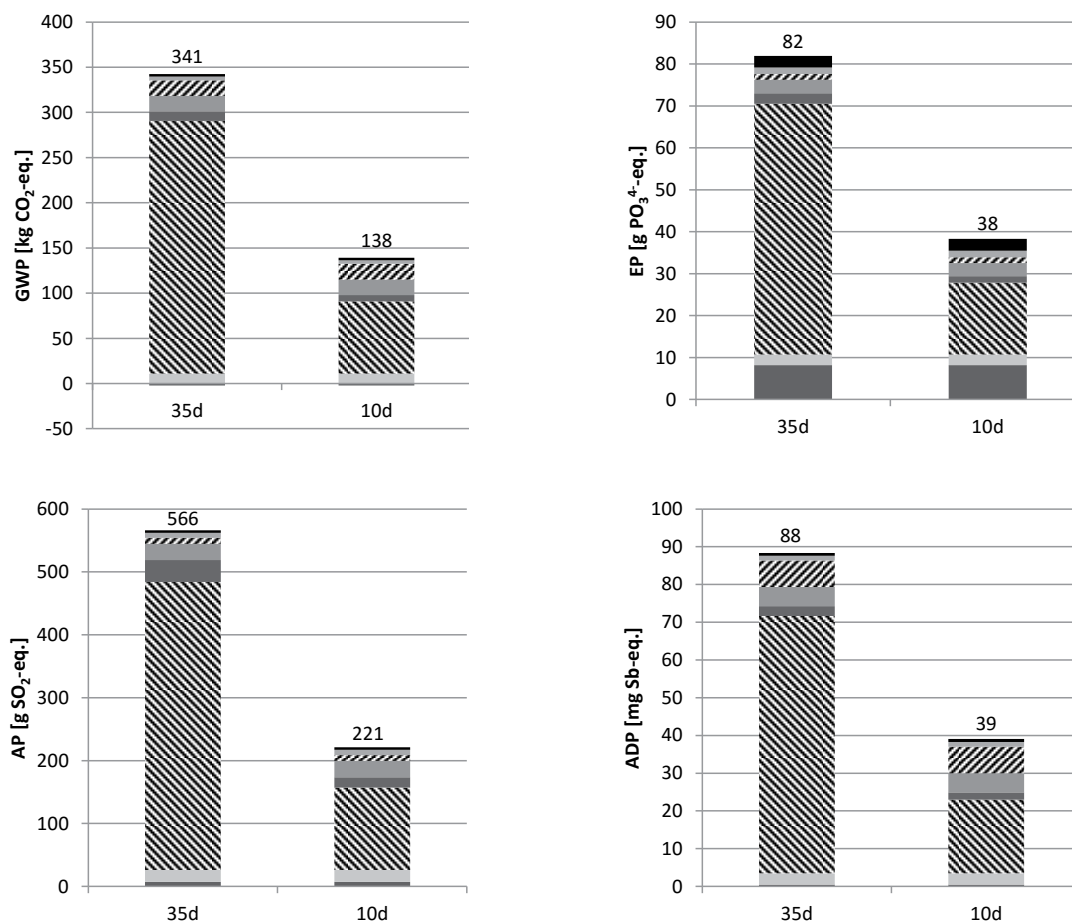
85 %. Petri dish cultivation and pre-culture together are responsible for 3 % of the GWP. The extraction and evaporation steps within the product separation phase contribute 5 % each. Other steps from MEL synthesis and separation contribute only negligible amounts. The substrate provision contributes negatively to the GWP, with -1 %. When fermentation time is reduced (scenario 10d), the GWP of the total MEL raw extract life cycle drops to 138 kg CO₂ equivalent because the GWP from electricity generation for the bioreactor is reduced from 279 to 80 kg CO₂ equivalent. A slight decrease is also found in the other steps of main fermentation.

EP: The Eutrophication Potential of MEL raw extract is 82 g phosphate equivalent with 35 days fermentation time and 38 g with 10 days. Electricity generation for the bioreactor is responsible for the greatest impact share in this category. Substrate production contributes positive numbers to the EP, with 8 g phosphate-equivalent per kg MEL raw extract.

AP: In the 35d scenario, the Acidification Potential of MEL raw extract is 566 g sulfur dioxide equivalent, compared to 221 g with 10 days fermentation time. Electricity supply clearly dominates the emission of acidifying substances.

ADP: Electricity generation for the bioreactor during the main fermentation was again identified as the major contribution. It is attributed with 81 % of the impacts in this category in the 35d scenario and 57 % in the 10d scenario.

PE fossil: MEL raw extract production and disposal consume roughly 4,700 MJ per kg raw extract in the 35d scenario and ca. 1,920 MJ in the 10d scenario.



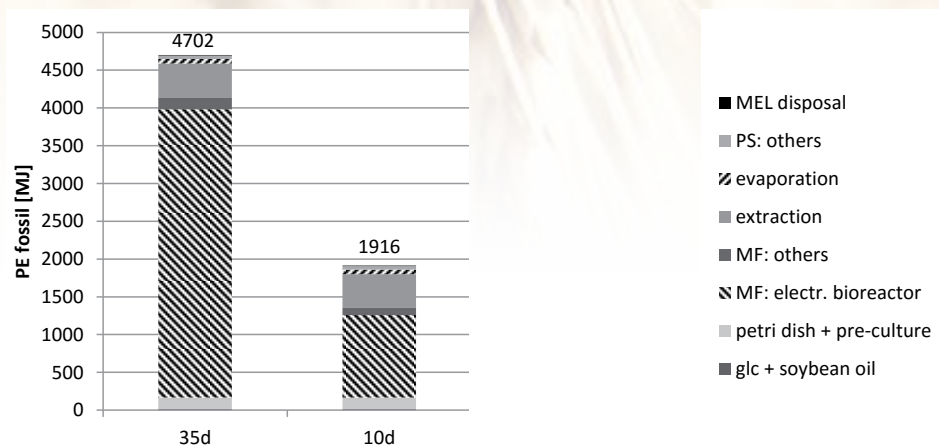


Figure 3: Life cycle impact assessment of MEL production in lab-scale: Results of the impact categories GWP, EP, AP, ADP and fossil Primary Energy Consumption. Values per functional unit. Left bars: 35d: Fermentation time is 35 days. Right bars: 10d: Scenario with 10 days fermentation. PS = product separation of MEL. MF = main fermentation. glc = glucose. electr. = electricity.

5. Interpretation

In all categories, electricity generation for the fermenter contributes most to the environmental impacts. The electricity in the LCI model refers to the German grid mix, which consists of roughly 60 % of electricity produced from fossil resources (40 % from coal). The contribution of electricity generation to the EP is mainly caused by nitrogen oxides emitted from thermal power plants. Nitrogen oxides and sulfur dioxide – also from electricity production – are responsible for most of the Acidification Potential. ADP and PE fossil are mainly caused by raw material extraction of fossil resources needed for electricity generation. The great influence of electricity generation in all categories is due to the relatively high energy consumption per MEL output. Lab scale production focuses on feasibility of the process by studying the best conditions for cell survival and MEL production rather than on energy efficiency. Figure 4 shows which potential for impact reduction is given by the reduction of energy consumption in the 10d scenario. Impacts decrease about 53 to 60 % across the various categories.

The use of renewable raw material as substrate comes with a benefit for the GWP. Due to carbon dioxide fixing during plant growth, the production of glucose and soybean oil contributes with a negative value to the greenhouse gas emissions. However, the energy consumption is so dominant in the GWP category that the CO₂ sequestration effect is substantially diluted. In categories other than the GWP, substrate production from plants doesn't have this effect. Especially the EP contribution of plant matter is relevant. In agricultural activities such as fertilizer application, nitrate, organically bound nitrogen and phosphate are emitted, which contribute to eutrophication.

Impacts from product separation are attributed mainly to the production of ethanol or its disposal through incineration. The production of ethanol from oil has been assumed in the model, which is why ethanol production influences the results of the ADP and PE fossil. Its disposal in a waste incineration plant causes the emission of CO₂, mainly contributing to the GWP. The total ethanol use per functional unit is high because ethanol is liberally used in the lab environment and is not recycled.

Energy efficiency is usually improved when production processes are scaled up. Follow-up studies will include MEL production with a higher fermentation volume. Another option for the reduction of electricity consumption is the reduction of the fermentation time, i.e. the time of the use of stirrer and foam kill. Experts from the laboratory estimate that a robust production process with less foam formation can be reached, eliminating the need for a foam kill entirely. Regarding material use, the recycling of ethanol use must be one aim in the course of the optimization of MEL production. In an upscaled production process, material consumption will probably play a more important role, even if

energy generation will still have a major influence. Then, measurements in order to diminish the environmental impact also have to focus on the origin and kind of the renewable substrates.

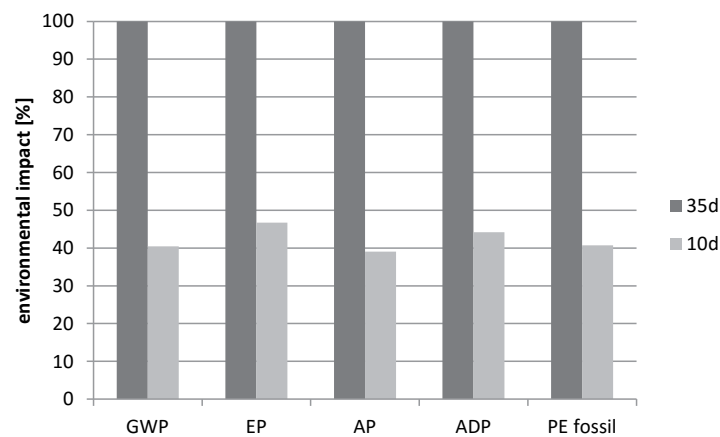



Figure 4: Reduction of the environmental impacts through reduction of the fermentation time from 35 to 10 days. The 35d-scenario is set to 100 %.

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90. An environmental techno-economic assessment of algal-based biorefineries

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ABSTRACT

Microalgae have been proposed as an important feedstock for the biobased economy. However, the economic profitability and environmental impact of microalgae-based biofuels remains an issue. A microalgae-based biorefinery, valorizing multiple products, has been stated as an interesting solution. To explore the feasibility of this concept, an assessment of both the economic feasibility and environmental impact is required. This paper extends a techno-economic assessment of an algal-based biorefinery with an integrated life cycle assessment (LCA). Four different scenarios, ranging from a basic scenario with conventional technologies to more advanced scenarios with innovative technologies, are assessed. Using this environmental techno-economic assessment, the biomass productivity was identified as a crucial parameter for both the economic and environmental feasibility. The inclusion of a membrane to enable the recycling of water and salt had a positive influence on both the economic profitability and the environmental sustainability of the project. The environmental techno-economic assessment can provide important information to optimize the economic profitability and environmental impact of new technologies and to catalyze the transition to a biobased economy.

Sustainability assessment, integrated assessment, microalgae

1. Introduction

The group of algae can be defined as plant-like organisms which contain chlorophyll α as the primary photosynthetic pigment and lack typical plant structures like stem, roots and leaves (Lee, 2008). Microalgae are the small algae which can generally not be seen with the naked eye (Lundquist et al., 2010). These microorganisms have a relatively high productivity and can accumulate multiple valuable components, such as carotenoids. They are also considered as a potential feedstock for biofuels, as they can accumulate high amounts of lipids (Mata et al., 2010). However, there are a few challenges concerning the commercialization of these applications. Firstly, the production price is still too high to enable the profitable valorization of algal-based biofuels (Cheng and Timilsina, 2011). Therefore, an algal-based biorefinery, which valorizes multiple products, has been proposed as a more promising commercialization strategy in the near term (Zhu, 2015). Secondly, the sustainability of biofuels has been questioned, for example due to the large freshwater consumption of large scale production (Chisti, 2013). A thorough sustainability assessment of algal-based biorefineries in an early stage of technology development is therefore required. Both problems are usually discussed independently, with economic assessments aiming to quantify the economic potential and life cycle assessments aiming to quantify the environmental impact. This study extends the existing techno-economic assessment (TEA) methodology as introduced by Van Dael et al. (2014) with an environmental assessment, based on the life cycle assessment (LCA) methodology, as suggested by Thomassen et al. (2016b). The case study is based on previous work of the authors (Thomassen et al., 2016a). The results will enable the identification of trade-offs or synergies between the different sustainability dimensions and the crucial parameters which should be enhanced to shorten the time-to-market for algal-based biorefineries.

2. Methods

A TEA aims to quantify the economic potential of new technologies during each stage of their technology development over the entire value chain. The mass and energy balance is integrated to enable a dynamic model, where an alteration in each technological or economic input assumption is

directly translated in the results. The environmental techno-economic assessment integrates the LCA framework into the TEA methodology and consists of five steps: (1) *Market study*, (2) *Process flow diagram (PFD) and mass and energy balance*, (3) *Environmental assessment*, (4) *Economic assessment*, and (5) *Interpretation*.

This paper focusses on the environmental part of the assessment, an in-depth description of the TEA methodology can be found in the previous study by the authors (Thomassen et al., 2016a). The environmental assessment uses the ReCiPe midpoint impact indicators (hierarchist perspective): climate change (CC), ozone depletion (OD), terrestrial acidification (TA), human toxicity (HT), photochemical oxidant formation (POF), particulate matter formation (PMF), freshwater eutrophication (FE), marine eutrophication (ME), ionizing radiation (IR), terrestrial ecotoxicity (TET), freshwater ecotoxicity (FET), marine ecotoxicity (MET), agricultural land occupation (ALO), urban land occupation (ULO), natural land transformation (NLT), water depletion (WD), mineral resource depletion (MD) and fossil fuel depletion (FD) (Goedkoop et al., 2013). The environmental impacts of all inputs and outputs of the process were extracted from the Ecoinvent database version 3.2 using the Simapro software and transferred to a spreadsheet model in order to directly link the environmental impacts with the technological analysis.

3. Case study

The case study is based on an update of previous work of the authors, where four algal-based biorefinery scenarios were discussed. An extended description of the case studies can therefore be found in Thomassen et al. (2016a). The four scenarios range from a basic scenario with conventional technologies to a more advanced scenario. An alternative scenario, using a different microalgae species with a different end product is assessed as well. All scenarios produce 170 tonnes dry weight (DW) biomass per year for the production of carotenoids and fertilizer. The biomass production was used as a constant factor to enable both the comparison of the cultivation phase and the downstream processing.

The basic scenario cultivates the microalgae *Dunaliella salina* in open ponds. This cultivation consists of two stages. During the first stage, optimal growth conditions are assumed to enable maximum growth of the microalgae. During the second stage, stress conditions are induced to enable maximum accumulation of the carotenoid β -carotene. The microalgae take up CO_2 with an efficiency of 45%, and convert it into O_2 (Pires et al., 2012). The resulting CO_2 is emitted to the environment. The nitrogen fertilizer is converted by the microalgae to N_2O ($2.35 \cdot 10^{-5} \text{ kg N}_2\text{O-N kg N}^{-1}$) and NH_3 ($4 \text{ kg N}_2\text{O kg N}^{-1}$) (Fagerstone et al., 2011, Yuan et al., 2014). Due to nitrogen-limiting conditions, no N_2O is produced in the second stage of cultivation (Fagerstone et al., 2011). The microalgae are harvested using a centrifuge. As *Dunaliella salina* survives in very saline conditions, a washing step is required to lower the salt content of the biomass. The biomass is dried using a spray dryer. Hexane is used as a solvent to extract the β -carotene. The extraction results in a hexane emission of $2 \text{ g kg biomass}^{-1}$ (Lardon et al., 2009). The residual biomass goes to an evaporation step where the hexane can be recycled. After this step the residual biomass is sold as fertilizer. The extract goes to a vacuum distillation where the carotenoids are purified and the hexane can be recycled.

The intermediate scenario adds a preliminary harvesting step between the cultivation step and the harvesting with centrifugation. For this step, the integrated permeate channel (IPC) membrane as developed by the Flemish Institute of Technological Research (VITO) is used to recycle the medium (De Baerdemaeker et al., 2013). The other steps of the production process remain the same as for the basic scenario. The advanced scenario uses a photobioreactor (PBR) for the cultivation step instead of open ponds. This results in a higher N_2O emission to the environment during the first stage of cultivation, $3.9 \cdot 10^{-3} \text{ kg N}_2\text{O-N kg N}^{-1}$ (Fagerstone et al., 2011). The other steps of the production process remain the same as in the intermediate scenario. The alternative scenario assesses an alternative microalgae-based biorefinery concept, based on the cultivation of *Haematococcus pluvialis* in a PBR. Unlike *Dunaliella salina*, *Haematococcus pluvialis* is a freshwater algae. Therefore, no

washing step is required. The cell wall of *Dunaliella salina* is relatively thin and breaks during centrifugation and drying (Oren, 2005). The cell wall of *Haematococcus pluvialis* is thicker and requires a cell disruption step, using a bead mill to enable the extraction of the cellular components (Mendes-Pinto et al., 2001). The other steps in the production process remain the same as in the advanced scenario.

The environmental assessment will use the total production process of the carotenoids and the fertilizer over the entire lifetime (10 years) as a functional unit to ensure one harmonized functional unit for the economic and environmental assessment. The impacts related to the conventional production of fertilizer, which is a coproduct in the algal-based biorefineries, were considered as avoided impacts. The environmental assessment adopted a cradle-to-gate perspective where the use and the disposal phase of the carotenoids and fertilizers were not included. The biorefinery scenarios were considered for Belgian conditions. For the environmental impact of the equipment, two proxy parameters were used. For all tanks and centrifuges, the mass of stainless steel of a centrifuge, adapted to the required capacity, was used as material. The evaporator, bead mill, distillator and spray dryer used the mass of stainless steel of the spray dryer. A linear sizing factor was used to adapt the weight to the required capacity. The end of life phase assumed that 95% of the plastic and stainless steel could be recycled and the other 5% would be landfilled. The upstream environmental impact of smaller equipment such as pumps and membranes was not included in the assessment.

4. Results

The mass and energy balance of the four scenarios is illustrated in Table 1. The addition of the medium recycling step in the intermediate scenario lowers the water and salt consumption and the amount of wastewater. The electricity consumption in the advanced and alternative scenario is much higher compared to the previous scenarios as pumping and mixing in a PBR requires more energy.

Table 1: Mass and energy balance

Parameter	Unit	Basic	Intermediate	Advanced	Alternative
Input					
Water	m ³	4,718,608	1,301,961	747,486	213,372
Salt	tonnes	629,552	129,867	48,974	0
Nutrients	tonnes	6,516	6,516	6,516	6,516
CO ₂	tonnes	6,904	6,904	4,935	4,930
Hexane	kg	3,827	3,827	3,827	3,936
Electricity	GJ	128,269	107,382	1,996,831	2,239,348
Heat	GJ	1,400,535	1,310,600	37,918	49,884
HDPE	tonnes	4,507	4,507	2,305	2,578
Stainless steel	tonnes	31	31	9	10
Intermediate product					
Algae biomass	tonnes	1,700	1,700	1,700	1,700
Output					
Fertilizer	tonnes	1,476	1,476	1,476	1,583
Carotenoid	tonnes	141	141	141	43
Wastewater	m ³	5,000,345	1,362,019	770,180	213,418
CO ₂ emissions	tonnes	3,797	3,797	1,826	1,824
N ₂ O emissions	kg	15	15	2,437	2,437
NH ₃ emissions	tonnes	16	16	16	16
Hexane emissions	kg	3,827	3,827	3,827	3,936
HDPE to recycling	tonnes	4,281	4,281	2,190	2,449
HDPE to landfill	tonnes	225	225	115	129
Stainless steel to recycling	tonnes	30	30	9	9
Stainless steel to landfill	tonnes	1.56	1.56	0.46	0.48

The results of the environmental impact assessment are illustrated in Table 2. The main process contributing to the environmental impact in the basic scenario is the cultivation. This process contributed more than 90% to all impact categories except for ionizing radiation (64%) and marine eutrophication (75%). In these impact categories, the energy consumption during the drying process and the wastewater treatment also had a major influence. The main contributors to the environmental impact during cultivation were the salt and heat consumption.

The environmental impact of the intermediate scenario is lower for all impact categories compared to the basic scenario. The medium recycling step results in a lower salt requirement in the intermediate scenario, which is an important contribution to the environmental impact. The cultivation process is again the main contributor for all environmental impact categories, although its contribution is lower compared to the basic scenario. In the intermediate scenario the heat and nutrient consumption were the main contributors during cultivation.

In the advanced scenario, twelve environmental impact categories are higher compared to the intermediate scenario and six impact categories are reduced. The cultivation process has a contribution of more than 90% to all environmental impact categories, except for marine eutrophication (89%). The main contributor during cultivation is the electricity consumption. The ionizing radiation and water depletion potential are higher than in the basic scenario.

In the alternative scenario, the biomass productivity was lower, which required more water, land and energy. This results in higher environmental impacts compared to the advanced scenario for categories such as climate change, ozone depletion, ionizing radiation, natural land transformation, water depletion and fossil fuel depletion. The other impacts are lower due to the lack of salt addition. In this scenario, the cultivation process has a contribution of over 90% to all environmental impact categories as well. The electricity consumption for the PBR is the main contributor during cultivation.

Table 2: Environmental results

Impact ^a	Unit	Basic	Intermediate	Advanced	Alternative
CC	ktonnes (CO ₂ to air)	308	161	217	226
OD	kg (CFC-11 to air)	26	13	16	16
TA	tonnes (SO ₂ to air)	1,393	580	694	677
FE	tonnes (P to freshwater)	132	37	72	70
ME	tonnes (N to marine water)	97	30	41	36
HT	ktonnes (14DCB to urban air)	166	47	63	57
POF	tonnes (NMVOC to air)	831	305	470	466
PMF	tonnes (PM ₁₀ to air)	627	213	255	237
TET	kg (14DCB to industrial soil)	31,779	12,271	6,949	5,407
FET	tonnes (14DCB to freshwater)	7,546	2,140	1,774	1,364
MET	tonnes (14DCB to marine water)	6,903	1,851	1,765	1,394
IR	ktonnes (U ²³⁵ to air)	54	30	377	420
ALO	ha year (agricultural land)	2,034	539	444	325
ULO	ha year (urban land)	464	172	122	103
NLT	m ² (natural land)	51,040	23,657	38,800	40,529
WD	dam ³ (water)	6,036	1,793	6,739	7,044
MD	tonnes (Fe)	37,983	9,639	5,978	3,449
FD	tonnes (oil)	85,463	47,612	59,934	63,178

^a CC: Climate change; OD: Ozone depletion; TA: Terrestrial acidification; HT: Human toxicity; POF: Photochemical oxidant formation; PMF: Particulate matter formation; FE: Freshwater eutrophication; ME: Marine Eutrophication; IR: Ionizing radiation; TET: Terrestrial ecotoxicity; FET: Freshwater ecotoxicity; MET: Marine ecotoxicity; ALO: Agricultural land occupation; ULO: Urban land occupation; NLT: Natural land transformation; WD: Water depletion; MD: Mineral resource depletion; FD: Fossil fuel depletion.

The relative importance of the different environmental impact categories to the overall environmental burden is illustrated in Figure 1. This relative importance was assessed by measuring

all the environmental impacts at the endpoint level. This way the different environmental impact categories which can be measured at the same endpoint level (i.e. Disability-adjusted life years (DALY), species year, \$) can be compared. Accordingly, the impact categories which contribute the most to the respective endpoint level can be identified. The impact categories which have the highest contribution are climate change (CC) and fossil fuel depletion (FD). The impacts of water depletion (WD) and marine eutrophication (ME) are not defined at the endpoint level and are therefore not included in Figure 1.

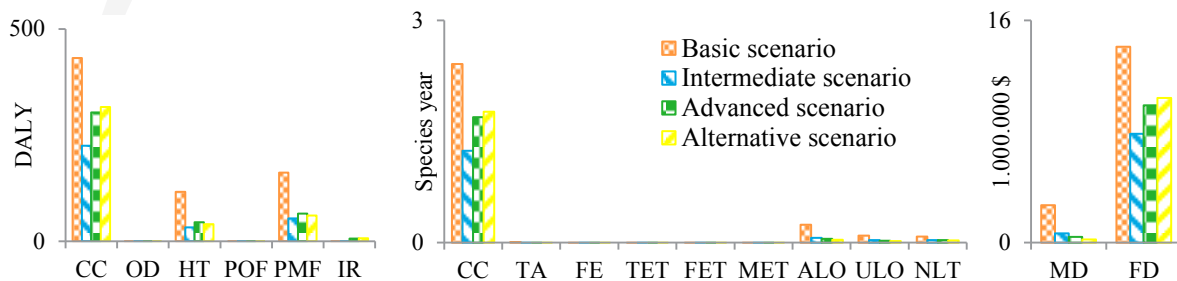


Figure 1: Relative importance of the environmental impact categories (CC: Climate change; OD: Ozone depletion; TA: Terrestrial acidification; HT: Human toxicity; POF: Photochemical oxidant formation; PMF: Particulate matter formation; FE: Freshwater eutrophication; ME: Marine Eutrophication; IR: Ionizing radiation; TET: Terrestrial ecotoxicity; FET: Freshwater ecotoxicity; MET: Marine ecotoxicity; ALO: Agricultural land occupation; ULO: Urban land occupation; NLT: Natural land transformation; WD: Water depletion; MD: Mineral resource depletion; FD: Fossil fuel depletion.)

Table 3 provides the economic results of the assessment over the entire lifetime (10 years). The basic scenario has the lowest NPV, due to the high operational costs. The medium recycling step, introduced in the intermediate scenario, drastically lowers the operational costs. This results in a positive NPV. The use of a PBR for cultivation increases the investment costs and the operational costs due to the high energy requirement during cultivation. Therefore, the advanced scenario is not economically viable. In the alternative scenario, the higher revenues, due to the higher price of astaxanthin compared to β -carotene, compensate for the higher investment and operational costs.

Table 3: Economic results

	Unit	Basic	Intermediate	Advanced	Alternative
NPV	€	-10,657,075	39,504,007	-3,059,254	22,578,733
Investment costs	€	11,135,402	10,728,447	42,330,148	45,275,638
Operational costs	€ year ⁻¹	16,803,355	7,124,789	11,197,953	11,003,314
Revenues	€ year ⁻¹	16,764,464	16,746,464	16,746,464	22,119,970

The main economic and environmental output categories are displayed in Figure 2. As the contribution of ME and WD at the endpoint level could not be assessed, they were also considered as main output categories. The intermediate scenario has the best score for all main output categories, i.e. combining the highest economic profitability with the lowest environmental impact. Although the alternative scenario has a high NPV, it also has a relatively high environmental impact. The basic scenario has the worst economic and environmental score, although the advanced scenario and the alternative scenario have a higher water depletion potential.

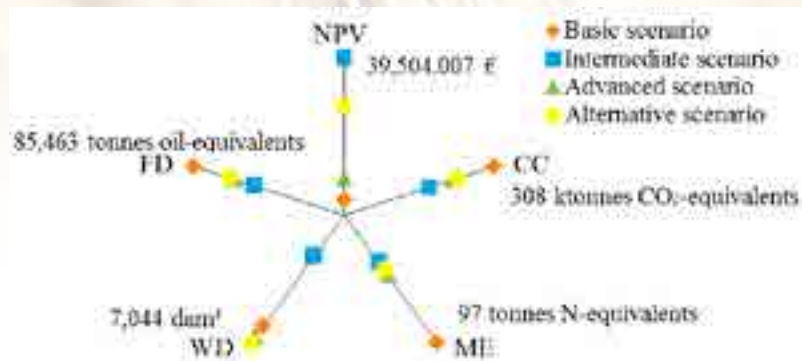


Figure 2: Main economic and environmental impact categories (NPV: Net present value; CC: Climate change; ME: Marine eutrophication; WD: Water depletion; FD: Fossil fuel depletion.)

Table 4 provides the parameters (economic, technical and environmental) with more than 10% impact on the main economic and main environmental output categories. The environmental output parameters were considered relative to the total production scale. The productivity was calculated based on the maximum concentration, the maximum specific growth rate of the species and the solar irradiation. These parameters are identified by almost all output categories as the most important parameters. Improving the productivity will therefore positively influence both the economic profitability and the environmental impact. For the economic profitability, the price of the carotenoid and the upscaling of the PBR are also important parameters. Other important parameters are the salinities of both cultivation stages. The difference between these salinities determines the amount of water and salt that can be recycled. The difference between the cultivation temperature and the surrounding temperature is also crucial as this influences the amount of heating that is required. The most important upstream environmental impacts are the impact of the electricity mix, the upstream impact of the water and the upstream impact of salt production.

5. Discussion

The sensitivity analysis illustrates that the main parameters to improve both the environmental impact and the economic profitability of an algal-based biorefinery are the productivity related parameters. The proposed biorefinery cultivates microalgae species with a relatively low productivity compared to other studies (Brennan and Owende, 2010). However, the selected species can accumulate large concentrations of high-value carotenoids. Using an alternative, faster growing species will therefore have multiple effects on the economic profitability of the biorefineries. The biomass production will increase, but the carotenoid concentration will decrease. The productivity is also influenced by regional characteristics, such as temperature and solar irradiation. A comparison with a biorefinery scenario in a warmer country is therefore an interesting field of further research.

The alternative scenario, i.e. the production of astaxanthin from *Haematococcus pluvialis*, was also assessed by Pérez-López et al. (2014). They analyzed the environmental effect of scaling up the process from a laboratory scale to a pilot scale. In their contribution analysis, they identified the electricity consumption as the main influence on the environmental impact. Pérez-López et al. (2014) only identified direct mass and energy flows as important parameters, therefore, the parameters related to the productivity could not be identified as crucial.

The selection of the functional unit is an important consideration during the interpretation of the results. In our study, the total environmental impact over the entire lifetime of the production process was used as the functional unit. If the mass of carotenoids would be the functional unit, the environmental impact of the alternative scenario would be much larger compared to the advanced scenario, as a smaller amount of carotenoids is produced in the alternative scenario. If the cost of carotenoids would be the functional unit, the alternative scenario would have a lower environmental impact than the advanced scenario, as the revenues from astaxanthin are much higher than the

revenues from β -carotene. Moreover, with these functional units, the sensitivity analysis would identify the carotenoid content and price as crucial parameters.

6. Conclusions

This paper performs an environmental techno-economic assessment of four algae-based biorefinery scenarios. The parameters related to the productivity are identified as crucial parameters for both the economic and environmental feasibility of the project. The use of a medium recycling step can reduce the environmental impact and increase the economic profitability of the algal-based biorefinery. The environmental techno-economic assessment provides a methodology to enable an integrated assessment of the technological, economic and environmental feasibility of a new technology. Therefore, it can act as guidance during technology development of new and innovative technologies in the biobased economy.

Table 4: Results sensitivity analysis

Parameter ^a	Basic scenario					Intermediate scenario					Advanced scenario					Alternative scenario					
	NPV	CC	FD	ME	WD	NPV	CC	FD	ME	WD	NPV	CC	FD	ME	WD	NPV	CC	FD	ME	WD	
Max. conc.	+23%	-18%	-13%	-42%	-24%	+19%	+37%														
Price carotenoid	+19%					+20%															
Carotenoid cont.	+20%																				
PBR sizing factor		+29%	+34%					+41%													
T. Belgium							-10%	-12%													
Spec. growth rate																					
Solar irr.																					
Imp. electricity				+13%	+14%																
Imp. salt				+14%	+12%																
Salinity stage 2									+41%	+39%											
Salinity stage 1									-27%	-26%											
Imp. wastewater					+11%																
Imp. water					+13%																

^a Abbreviations = Max. conc.: Maximum concentration; cont.: content; T.: Temperature; Spec.: Specific; irr.: irradiation; Imp.: Impact; NPV: Net present value; CC: Climate change; FD: Fossil fuel depletion; ME: Marine eutrophication; WD: Water depletion.

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296. Life Cycle Sustainability Assessment of Convenience Food: Ready-made Meals

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ABSTRACT

This study analyses the environmental, economic and social sustainability in the UK ready-made meal sector using a life cycle approach. Life cycle assessment, life cycle costing and social life cycle assessment have been used for these purposes. Global warming potential (GWP) at the sectoral level is estimated at 4.94 Mt CO₂ eq./yr from 'cradle to grave'. Excluding the consumption stage to consider the impact from 'cradle to retail', GWP is equal to 4.45 Mt CO₂ eq./yr, which represents 3% of the GHG emissions from the food and drink sector and 1% of the UK total emissions. The annual life cycle costs are estimated at £872 million from 'cradle to grave' and at £755 million from 'cradle to retail'. By comparison, the estimated market value is £1,974 m suggesting a significant disparity between the costs and profits. From the social sustainability perspective, wages and forced labour are found to be critical issues associated with the agricultural stage; worker injuries and fatalities in the production of meals; and food security and health for consumers. This research identifies places in the supply chain where the sustainability issues should be addressed by relevant stakeholders to improve the sustainability in the ready-made meals sector.

Keywords: Life cycle assessment, life cycle costing, social life cycle assessment, life cycle sustainability assessment, ready-made meals

1. Introduction

Ready-made meals are an essential part of British life, with the UK sector leading the European market (Winterman, 2013) and occupying the second place worldwide, after the US (Sheely, 2008). In 2013, the UK ready-made meals market was valued at £1.97 bn (Key Note 2013; CFA 2014) and it is expected to grow by a further 20% by 2017 (Key Note 2013).

The quest for more sustainable solutions within the food industry has led to numerous environmental studies, but most focussing on single food items and usually only greenhouse gas (GHG) emissions. As the same time, socio-economic issues, such as food affordability, health and nutritional quality are becoming increasingly more important to the consumer. Despite this and a growing interest in environmental impacts, studies considering life cycle environmental, economic and social aspects of different foods and food sectors are scant. Therefore, this study seeks to analyse the life cycle sustainability of an important growing market within the British food industry: the ready-made meals sector.

2. Goals and Scope

The goal of the study is to assess the environmental, economic and social sustainability in the UK ready-made meals sector on a life cycle basis. For these purposes, 13 most consumed meals, representing 85% of the market sales by value in the UK, are considered across the following four cuisines: English, Italian, Indian and Chinese. Both chilled and frozen meals are evaluated.

The sustainability assessment has been carried out using life cycle assessment (LCA) in accordance with the ISO standards (2006a, 2006b). Eleven environmental impacts have been estimated following the CML 2001 impact assessment method (Guinée et al., 2001). The methodology proposed by Hunkeler et al. (2008) and Swarr et al. (2011) has been applied to estimate life cycle costs (LCC) while social LCA (S-LCA) has been carried out following the UNEP and SETAC (2013) guidelines. The scope of the study is from 'cradle to grave', comprising the whole supply chain, including agriculture, manufacture, consumption and post-consumer waste management. The functional unit is defined as the 'annual sales of ready-made meals in the UK', equal to 483,100 kt in 2013.

3. Life Cycle Inventory

Figure 1 outlines the 13 meals considered and the share of different cuisines according to retailer information obtained as part of this research. First, LCA and LCC have been carried out at the level of individual meals following the approach detailed in Schmidt Rivera et al. (2014) and Schmidt Rivera and Azapagic (2016), respectively. The results have then been scaled-up to the sectoral level based on the market share and sales volume (Table 1). For S-LCA, the social sustainability indicators considered at the sectoral level are wages, working hours, forced labour, fatal injuries, food security and human health issues. These indicators were chosen based on the following two criteria: the representativeness of critical social issues for the sector and data availability, not only at the sectoral level, but also across the supply chains.

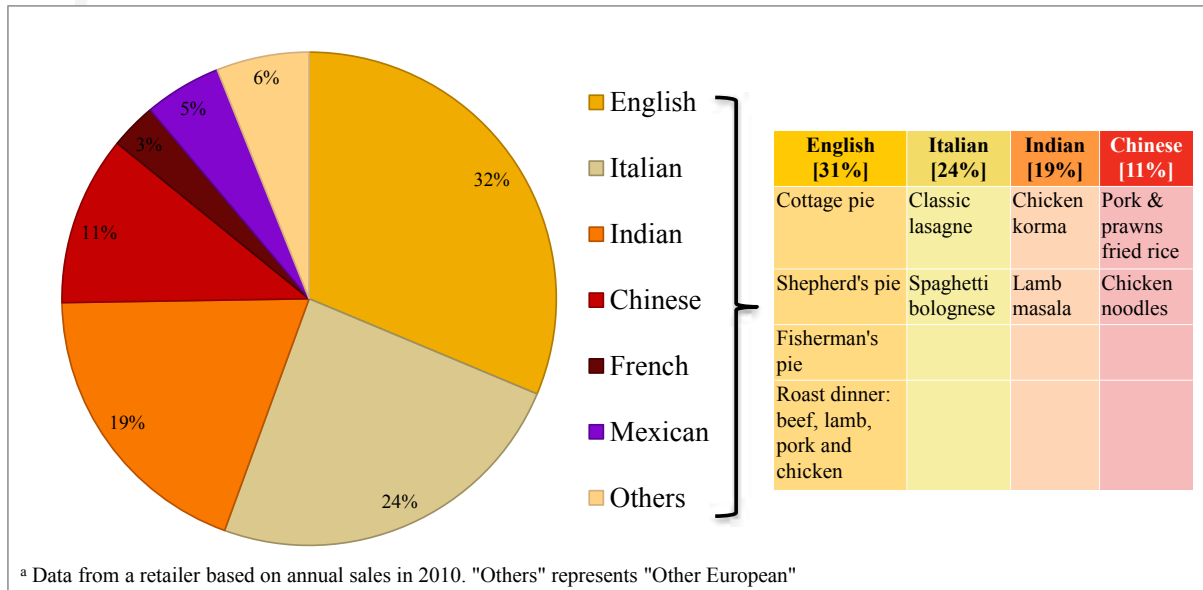


Figure 1 Market share of different cuisines and representative ready-made meals considered in the study.

Table 1 Market share of different cuisines and sales volume of ready-made meals in 2013 (Key Note 2013)

Parameter	Sub-sector	English	Italian	Indian	Chinese	Total
Cuisine share	Chilled & frozen	31%	24%	19%	11%	85%
Volume	Chilled (kt)	119.8	92.7	73.4	42.5	328.5
	Frozen (kt)	56.4	43.6	34.6	20.0	154.6
	Frozen (£m/yr)	113.8	86.2	65.4	28.9	294.3

4. Results and Discussion

Figure 2 shows the results for GWP as an example of LCA impacts for different system boundaries; the breakdown for the chilled and frozen sub-sectors is given in Table 2. GWP is estimated at 4.94 Mt CO₂ eq./yr from 'cradle to grave' with the chilled meals sub-sector being the main contributor (71%). This is due to the lower efficiency of the chilled supply chain, using higher amounts of refrigerants and generating more waste along the life cycle. Excluding the consumption stage, the GWP from 'cradle to retail' is equivalent to 4.45 Mt CO₂ eq./yr, which represents ~3% of the GHG emissions from the food and drink sector and ~1% of the total UK emissions.

The other impacts follow similar trends to GWP, except for ozone layer depletion potential, where the chilled market is the main contributor with 96%. This is due to the lower efficiency of the cold chain compared to the frozen (open vs closed cabinets) and the refrigerant use (R314 vs ammonia).

As also indicated in Figure 2, the annual life cycle costs are estimated at £872 million from ‘cradle to grave’ and at £755 million from ‘cradle to retail’; the latter is 62% lower than the estimated market value of £1,974 m. There could be many reasons for this difference, including the data and assumptions, as well as various taxes along the supply chain, but even so, the results suggest relatively high profit margins in the sector (see Figure 2). However, it is not clear how the profits are distributed across the supply chain and who benefits the most.

From the social sustainability perspective (see Table 3), the agricultural stage represents a very high risk for low wages, high working hours and forced labour. These are related to the seasonal characteristics of agricultural activities and a general lack of regulation of the labour market in this sub-sector. A similar situation is found in the wholesale part of the supply chain where the turnover of personnel, long opening hours and short-term contracts put this activity at a very high risk of low wages and a medium risk of high working hours and fatalities. Finally, manufacturing and transport have a very high risk of fatalities, with the former being due to the highly industrialised processing chain and the latter because of the risks associated with driving.

For consumers, critical social sustainability issues include food security and health. In the past few years food prices have risen, decreasing food affordability; for instance, in 2012 people spent 17% more on food compared to 2007, but buying 4.7% less (Cooper et al., 2014). Moreover, since 2007 there has been a 22% increase in food prices in the UK, almost double (12%) that in other EU countries (Defra, 2014). Consequently, the health and wellbeing of the population, in particular those on lower incomes, have been deteriorating, with around a million Britons using food banks (Milligan, 2014) and, at the other end of the scale, half of the population being obese and overweight (HSCIC, 2013).

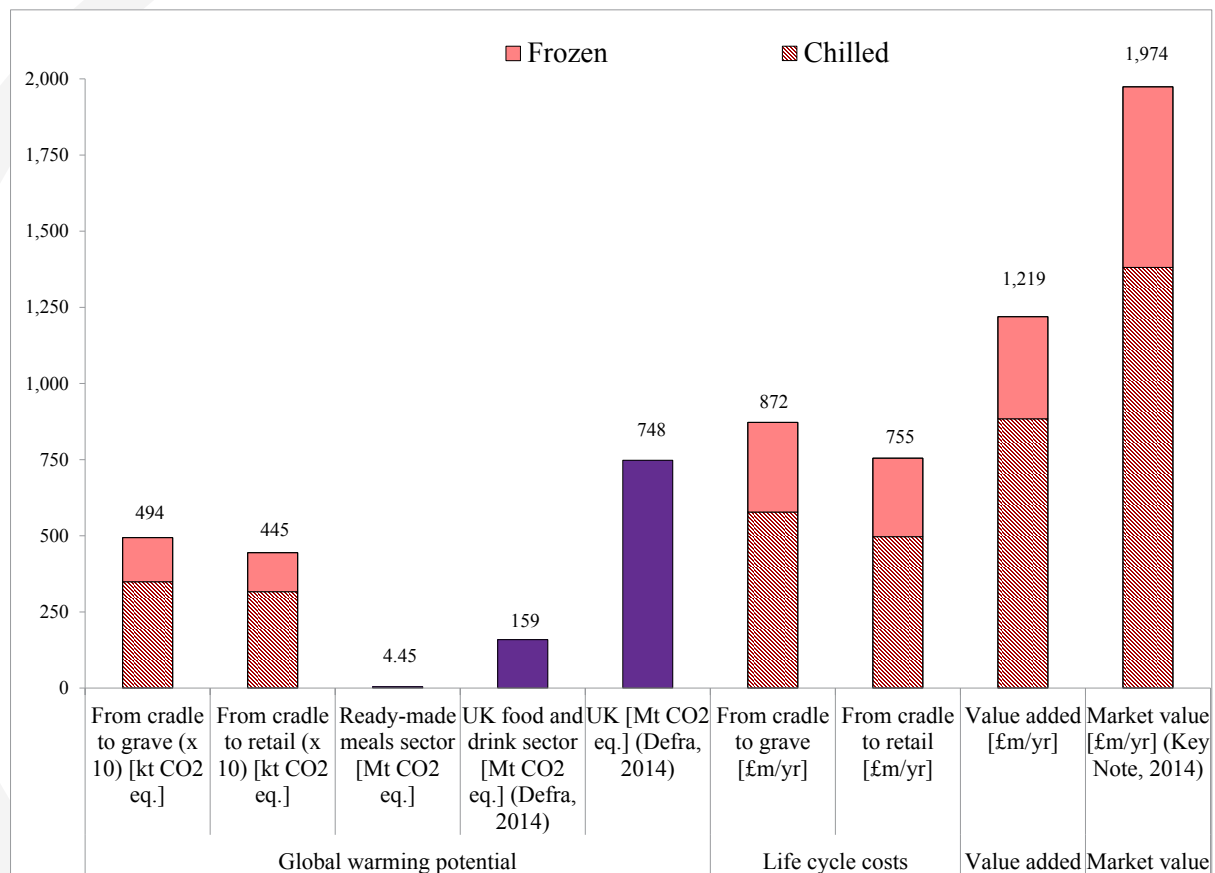


Figure 2 Global warming potential (GWP) and life cycle cost (LCC) in the UK ready-made meals sector

Table 2 Global warming potential (GWP) and life cycle cost (LCC) at the sectoral level showing the breakdown for different cuisines

Parameter	Sub-sector	English	Italian	Indian	Chinese	Total
GWP ^a	Chilled (kt CO ₂ eq./yr)	1186.1	1189.0	797.1	317.8	3490.0
	Frozen (kt CO ₂ eq./yr)	489.2	503.6	332.2	126.0	1450.9
LCC ^a	Chilled (£m/yr)	223.6	169.9	128.6	55.9	578.1
	Frozen (£m/yr)	113.8	86.2	65.4	28.9	294.3

^a Values calculated for each meal first and then scaled-up to the sectoral level.

Table 3 Social hotspots for employees in the ready-made meals supply chain

Social indicators	UK, FDS & RMMS ^a	Agriculture	Manufacturing	Retail	Transport & storage
Risk of low wages ^b					
Risk of high working hours ^b					
Risk of forced labour ^c					
Risk of fatal injuries ^b					

Very high	High	Medium	Low	No data

^a FDS: food and drink sector; RMMS: ready-made meals sector.

^b Data from ILO (2014).

^c Data from Skrivankova (2014) and Dugan (2013).

5. Conclusions

This study is the first attempt to analyse the sustainability in the UK ready-made meal market, integrating environmental, economic and social aspects. The results reveal that the ready-made meal sector contributes ~3% to the GHG emissions from the food and drink sector and ~1% to the national emissions. The chilled meals sub-sector contributes the vast majority of the total GWP (71%), which is not only due to the higher market share but also due to the less efficient chilled chains in terms of the use of refrigerants and waste generation.

From an economic perspective, the study also shows the value added generated by the sector and the contribution of different parts of the supply chain. Furthermore, the research highlights the critical social sustainability issues within the sector and how they affect different stakeholders in the supply chain, identifying where interventions are needed to improve the overall sustainability in the ready-made meals sector.

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30. Potential environmental and economic benefits from local food production in Mediterranean rooftop greenhouses

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ABSTRACT

Due to growth population and its concentration in cities, local resources and food production will be subjected of political and social concern. Food security will be a key strategy to ensure societies stability in the future. Social urban agriculture is known as a key strategy to ensure feeding urban areas. Also, a new technological and business model of urban agricultural, which is gaining popularity, is taking place in cities. One of this multiple new models consist on the named Integrated Rooftop Greenhouses (i-RTGs) that are ecoinnovative systems, which allow food production on the top of building to improve environmentally and economically crops and buildings efficiency. To increase both systems efficiency, i-RTGs allow a water, energy and CO₂ flow exchange between buildings and rooftop. This article focuses on the specific case study of a i-RTG (laboratory scale), called the i-RTG-Lab used to determine the potential environmental and economic benefits of integrating greenhouses with buildings and exchange water, energy and CO₂ flows within the framework of the Fertilecity project. The study presents main data collected during the first experiment of the project, for the elaboration of a life cycle inventory, regarding the flow exchanges between the i-RTG-Lab and the building. Moreover, it provides a first approximation of the potential environmental and economic benefits that such systems could provide in comparison with non-integrated roof top greenhouse. Main results show that i-RTGs has the potential to avoid 99.8kg CO₂ eq. /m²-year and save 20.02€/m²-year, in comparison with a nonintegrated RTG, and the capacity to produce during 6-month summer crops 16.2kg/m².

Keywords: Rooftop Greenhouse; Urban Agriculture; Building-integrated agriculture; Food self-sufficiency, Industrial Ecology, Water-Food-Energy Nexus

1. Introduction

World population has been foreseen to reach 9.550 million habitants by 2050(UN, 2012). By then, more than the 70% of this population will live in cities. Cities concentrate the highest levels of population density. Higher population densities reduce land occupation areas; however, it has associated an increase of input and output flows between cities and its surroundings. If world population keeps growing an overexploitation of local supply services and resources will occur, resulting in a scarcity of fresh water and food supply, which may cause several social and political conflicts (Allouche, 2011).

Urban agriculture (UA) has a great potential to increase cities food self-sufficiency(Orsini et al., 2013; E Sanyé-Mengual et al., 2015; Specht et al., 2013); improve the environmental performance of actual cities' feeding systems (Esther Sanyé-Mengual et al., 2015b; Tomlinson, 2011) and provide several social benefits (Orsini et al., 2013; Esther Sanyé-Mengual et al., 2015a; Specht et al., 2013). Nowadays, different agriculture typologies have already been developed in urban areas, all of them subsumed within the term UA. Vertical farming is a subcategory within this concept which includes all those urban agriculture developed in new or retrofitted buildings (Despommier, 2008). Vertical farming encompasses buildings specifically oriented to food production and buildings that combine housing with food production. This paper specifically focuses on Rooftop Greenhouses (RTGs), a specific typology of vertical farming systems which consist on installing greenhouses on the roof of buildings.

Inspired in an industrial ecology approach, RTGs can integrate their flows with the metabolism of the buildings they are placed in. Integrated RTGs can exchange energy, water and CO₂ flows to optimize both the building and greenhouse efficiency (Figure 1). This approach is the basis of the integrated Rooftop Greenhouse Lab (i-RTG-Lab), a research-oriented i-RTG in Bellaterra (Barcelona, Spain) placed on the rooftop of the ICTA-ICP building (Latitude 41°29'51.6"N; Longitude 2°06'31.9"E). The i-RTG-Lab is the case study of the "Fertilecity" project¹. The project aims to demonstrate the technical, environmental and economic feasibility of producing food in i-RTGs in Mediterranean cities. The present paper collects the first data from energetic, CO₂ and water flows collected for the elaboration of a Life Cycle Inventory (LCI) of the i-RTG-Lab and discusses the potential environmental and economic benefits of integrating RTGs in contrast with conventional RTG.

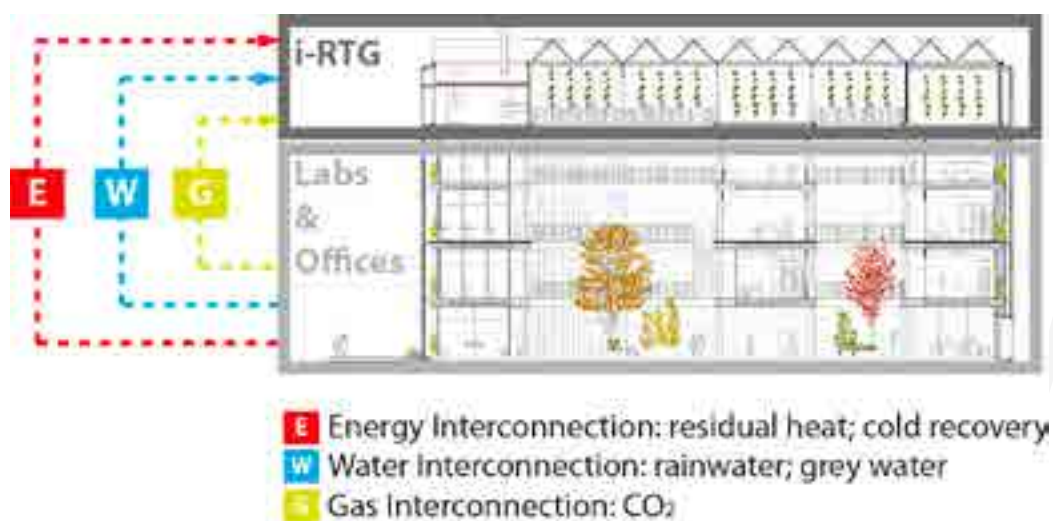


Figure 1. Exchange of flows between the new ICTA-ICP building and the i-RTG.

¹www.fertilecity.com/en

2. LCI method

2.1. Crop description

The i-RTG-Lab consist on a greenhouse with South-East orientation and a total area of 122.8 m², from which 75 m² are dedicated to food production (Figure 1), in particular tomato production (beef tomato type). During the project 3 tomato crops (between February 2015 and July 2016) will be done to strengthen results, compare crops done in different seasons and improve resource efficiency of food production, according to the requirements of the specific climate conditions and crop of the i-RTG-Lab. Results shown in the present publication refer to the data collected during the first crop of the project (February - July 2015).



Figure 2. Left: ICTA-ICP building (Latitude 41°29'51.6"N; Longitude 2°06'31.9"E). Right: second tomato crop done in the i-RTG-LAB.

2.2. Water flows

The interconnection between the building and the i-RTG-Lab regarding water flows includes the use of rainwater for the irrigation of crops. The ICTA building collects rainwater and stores it in water tanks with a total capacity of 135 m³. This water is used both for the irrigation of crops and ornamental plants and for toilets and washing machines. The use of rainwater together with water efficiency measures implies a significant reduction in the consumption of water in the building.

This synergy has been studied from a quantitative and qualitative perspective. Flow meters have been used to measure the quantity of water consumed for each use. Additionally, analysis have been made periodically for the assessment of water samples, especially for determining the quality of rainwater delivered for crop irrigation and the leachates disposed (nitrates, phosphates, etc.).

Avoided emission quantified does not include emissions due to added infrastructure (i.e. water tank). Only emissions from the avoided use of tap water were quantified. For the costs, only were quantified savings from tap water that was not consumed too.

2.3. Energy and CO₂ flows

The ICTA-ICP climate system continuously recirculates office and lab air to reduce energy consumption from conditioning outside air to the ideal temperatures required in the spaces by its users. However, the CO₂ concentration of this air is continuously increasing due to human respiration. For this reason, CO₂ concentrations in offices and labs is monitored. When the CO₂ sensors detect that concentrations in spaces are too high, the climate system automatically injects fresh air in the spaces and ejects waste air with high CO₂ concentrations. The waste air, with an ideal temperature of 20-24°C can be injected to the RTG to cool greenhouse daily temperatures and warm the greenhouse during the night, while providing air rich in CO₂ that stimulates crop productions.

To quantify the energy savings through flow exchanges between the building and the i-RTG-Lab, the following parameters are measured: in and outside i-RTG-Lab temperatures, building temperatures and waste air temperatures. For CO₂ flows, the amount of CO₂ provided through waste air from the building to the greenhouse is measured with a CO₂ sensor and a flowmeter. The building has 4 different outlets for the waste air. In particular, the outlet that inject waste air to the i-RTG-Lab

collects exclusively waste air from the laboratories from 4th floor. Moreover, at the end of the tomato crop the amount of CO₂ emissions fixed by plants were quantified.

2.4. Avoided GWP and potential economic savings

For water and energy flows the global warming potential (GWP – kg CO₂ eq.) and economic savings of the i-RTG-Lab are calculated through LCA (ISO, 2006) and costs assessment (ISO, 2008) methodologies as a first approach and compared with a non-integrated RTG. This estimation includes: (1) quantification of the environmental and economic advantages of using rainwater to irrigate the crop, without including the equipment (i.e., rainwater tanks) installation costs and (2) determining the advantages of using waste air and thermal inertia of the building to warm the greenhouse when both systems are integrated. This energetic benefits were calculated through building's thermal simulation systems.

2.5. Reference system: conventional RTGS

A non-integrated RTG was used as reference system to describe the potential water, energy and CO₂ saving of the i-RTG provided in present research. For the non-integrated RTG following assumption were used:

- Water is specifically provided though tap water. None rainwater is used.
- No waste energy from the building is used to heat de RTG.
- No waste air from the building, with high CO₂ concentrations, is provided to the crop.

3. LCI results& discussion

According to results shown in table 1, due to the exchanges of water and energy flows, 20,02€/m² and 99.8kg CO₂ eq./m² could be saved each year with the integration of RTGs with buildings in comparison with a nonintegrated RTG.

During the first 6-month crop conducted, a production of 16.2kg of tomatoes per m² was obtained. Regarding water savings, for this crop, 60% of water used for the irrigation of tomatoes is rainwater. In absolute terms, around 1.1m³/m²·year of rainwater could be saved.

Energy savings (Table 1) are mainly obtained through the thermal inertia of the building, which enables heating the i-RTG-Lab during cooler periods. The building has the capacity to store large amount of thermal inertia during the day due to its high mass. If the i-RTG-Lab is properly managed, building thermal inertia can be provided to the greenhouse and keep minimum temperatures during winter above 14°C without using climate systems (figure 3). However, heat provided through the waste air of the building, described in the methodology, does not seem to provide enough heat to warm significantly the greenhouse.

Water flows	
Description	Rainwater from the building to the RTG
Water savings	80 m ³ /year
Avoided GWP (kg CO₂ eq.)	0.37 kg CO ₂ eq./m ² · year
Potential economic saving (€)	3.5€/m ² · year
Energy flows	
Description	Waste air from the building + building thermal inertia to the greenhouse
Energy savings	387 kWh/m ² · year
Avoided GWP	99.4kg CO ₂ eq./m ² · year
Potential economic saving (€)	19.65€/m ² · year
CO₂ flows	
Description	Waste air from the building to the greenhouse
CO₂ flow	42 kg CO ₂ /year
CO₂ fixed by crop	1.9 kg CO ₂ /m ² · year

Table 1. Potential annual environmental and economic savings from water, energy and CO₂ flows in the i-RTG-Lab in comparison with a non-integrated RTG.

According to the crop area of 75 m² and values from table 1, the i-RTG-Lab has the potential to fix 7,485 kg CO₂ eq./year. Then, the crop has the potential to absorb 3.4 times the amount of CO₂ that could be injected to the i-RTG-Lab through the waste air of the building which is 42 kg CO₂/year (table 1). Therefore, the crop contributes to reduce direct CO₂ emissions from the building due to users' respiration.

Moreover, according to Sanyè et al. study, tomato production in urban RTGs could avoid annually indirect 6.61 CO₂ eq./m² due to the simplification of logistics in urban agriculture (Sanyé-Mengual et al., 2013). Mainly because of the reduction of: transportation distances, packaging use and food losses. Therefore, if avoided emissions of food production in i-RTGs due to the simplification of logistics and the exchange of flows are added, i-RTGs has the potential to save 106.4 kg CO₂ eq./m² per year.

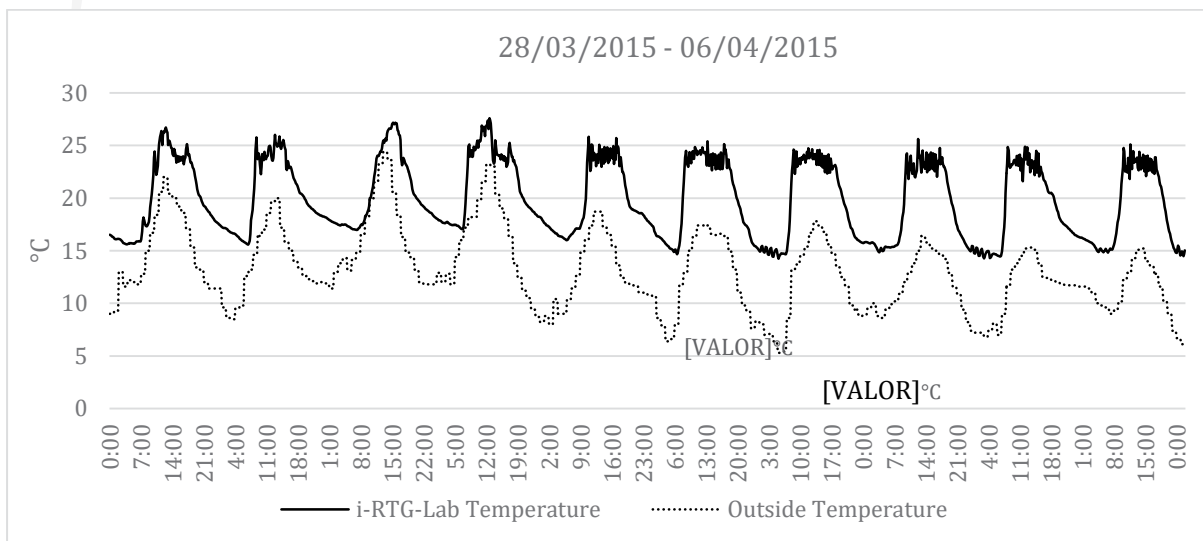


Figure 3. Comparison of inner and outside temperatures of the i-RTG-Lab during a cool period.

5. Conclusions (or Interpretation)

- Thermal benefits during cooler periods of the i-RTG-Lab due to the high thermal inertia of the building, which allow producing food without using extra heating systems, could be a key issue to ensure the environmental and economic benefits of i-RTGs by saving annually 0.2 kg CO₂ eq./m² and 33.3€ / m².
- The greatest environmental benefits of urban i-RTGs may be provided by the simplification of distribution logistics, which could reduce indirect emissions by 6.61 kg CO₂ eq./m² during one year.
- The integration of RTG with buildings and the exchange of flows between both systems could help to save annually 2.47 kg CO₂ eq./m² and 36.8 €/m² for tomato production.
- Results provided represent the environmental and economic potential benefits of i-RTGs. These results are based on data from the first crop developed within the Fertilecity project (February 2015-July 2015). At the end of the 3rd crop of the project (July 2016) more data will be available to strength results.
- Further research is required to look for other potential flows between the i-RTG and the building that could help to increase both system efficiencies, such as: (1) study the potential to export daily waste heat from the i-RTG during cool periods to the cooler zones of the bottom of the building; or (2) analyze the viability of using crop leachates for building purposes.

6. Acknowledgements

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172. Could cellular agriculture reduce environmental impacts?

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ABSTRACT

Cellular agriculture means production of agricultural products by cultivating cells in industrial settings instead of growing crops and livestock on farms. In this paper, we review the existing literature on environmental impacts of cultured meat, and present first estimates of land occupation, water resource depletion, fossil resource depletion and global warming potential (GWP) of yeast-derived milk (YDM). Previous studies have shown that large-scale production of cultured meat can potentially have lower carbon and water footprint and require substantially less land than conventionally produced meat, especially beef. However, energy input requirements may be higher. The preliminary results presented in this paper suggest that YDM has lower land use and water resource depletion compared to conventionally produced milk, and lower GWP and fossil resource depletion when low emission energy sources are used. As the products of cellular agriculture are still at the development stage, the uncertainty of the results is high. Regardless of the uncertainty, it can be concluded that cellular agriculture has the potential to substantially reduce negative environmental impact of agriculture.

Keywords: cultured meat, livestock products, milk, life cycle assessment, carbon footprint

1. Introduction

Food systems are facing the challenge of securing adequate and sustainable nutrition to the increasing world's population under the changing environmental conditions. Climate change together with reduced fresh water availability, limited land resources, changes in quality of soils, air and water, and depletion of mineral resources have been predicted to have substantial impacts on agricultural production, nutrition and human health during the next decades (Whitmee et al. 2015).

Agriculture, especially livestock farming, is also a major contributor to environmental change. Livestock production alone has been estimated to contribute 15% of global greenhouse gas (GHG) emissions (Gerber et al. 2013), 33% of the global land use (FAO 2006) and 27% of global fresh water use (Mekonnen & Hoekstra 2011). Furthermore, livestock production is one of the main drivers of deforestation due to land clearance for feed production.

New agricultural and food production technologies that provide solutions both to adaptation to environmental change and reduction of environmental impacts of agriculture are needed. Cellular agriculture, which uses modern technologies for cultivating agricultural products in industrial settings, has potential to meet both targets. The closed systems provide secure production conditions that are not directly exposed to environmental changes. The emissions to air, water and soil can be controlled more efficiently than from open agricultural systems. Cellular agriculture is a novel rapidly developing field. The applications currently under development include cultured meat (i.e. in vitro meat or lab-grown meat), yeast-derived milk (YDM) and eggs, animal-free gelatin and lab-grown rhino horns.

In this paper we review the existing life cycle assessments (LCA) of cultured meat and present the preliminary estimates of water resource depletion, land use, fossil resource depletion and global warming potential (GWP) of YDM. Finally, we discuss the potential of cellular agriculture to contribute to the reduction of negative environmental impacts in the future.

2. Review of LCA studies of cultured meat

To date, two peer-reviewed journal papers (Mattick et al. 2014 and Tuomisto & Teixeira de Mattos 2011) and a conference paper (Tuomisto et al. 2014) have estimated the environmental impacts of cultured meat. Tuomisto & Teixeira de Mattos (2011) modelled the impacts of large-scale cultured meat production and found that cultured meat could have substantially lower GHG emissions, water footprint (including blue, green and grey water) and land use when compared to conventionally produced meat, whereas energy use was higher than for poultry production, but lower than for pork, beef and lamb. In the study, it was assumed that cyanobacteria hydrolysate was used as feedstock for the cell culturing, and 100% of the impacts of producing the animals were allocated to the edible parts of the animal.

Tuomisto et al. (2014) amended the previous LCA study by using a new system design and different choices in the LCA methodology. Economic allocation was used for allocating the impacts of livestock production to the co-products (i.e. 90% of the impact of producing the whole animal was allocated to the edible part). The water footprint method was changed to include only blue water with country specific scarcity indexes. The system design was modified by assuming a use of a hollow fiber bioreactor instead of stirred-tank bioreactor and various feedstock options for the cell culturing were considered. As a result of the adopted modifications, the absolute results of the environmental impacts of cultured meat increased compared to the previous study. Furthermore, the relative difference of the impacts between cultured meat and conventionally produced meat reduced. The results showed that energy use was higher than for conventionally produced meat, whereas GHG emissions, water footprint and land use were lower (except water footprint was higher than that of conventionally produced poultry).

Mattick et al. (2014) modelled the impacts of cultured meat production in the United States (US) and found that cultured meat has higher energy use than conventionally produced meat, higher GHG emissions compared to poultry and pork, and lower land use and eutrophication potential. The absolute impacts were higher than those found in the earlier studies (Figure 1). The differences were caused by the differences in the assumptions, such as feedstock ingredients, bioreactor design and scaffold material used.

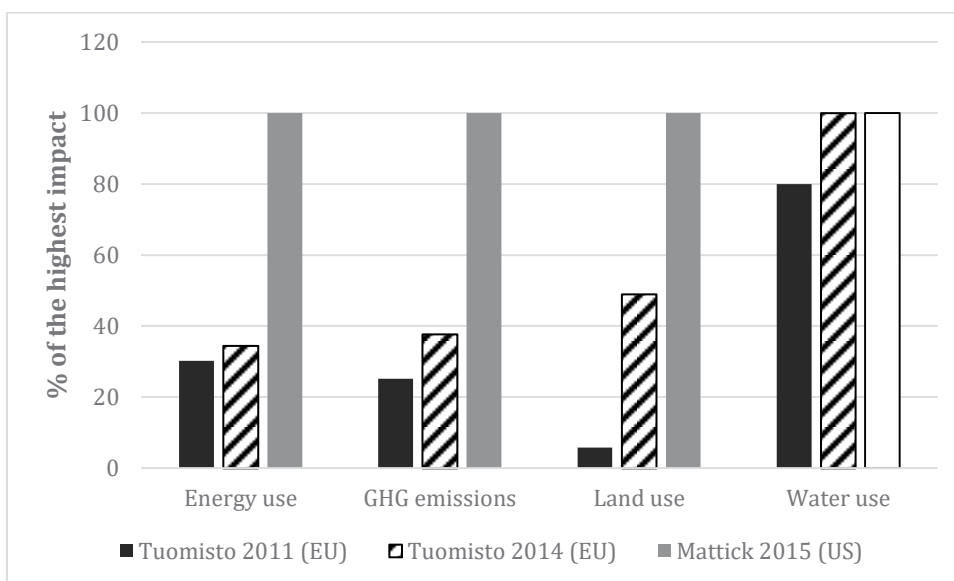


Figure 1: Comparison of LCA studies of cultured meat (white block = no data).

3. Environmental impacts of yeast-derived milk

3.1 Methods

A LCA based land occupation, water depletion, fossil depletion and global warming potential (GWP) of yeast-derived milk (YDM) were estimated and compared with conventionally produced milk. The functional unit was one kg of energy and protein corrected milk leaving the factory gate. The system boundaries were from cradle-to-factory gate as it was assumed that the processes after the products leave the production facility are indifferent from those of conventionally produced milk. The processes included were: production of input materials, production of feedstock, culturing of yeast protein, mixing the ingredients included in the final product and waste management. The manufacturing, maintenance and disposal of infrastructure were excluded. The production was assumed to take place in the US.

The yeast-derived milk is produced by synthesizing yeast protein in a bioreactor in a nutrition solution consisting of sucrose and various micronutrients. After the culturing process, the yeast protein is extracted by using filtration, and it is mixed with coconut oil and water. The precise recipe is still under development. For this study, the ingredient list was obtained from the researcher developing the YDM production technology. Due to the confidentiality of the information, the detailed ingredient list is not presented here. The bioreactor energy use for the yeast protein culturing were modelled based on the data for the growth condition requirements.

The life cycle assessment was carried out in SimaPro 8.0 by using Ecoinvent database and Recipe Midpoint method. The results were compared with conventionally produced milk based on data found in Ecoinvent database. It was assumed that wind power was used as source of electricity for running the bioreactor.

3.2 Results

The preliminary results suggest that YDM has lower land use, water resource depletion, fossil resource depletion and GWP when compared to conventionally produced milk (Figure 2). The contributions of the different processes to the total land use, water resource depletion, energy use and GWP are presented in Figure 3. The production of sugar has the highest contributions to land use, whereas the production of micronutrients is the main contributor to water resource depletion and GWP. Bioreactor energy use has the highest contribution to the total energy use, but due to the assumption of using wind energy as a source of electricity, the contribution of bioreactor on GWP is relatively low.

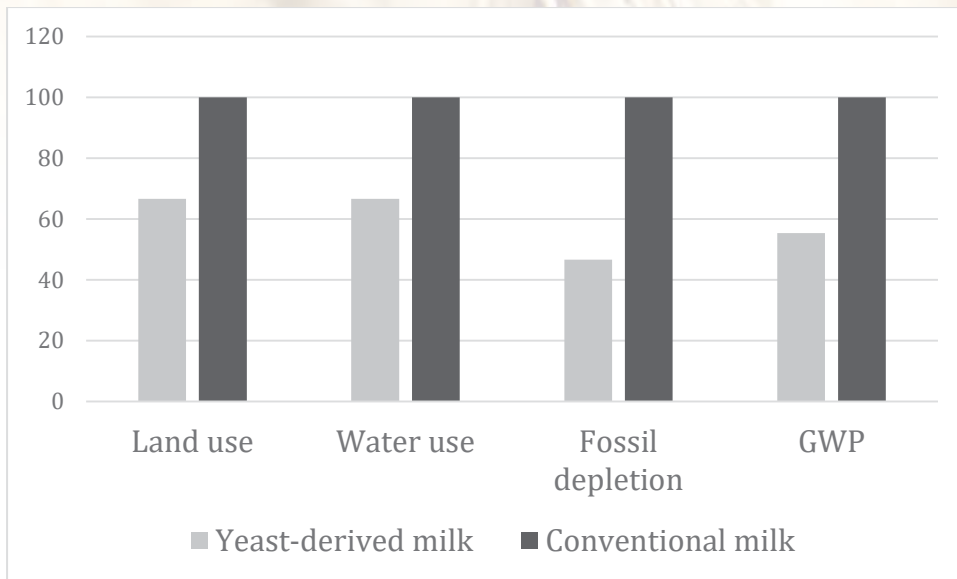


Figure 2: The relative results of land occupation, water resource depletion, fossil resource depletion and global warming potential (GWP) of producing 1 kg yeast-derived milk (YDM) compared to data for conventional milk production.

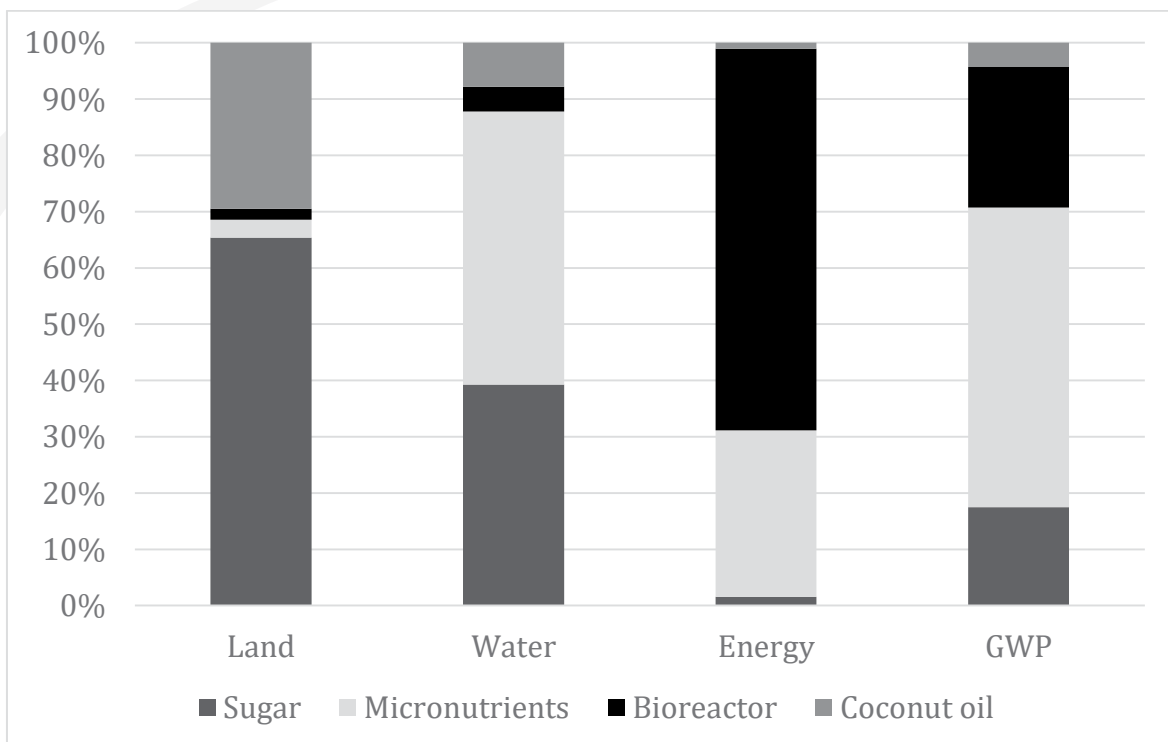


Figure 3: Contribution of different processes to the total impacts of yeast-derived milk production.

3.3 Discussion

While these results must be treated as preliminary estimates and subject to change, they provide an indication that YDM production systems can provide efficiencies in terms of land and water use, and can help to reduce GHG emissions provided that low emission energy sources are used in the production process. The efficiencies of YDM compared to conventional milk production are mainly

due to higher feed-to-milk conversion ratio and avoidance of methane emissions that are caused by enteric fermentation and manure management in cattle farming.

The main uncertainties in the current analysis are related to the bioreactor energy consumption and composition of the growth media used for the yeast protein culturing. The optimization of the current process design can lead in substantial reduction in the energy use requirements and environmental impacts.

4. Conclusions

The LCA studies of cellular agricultural systems have to rely on high assumptions as large-scale production facilities do not currently exist. The main sources of uncertainty are related to the data quality, and assumptions regarding the ingredients of the growth media and the bioreactor energy requirements. Different methodological choices used in the LCA studies can also have a high impact on the results.

The estimates of the environmental impacts of large-scale cellular agricultural systems could be improved by obtaining higher quality data of the production processes by using system modelling or collecting data from industries that use similar processes for other applications (e.g. from pharmaceutical companies). Future research should also include wider range of environmental impact categories and test the impact of different LCA assessment approaches, such as attributional vs. consequential modelling.

Regardless of the high uncertainty, it can be concluded that cellular agriculture has a potential to reduce negative environmental impacts compared to livestock farming. High energy requirements in cellular agriculture could be compensated by developing production systems that incorporate renewable energy production at the site (e.g. wind power, solar panels or anaerobic digesters).

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270. Life cycle assessment of meals based on vegetarian protein sources

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ABSTRACT

This study investigates the environmental impacts of vegan and vegetarian food products which are rich in vegetable proteins from plants or fungi and meals prepared with such products. The main goal of the study was to provide reliable life cycle inventory data for this type of products and to assess the impacts for single ingredients and meals prepared with such products and further ingredients. This pilot study shows that the impact of vegetable protein products can only be comprehensively assessed when considering the later usage in a meal that is ready for consumption. The results of this study for some individual examples still do not allow general statements for the evaluation of vegetable proteins from an environmental perspective.

Keywords: vegetable protein, meals, ingredients, vegan, vegetarian

Introduction

So far, many LCA studies investigated the importance of meat in diets. These studies show that meat and animal products are responsible for a major share of environmental impacts due to food consumption (Nemecek, Jungbluth et al. 2016). Thus, alternatives are necessary which can provide similar nutritional value, but can be produced with lower environmental impacts.

LCA studies or public data for such alternatives were so far not available in Switzerland. This study investigates the environmental impacts of food products which are rich in vegetable proteins from plants or fungi (sunflower seeds, almonds, mushrooms, dried soybeans, dried chickpeas, dried lentils, canned chickpeas, soy milk, pre-fried falafel, tofu, textured soy protein and mycoprotein).

Goal and Scope

The main goal of the study was to provide reliable life cycle inventory data for products rich in vegetable proteins and to assess the impacts for single ingredients and meals prepared with such products in Switzerland (Jungbluth, Nowack et al. 2016).

The products are assessed as a single ingredient or as part of a home-cooked meal. The recipes for meals are prepared in a way that they provide a good balance of different nutrients. However, meals are not fully equivalent which has to be considered in the comparison and interpretation of results. The main focus was on the protein content. Other nutrients, e.g. the mineral or vitamin content of meals were not considered for this definition. The following meals are investigated in this study:

- falafel with potatoes and yoghurt sauce with herbs
- Bircher muesli (including almonds, soy milk and yoghurt)
- chickpeas with raisins and rice
- brown lentils and polenta
- rice pan with tofu and vegetables
- spaghetti Bolognese with soya mince
- quorn mince and champignon sauce, with noodles

Single ingredients represent quite different nutritional values and are thus not directly comparable. The following single ingredients are investigated as a typical portion:

- soymilk (200 ml)
- sunflower seeds (25 g)

- white mushrooms (120 g)
- soybeans, soaked and cooked (120 g)
- chickpeas, canned, warmed up (125 g)

The impact assessment methods used are the Swiss ecological scarcity method 2013 (Frischknecht, Büsser Knöpfel et al. 2013). The different products are analysed per mass, portion, calorific value and protein content.

Life cycle inventory analysis

Several new life cycle inventory data have been collected and documented. The investigation covers all life cycle stages from agricultural production, processing, distribution, transportation to home, cooling and preparation of the product. Food waste in different stages of the life cycle (but not including food waste after preparation of the meal) is assessed with standard factors for product categories (Flury, Jungbluth et al. 2013). The life cycle inventory analysis uses literature data and direct information by producers. The life cycle inventory data newly investigated for this project are publicly available on www.lc-inventories.ch.

Impact assessment

Figure 3 shows the environmental impacts per portion. Other ingredients than the main protein sources are shown in a separate section or as part of the consumption. For some products the agricultural production is the dominant stage in the life cycle e.g. sun flower seeds. Impacts for some other products, e.g. canned chickpeas, are dominated by the processing or packaging. Environmental impacts of the meals are often considerably influenced by other ingredients than the protein source. The protein source is mainly relevant in case of the meal with quorn and for the Bircher muesli.

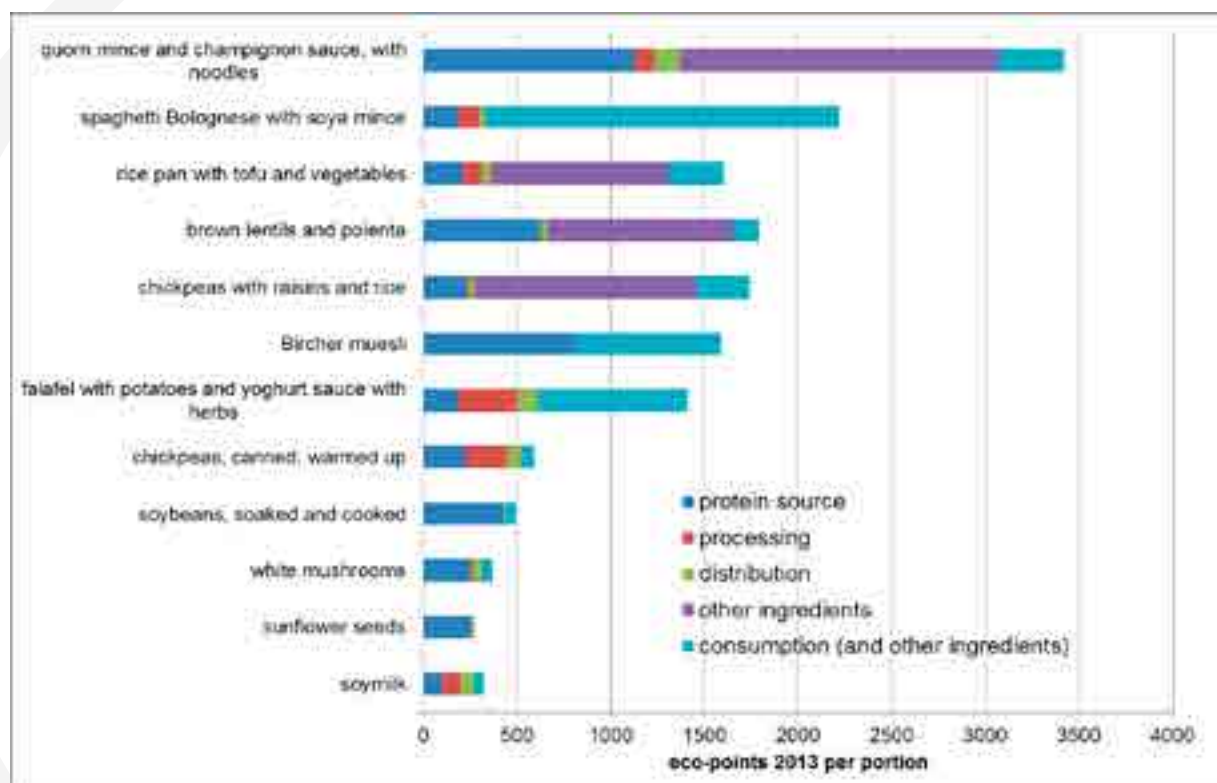


Figure 3
portion)

Environmental impacts of different single ingredients and meals (eco-points 2013 per

Evaluating the environmental impacts for one portion does not reflect differences in nutritional values. Therefore impacts have also been evaluated in relation to calories and protein content of the portions. Figure 4 shows the impacts per gram of protein. Differences between different meals get less pronounced, but still the meal with quorn, which includes a large share of protein from eggs, shows the highest impacts. Mushrooms show rather high impacts due to their low protein content.

If environmental impacts are assessed in relation to the protein content, it is also important to take the biological value of the proteins for nutrition into account. For single ingredients the biological value is lower than for different well combined protein sources. In order to achieve a high biological value and thus a high availability of proteins it is necessary to smartly combine different food products (e.g. rice and lentils, maize and beans, potatoes and milk). This helps to cover the amount of essential amino acids required. The recipes used in this study are based on such considerations.

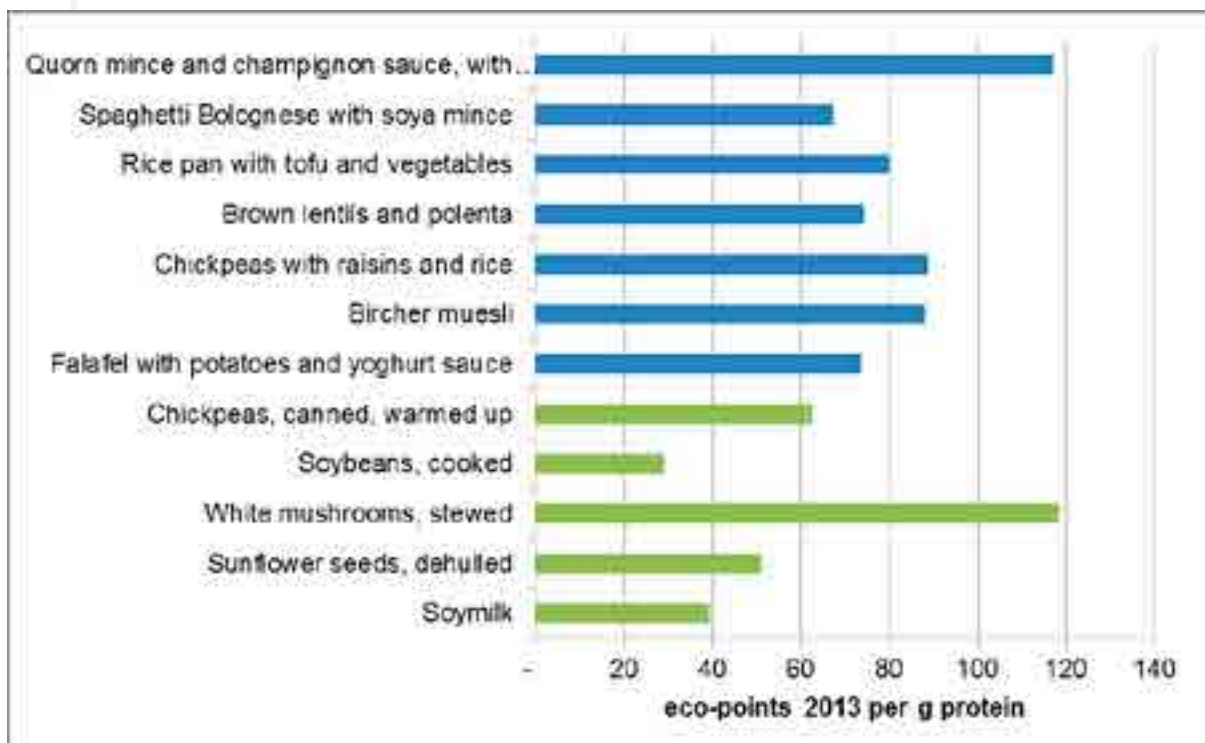


Figure 4 Environmental impacts of different single ingredients (green) and meals (blue) (eco-points 2013 per g protein)

Another evaluation in relation to the energy content of the meals and single ingredients is made in Figure 5. Again mushrooms have a low nutritional value and thus higher impacts than other single ingredients or meals. The ranking between different meals changes with this functional unit and e.g. quorn does not have so much higher environmental impacts than other meals anymore.

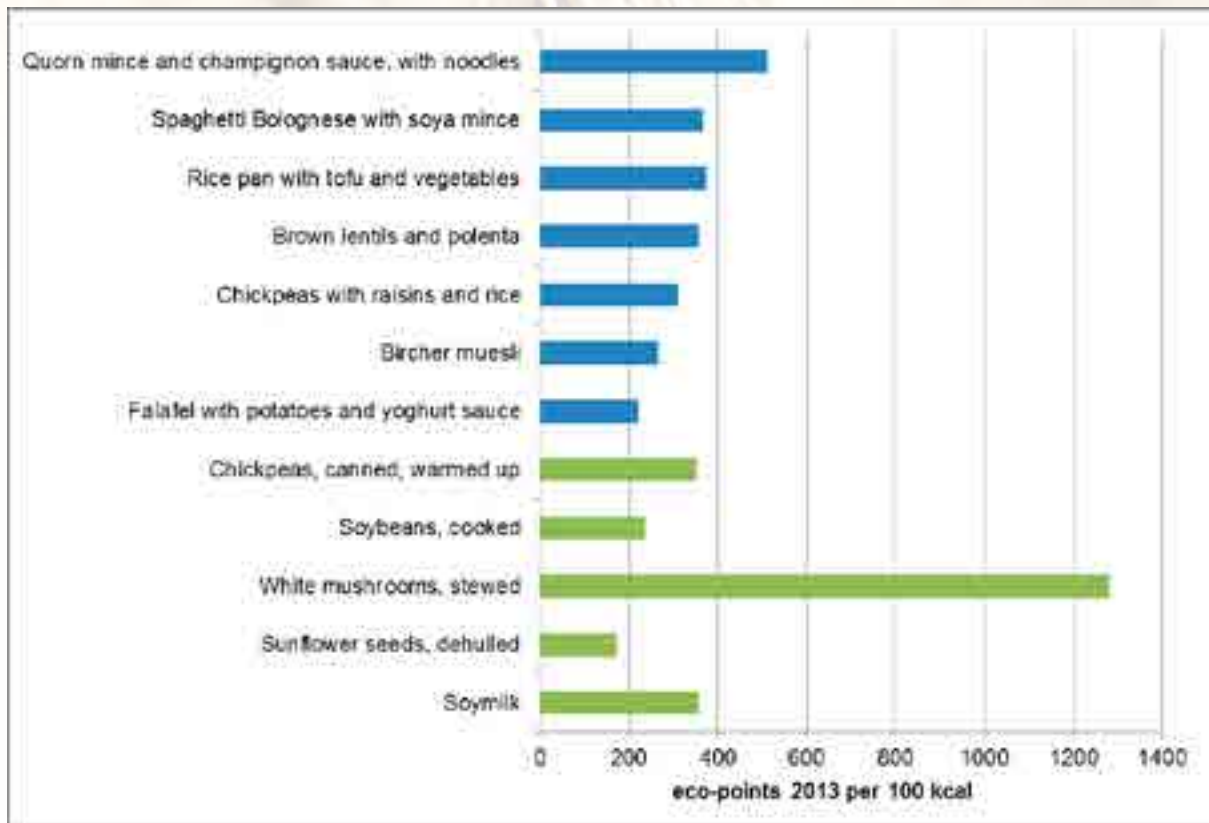


Figure 5 Environmental impacts of different single ingredients (green) and meals (blue) (eco-points 2013 per 100 kilocalories)

The results of adequate meals with vegetarian proteins can also be compared with typical meals including meat or fish as a protein source (Stucki, Jungbluth et al. 2012; Jungbluth, Flury et al. 2013-2016). This evaluation shows that the impacts of meals based on vegetarian proteins are lower than meals including fish or meat, but also provide sufficient nutritional value (Figure 6). For most of the products, the main impacts over the life cycle arise from effects due to land occupation, climate change, air and water pollutants.

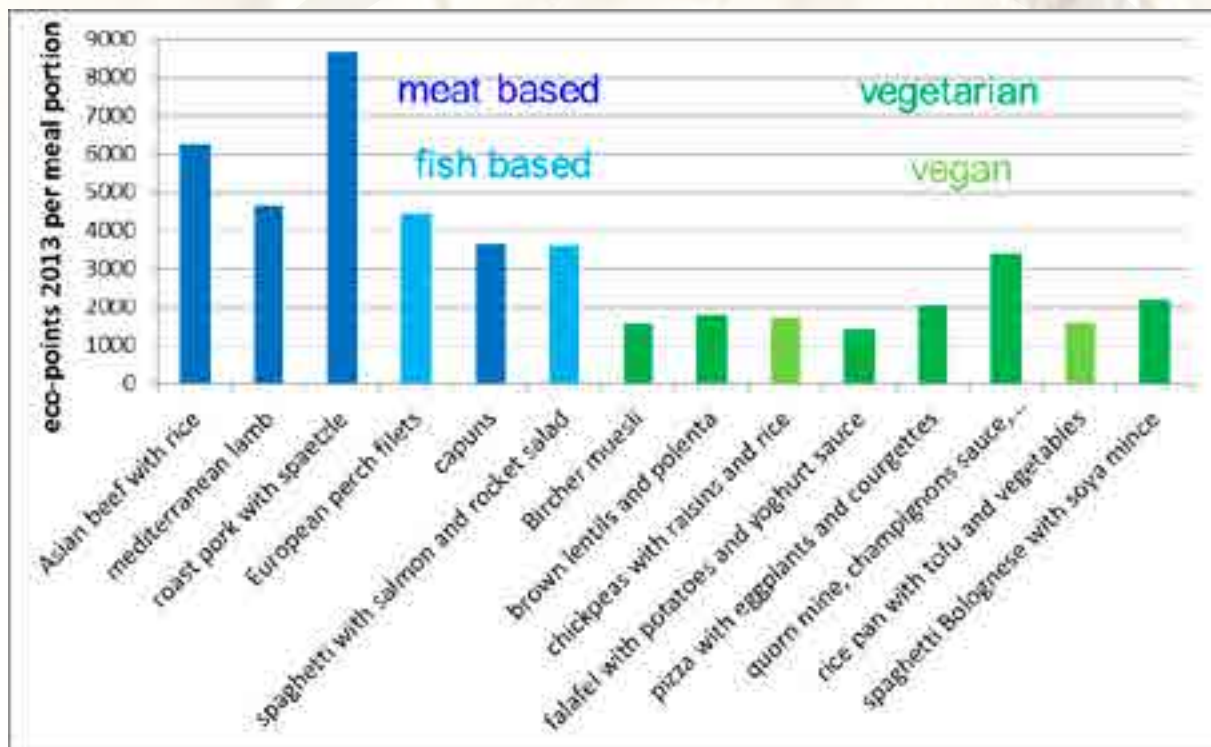


Figure 6 Environmental impacts of different meals (eco-points 2013 per portion)

Implications

This pilot study shows that the impact of vegetable protein products can only be comprehensively assessed when considering the later usage in a meal that is ready for consumption. An evaluation at a preliminary production stage or at the level of a single ingredient is not sufficient for a comprehensive assessment because the combination of protein rich products with other ingredients has a decisive effect on the nutritional value and the environmental impact. Another conclusion of the study is that the environmental impact of food should be specified not only per kilogram or per portion but also per nutrient content.

This study investigates only some examples of products rich in vegetable proteins. The results of this study for some individual examples still do not allow general statements for the evaluation of vegetable proteins from an environmental perspective.

The study lays the foundation for more detailed assessments by developing the necessary methodology and providing transparent data. Furthermore, the data are used for studying the environmental impacts of food consumption patterns (Jungbluth, Eggenberger et al. 2016).

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ORAL PAPERS

METHODS AND DATA 2

20. Methods and data (1)

266. Design and management of sustainable Swiss agricultural systems by involving stakeholders

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ABSTRACT

We conducted a stakeholder survey in the context of the Agroscope research initiative “Production2020”. This initiative aims at developing future-oriented agricultural knowledge and know-how for the design and management of agricultural systems in Switzerland. The objective of the survey was to inform stakeholders about Production2020, to find out about their specific needs and expectations and to involve them into a concerted action supporting Production2020. We selected and interviewed stakeholders representing the food value chain and performed a content analysis. We found that in principle, the stakeholders accepted the holistic approach of live cycle thinking. Domestic food production cannot be reduced on the cost of environmental impacts that are incurred abroad. Stakeholders wish to produce under equal sustainability standards as their foreign competitors. The stakeholders confirmed that the food value chain is only effective and viable together with the agricultural sector. We conclude that Stakeholder in principle accept the LCA approach and support the design of a sustainability impact assessment for the Swiss food system. We deduced 4 main topics from the stakeholder analysis for strategical development of research topics. These were: site-specific farming systems, effective and efficient production, domestic production versus outsourcing – positioning of primary production, and the perspective for the profession of farmer in the future.

Keywords: stakeholder analysis, site-specific farming systems, food value chain,

1. Introduction

The introduction of the proof of ecological performance (PEP) and the ethology programmes in Swiss agriculture in the 1990’s were well received and created consumer confidence. In terms of ecology and animal welfare, Switzerland achieved an internationally leading position. Although the PEP has achieved substantial improvement in environmental conservation, in recent years limitation in attaining ecological targets were observed. A comprehensive analysis of farming systems in Switzerland showed that several benchmarks were not achieved. For instance, the nitrogen load is still too high and the quality of ecological compensation areas should be improved (Herzog *et al.*, 2008). Furthermore, in intensive production regions pesticide concentrations are above water quality standards (Moschet *et al.*, 2014). In general, the sustainability development process of Swiss farming systems has stagnated in recent years. Moreover, groups of comparable farms, e.g. farms belonging to the same farming type, have a high variability in terms of various aspects such as environmental impact, energy use, and earnings from farming, indicating a considerable potential for improvement.

In this context and considering a multitude of challenges such as free trade, environmental impact and climate change facing the agricultural and food sector, Agroscope has launched a research initiative named “Production2020”. It aims at the development of future-oriented agricultural knowledge and know-how for the design and management of agricultural systems in Switzerland. In addition, Production2020 has two main operative goals: (1) providing options for action in plant production, animal production and management, fostering and establishing sustainable, site-specific and internationally competitive farming systems, and (2) developing a methodology for the assessment and design for sustainability of agricultural farms. Production2020 follows seven main principles to reach the goals: (1) considering all three pillars of sustainability, (2) the focus is on the primary production, (3) actions are taken on the farm-manager level, (4) it is embedded in the food value chain with optimal use of resources, (5) it is based on life cycle principles, (6) focusing on impact goals, and (7) strengthening the personal responsibility of the action-taking person. The research initiative Production2020 is understood as an ongoing process to find solutions for improvement to reach the goals. Options of action are developed that could be implemented by farmers on farm level and their impact on sustainability is measured by an appropriate set of indicators improving the design of sustainability of the farm (Roesch *et al.* 2016).

Production2020 is accompanied by a Scientific Advisory Group that advises on scientific and strategic issues to guarantee high scientific quality of the research initiative. Furthermore, stakeholders from the food sector were invited to participate in an Advisory Board to support and strengthen Production2020 and to increase its practical relevance. We conducted a stakeholder consultation in order to determine their acceptance of life cycle thinking in such a context and to give them the opportunity to contribute to the goals of Production2020. The objective of the survey was to inform Stakeholders about Production2020, to find out about their specific needs and expectations and to involve them into a concerted action supporting Production2020. Here we present the outcome of the Stakeholder survey and its conclusions for Production2020.

2. Methods

From the great number of stakeholders from the food value chain we selected 12 representative organisations: 1 public authority, the Swiss farmers' association, 1 extension service, 1 environmental NGO, 2 label organisations, 2 retailers and 4 producers' association. We presented our analysis of the current situation of Swiss agriculture for examination to the representatives of these organisations. This analysis included suggestions for required actions towards more sustainable production systems based on the life cycle thinking, such as focusing on impact goals. This information, together with a description of Production2020, was sent to Stakeholders before the interview took place. Afterwards, in the first quarter of the year 2015, we conducted 12 semi-structured interviews with these stakeholders, asking open-ended questions and discussion may diverge from the interview guide. Each interview lasted 2 hours.

At the beginning of the interview, stakeholders were asked to critically comment on our analysis about the current situation of Swiss agriculture as well as on the suggestions towards sustainable agricultural systems. Then, the interview comprised the following topics:

- Opportunities and challenges of the agriculture and food sector
- Identifying the most important limited environmental resources
- Specific challenges of each stakeholder
- Challenges in the national and international context
- Strategies of the stakeholders to answer these challenges
- Priorities of the stakeholders respecting the agriculture and food sector
- Outlook on their business sector in the next five years ideally and realistically
- Wishes, needs and expectations for Production2020
- Open research questions that Production2020 should answer

The interviews were recorded on tape and transcribed. The software MAXQDA was used to perform a content analysis (Mayring, 2003).

3. Results

The stakeholders widely agreed that the agriculture and food sector has a good reputation and enjoys the confidence of large parts of the Swiss society. Switzerland is a rich country with high purchasing power and there is a large willingness among people to pay a good price for sustainable produced food. According to the stakeholders, society widely accepts and supports the multifunctionality of agriculture, which includes sustainable and market-oriented food production, maintaining the natural foundations of life, maintaining the countryside, and decentralised settlement.

In contrast, the stakeholders were concerned about the increasing liberalisation of agricultural markets and of the competitive cost environment, which challenges the agricultural and food sector. Consequently, the stakeholders agree that additional efforts are needed to increase competitiveness of the sector.

Several stakeholders agreed that the production system of PEP needs to be developed further towards a more sustainable food production, including optimal use of natural resources. Creativity and innovation have to be promoted. At present, farmers are heavily depending on direct payments, which affects their free entrepreneurship. In addition, the agricultural income is under pressure due to decreasing producer prices. Improvements are possible and necessary, and there was consensus among the stakeholders that a sustainable production along the entire food value chain with a partnership-oriented business relations would be a feasible way to strengthen the food value chain. Grassland-based milk and meat production was propagated by some of the stakeholders; they perceived this production system to be well adapted to the natural conditions. In Switzerland, geographical zones (mountain, hill and lowlands) have to be considered in any decision-making process for food production. Another important topic for the stakeholders was the supply of protein for farmers and the domestic production of protein sources, which generally is a subject under intensive discussion in Switzerland. Several stakeholders said that the food value chain is only meaningful together with the primary production.

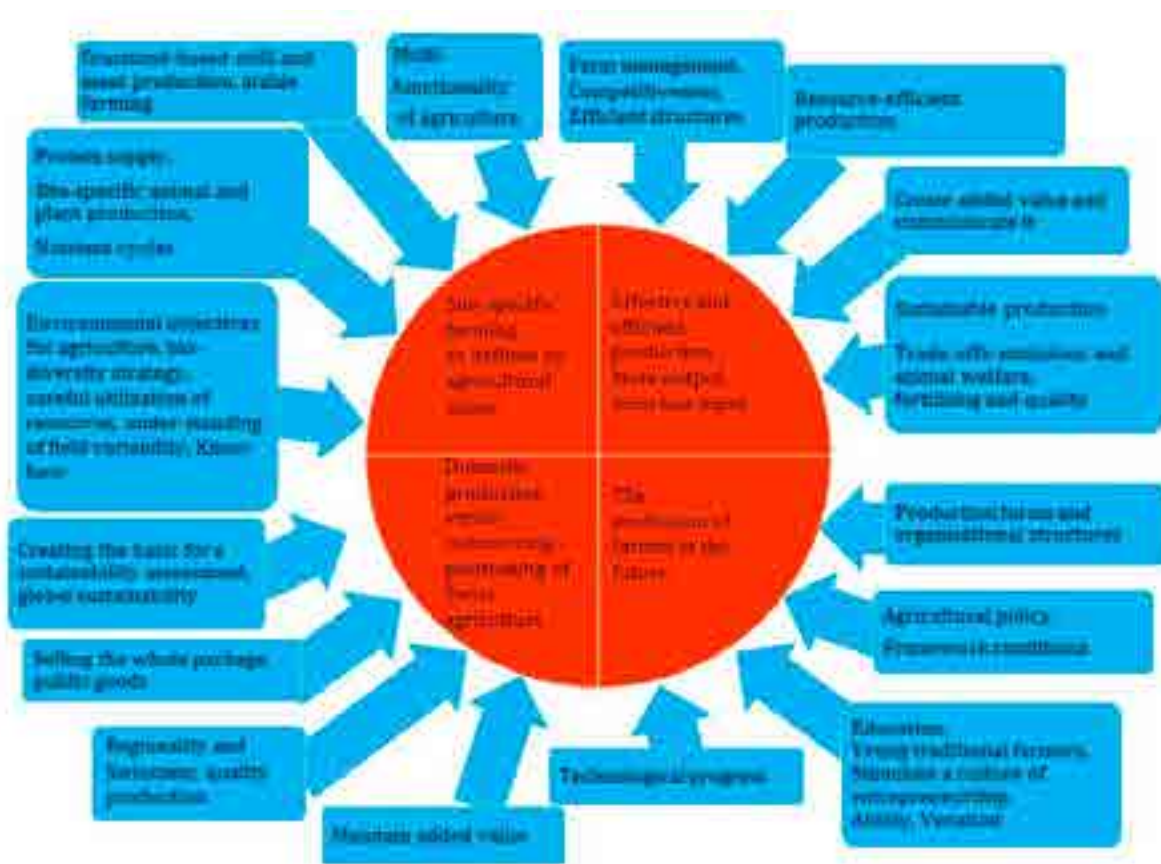


Figure 1: Illustration of the content analysis of the stakeholder survey (in blue) including the 4 main topics (in red) that were deduced for the research initiative Production2020.

Regarding social aspects, the stakeholders were of the opinion that farmers are exposed to high demands concerning know-how and workload. Low income hinders building up sufficient retirement provision. In the event of divorce farmers risk financial constraints. Often there is no alternative to the occupation of being a farmer, and tradition does not allow to give up the farm. The stakeholders saw a problem in the fact that the average age of farmers increases while, at the same time, tradition or diverse reasons make it difficult for newcomers to start an agricultural business, e.g. by purchasing a farm.

The stakeholders agreed that training and advising are key to develop a more sustainable agricultural system and to foster an atmosphere of innovation.

Regarding consumers, the stakeholders observed that these differ considerably in their purchase and consumption behaviour. There was perceived to be a growing interest for food production in general and food waste in particular. Consumers give increasing attention to fair trade, to disclosure of flows of commodity, and to retraceability. This relates mainly to fresh products and is much less pronounced regarding processed food or gastronomy. However, consumer habits are changing, as manifested by an increasing demand for convenience food and for locally-produced food as well as the increasing role of online shopping.

In summary, we found that the stakeholders accepted the holistic approach of life cycle thinking to improve sustainability of the Swiss food system. Three main concerns of the stakeholders were:

- A majority of stakeholders dislike a reduction of domestic food production at the cost of food imports, which would result in exporting environmental impacts and weaken the productive Swiss agricultural sector.
- Most Stakeholders do not want an optimisation which takes into account only one dimension of sustainability (e.g. ecology) but wish to produce under equal sustainability standards. In terms of research need, this would support the need of sustainability impact assessment methods.
- The Swiss food system is only meaningful if the entire food value chain can be maintained. There was consensus among stakeholders that this requires a sustainably optimized food value chain and not only punctual improvements in the agricultural sector.

From this stakeholder analysis, we deduced 4 main topics for strategical development of research topics. We subsequently presented these to the stakeholders at an information event, where they confirmed the topics. These were: site-specific farming systems, effective and efficient production, domestic production versus outsourcing – positioning of primary production, and the perspective for the profession of farmer in the future (Figure 1). These topics cover a wide range of research questions from the stakeholder, which are associated with increasing competitiveness, reducing environmental impacts and improving the social dimension such as development possibilities of small family-owned farms and animal welfare.

4. Discussion

The stakeholders we selected represented the entire agricultural and food sector with the exception of the Swiss Federal Office for Agriculture (FOAG), which could not participate for structural reasons. Consequently, they reflected a broad range of different partial interests, which resulted in a plurality of views and priorities. Moreover, despite this heterogeneity, we observed more statements of consensus such as the importance to maintain the entire food value chain or with few exceptions the disagreement about additional market-openings for agricultural products and foodstuffs from abroad. The basic strategic role of food production and its multifunctionality in Switzerland was widely accepted. Stakeholders also agreed to develop the agriculture and food sector towards a more sustainable food supply of the population. Dissent was observed in questions of details reflecting the different direct interests of the respective stakeholders. Most debated were the trade-offs between economic, ecologic and social objectives. Here the hope of a widely accepted method of sustainability assessment was expressed, which should allow a more objective debate about these issues. In general, the stakeholders were pleased to support Production2020 and consented to further support within their capabilities.

5. Conclusions

The analysis of the Swiss agricultural production system that was made by Agroscope, i.e. several benchmarks were not achieved, sustainability development process of Swiss farming systems has stagnated and variability in various aspects, indicating potential for improvement, was commonly agreed on by the stakeholders, although with different orders of priority. The process to develop the Agroscope research initiative with its goals was very well received among the stakeholders. There was a broad consensus among stakeholders that keeping up the current status quo is not an option, and that improvements and innovations are crucial to support the agricultural and food sector. We conclude that the stakeholders accepted the LCA approach as well as the design of a sustainability impact assessment for the Swiss food system. We deduced 4 main topics from the stakeholder analysis for strategical development of research topics. These were: site-specific farming systems, effective and efficient production, domestic production versus outsourcing – positioning of primary production, and the perspective for the profession of farmer in the future.

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275. The balance of core and noncore foods: a critical intervention point to concurrently address both healthy eating and dietary GHG emissions reduction objectives

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ABSTRACT

In Australia and elsewhere, the escalating rates of overweight, obesity and diet-related chronic disease are major public health nutrition concerns. These issues are closely related to the interrelated problems of excess energy intake, excess consumption of energy-dense nutrient-poor noncore (or discretionary) foods and overall nutrient deficiency which characterize typical diets. In this study, greenhouse gas (GHG) emissions were calculated for more than 9,000 adult daily diets reported in the 2011/2012 Australian Health Survey. When higher quality and lower GHG emission (HQLE) diets were compared to lower quality and higher GHG emission (LQHE) diets, the difference in GHG emissions was 44% for males and 46% for females. The major differentiating factors were the total energy intake and the content of discretionary foods. However, compared to the national dietary guidelines, even males and females in the HQLE subgroup consumed, on average, less than the recommended minimum number of servings of vegetables, meats (and alternatives) and dairy products (and alternatives). Females in the HQLE subgroup also consumed, on average, less than the recommended number of serves of grains. Considering both males and females together, the average adult daily diet had GHG emissions 12% above a nutritionally complete dietary scenario based on the Australian Dietary Guidelines.

Keywords: Australian National Nutrition and Physical Activity Survey, discretionary foods, energy intake, greenhouse gas, sustainable healthy diet

1. Introduction

While the research literature pertaining to sustainable diets has grown rapidly in recent years (Auestad and Fulgoni 2015, Drewnowski 2014, Garnett 2014, Hallström et al. 2015, Johnston et al. 2014), the discourse has tended to focus predominantly on the potential to reduce dietary greenhouse gas (GHG) emissions through reduction in the consumption of livestock products (Aleksandrowicz et al. 2015, Berners-Lee et al. 2012, Hallström et al. 2014, Scarborough et al. 2014, Springmann et al. 2016, etc.). On the one hand, this is understandable since livestock products usually make up a significant proportion of total dietary GHG emissions. However, this focus overlooks the interrelated problems of excess energy intake, excess consumption of energy-dense nutrient-poor noncore foods, and the overall nutrient deficiency that characterize most real diets and are a major and longstanding focus of public health nutrition professionals.

Over the past two decades, numerous public campaigns have been run in Australia to encourage healthy eating and increased levels of physical activity, and to warn about the dangers of overweight and obesity. Federal and State governments as well as a variety of community based organizations have been involved (Australian Government Department of Health 2016, Australian National Preventive Health Agency 2016, National Heart Foundation of Australia 2016, Nutrition Australia 2016) using a wide range of public education strategies, including prominent print and national television advertising. The CSIRO Total Wellbeing Diet (Noakes and Clifton 2005) and related programs have also been prominent in Australian society. These initiatives have all emphasized eating habits which are consistent with national dietary guidelines as described in the Australian Guide to Healthy Eating (NHMRC 2013; Tables 1 and 2). In general, Australians need to reduce consumption of energy-dense and nutrient-poor noncore foods (Table 2) and eat more whole fruit, vegetables, legumes and dairy products (or alternatives). It is also suggested that some children and young women

may benefit from the additional nutrients associated with an increase in red meat consumption (NHMRC 2013).

The purpose of this study was to use data from the latest release of Australia's national nutrition survey (ABS 2014a) to explore the role of noncore (or discretionary) foods (for examples see Table 2) in the Australian diet and their significance with respect to dietary GHG emissions. According to Australian dietary guidelines (NHMRC 2013), discretionary foods should only be eaten occasionally and in small amounts. If reduced discretionary food intake can make a meaningful contribution to lower dietary GHG emissions, then there is a reinforcing environmental message to support existing public health education and the problems associated with multiple, and potentially inconsistent, public education about diets can be avoided (Ridoutt et al. 2016).

Table 1: Summary of Australian Dietary Guidelines for adults 19-50 years (NHMRC, 2013). The recommended minimum number of serves is described, except for discretionary foods where it is the recommended maximum.

Food group	Males	Females
Fruit	2	2
Vegetables and legumes/beans	6	5
Grain (cereal) foods	6	6
Lean meats and alternatives	3	2.5
Milk, yogurt, cheese and/or alternatives	2.5	2.5
Discretionary foods	3	2.5

Table 2: Examples of serves used in the Australian Dietary Guidelines (NHMRC, 2013)

Food group	Standard serve
Fruit	150g fresh fruit
Vegetables and legumes/beans	75g vegetables 1 cup of raw leafy greens
Grain (cereal) foods	1 slice of bread 1/2 cup of cooked rice, pasta, noodle, polenta
Lean meats and alternatives	2/3 cup wheat cereal flakes 65g cooked lean red meat 80g cooked lean poultry 100g cooked fish fillet 2 large eggs (120g) 1 cup of cooked or canned legumes/beans 170g tofu 30g nuts and seeds
Milk, yogurt, cheese and/or alternatives	250ml milk 40g hard or semi-hard cheese 120g ricotta cheese 200g yogurt 250ml cereal drink
Discretionary foods	75g ice-cream 2 slices (50-60g) processed meat 30g salty crackers or crisps 35g sweet plain biscuits 1 tblsp jam or honey 25g chocolate 20g butter 375ml soft drink (sugar sweetened) 60g fried hot chips 200ml wine 400ml standard beer

2. Methods

Dietary intake data, collected using a structured 24-hour recall process and over a 13-month period from May 2011 through to June the following year, were obtained from the Australian National Nutrition and Physical Activity Survey (ABS 2014a). The data covered 9,341 individual adult daily diets and more than 4,500 different foods that were reported to have been eaten. After disaggregation of multicomponent foods and mixed dishes into basic ingredients, the dietary intake data were aligned with the 192 food-related sectors of a highly disaggregated environmentally extended input-output model of the Australian economy (Lenzen et al. 2014) using median prices in a supermarket pricing database. Adjustments were made for under-reporting, following Australian Bureau of Statistics estimates (ABS 2014b), with consistent application across all food items. Whilst it is possible that under-reporting was biased toward certain types of foods (e.g. discretionary foods), insufficient evidence existed to support a method which involved specific allocation of under-reported food energy. In short, food intake was adjusted to achieve an average adult ratio of energy intake to basal metabolic rate (EI/BMR) of 1.55, being the average energy requirement for a normally active but sedentary population. For each individual, energy intake (kJ), GHG emissions (kg CO₂e) and diet quality score (Golley et al. 2014) were then calculated. The diet quality score assessed overall compliance with the Australian Dietary Guidelines (NHMRC 2013). For the purpose of analysis, individuals were grouped into four quadrants, ranking them according to total dietary GHG emissions and diet quality score relative to the mean (Fig 1). Comparisons were made between the average adult diet, those diets with lower diet quality score and higher GHG emissions (LQHE), those diets with higher diet quality score and lower GHG emissions (HQLE), and a diet consistent with the Australian Dietary Guidelines (NHMRC 2013; see Hendrie et al. 2014 for details). Males and females were considered separately.

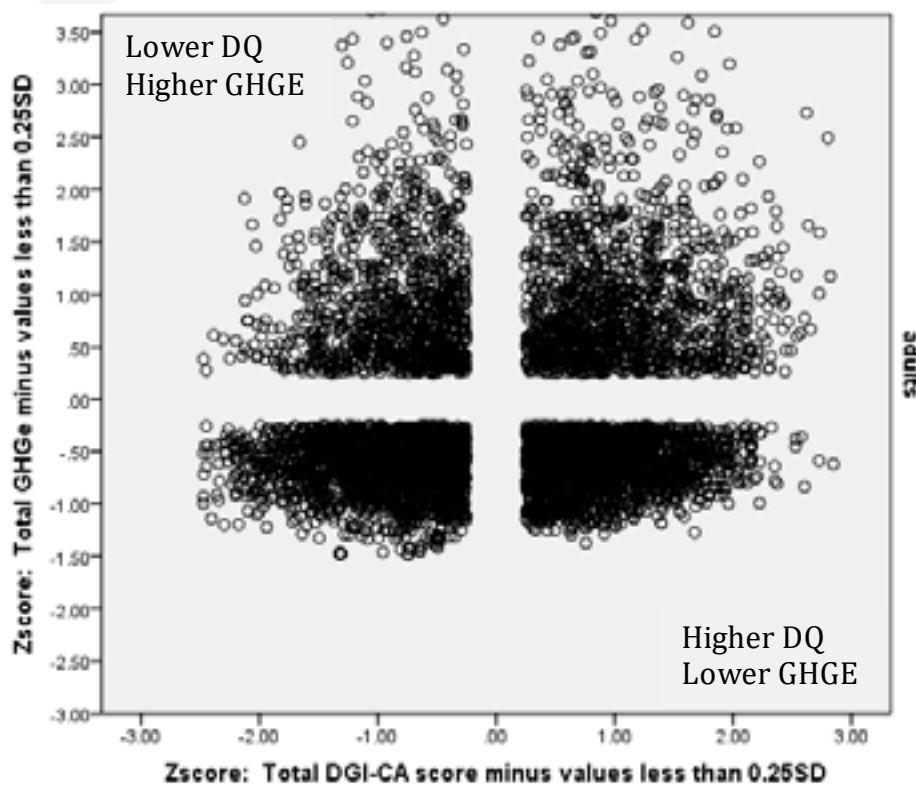


Figure 1: Matrix of diet quality and GHG emission for individual Australian adult daily diets

3. Results

Individuals in the HQLE subgroup had much lower dietary GHG emissions than individuals in the LQHE subgroup (44% lower for males, Table 3 and 46% lower for females, Table 4). The overwhelming difference between the HQLE and LQHE diets was the intake of discretionary foods

and resultant total energy intake. For the HQLE diet, males consumed 2.6 serves per day of discretionary foods, which is marginally less than the recommended maximum of 3.0 serves per day. In contrast, males who were part of the LQHE subgroup consumed an average of 14.8 serves per day of discretionary foods. Females who were part of the HQLE subgroup also consumed marginally less than the recommended maximum number of serves of discretionary foods (2.4 serves per day compared to 2.5 serves per day). However, like males in the LQHE subgroup, females in this subgroup also vastly exceeded the recommended intake with an average of 11.3 serves of discretionary foods per day.

Compared to the LQHE subgroup, the HQLE subgroup had a higher average daily intake of fruits, vegetables and grains and a lower average daily intake of meats (and alternatives) and dairy products (and alternatives) (Tables 3 and 4). However, the differences were small in comparison to the very large differences in discretionary foods described above. Compared to the recommended diet, both males and

Table 3: Food intake (in serves) and daily GHG emissions (kg CO₂e) for four Australian dietary patterns (see text for details) for males aged 19-50 years

Food group	Higher quality/Lower emissions (9550 kJ)		Lower quality/Higher emissions (16,824 kJ)		Average daily diet (13,614 kJ)		Recommended diet for males aged 19-50 years (11,047 kJ)	
	Intake	Emissions	Intake	Emissions	Intake	Emissions	Intake	Emissions
Fruit	3.0	1.0	2.4	0.8	2.6	0.9	2.0	0.7
Vegetables and legumes/beans	4.4	1.6	2.7	1.0	3.5	1.3	6.0	2.2
Grain (cereal) foods	6.0	2.9	4.8	2.3	5.5	2.6	6.0	2.8
Lean meats and alternatives	2.2	7.8	3.7	13.2	3.5	12.7	3.0	10.8
Milk, yogurt, cheese and/or alternatives	1.6	1.8	1.8	2.2	1.8	2.2	2.5	2.9
Discretionary foods	2.6	2.1	14.8	11.3	8.5	6.5	3.0	2.3
<i>Total GHG emissions</i>		<i>17.2</i>		<i>30.7</i>		<i>26.2</i>		<i>21.8</i>

Table 4: Food intake (in serves) and daily GHG emissions (kg CO₂e) for four Australian dietary patterns (see text for details) for females aged 19-50 years

Food group	Higher quality/Lower emissions (7975 kJ)		Lower quality/Higher emissions (13,974 kJ)		Average daily diet (10,226 kJ)		Recommended diet for females aged 19-50 yrs (10,148 kJ)	
	Intake	Emissions	Intake	Emissions	Intake	Emissions	Intake	Emissions
Fruit	2.3	0.8	1.9	0.7	2.0	0.7	2.0	0.7
Vegetables and legumes/beans	3.8	1.4	2.5	1.0	3.3	1.2	5.0	1.8
Grain (cereal) foods	4.4	2.2	3.5	1.7	4.2	2.0	6.0	2.8
Lean meats and alternatives	2.0	7.3	3.8	13.7	2.6	9.4	2.5	9.0
Milk, yogurt, cheese and/or alternatives	1.4	1.7	1.9	2.3	1.6	1.8	2.5	2.9
Discretionary foods	2.4	1.8	11.3	8.7	5.9	4.4	2.5	1.9
<i>Total GHG emissions</i>		<i>15.1</i>		<i>27.8</i>		<i>19.7</i>		<i>19.2</i>

females in the HQLE subgroup consumed, on average, less than the recommended minimum number of servings of vegetables, meats (and alternatives) and dairy products (and alternatives). Females in the HQLE subgroup also consumed, on average, less than the recommended number of serves of grains (4.4. serves per day compared to 6.0). Males in the HQLE subgroup met the recommended minimum intake only for fruits and grains. Females in this subgroup met the recommended minimum intake only for fruits. As such, even though the HQLE diets were superior to the LQHE diets, they were, on average, far from ideal.

Interestingly, there was no obvious difference between the GHG emissions intensity of the meats (and alternatives) food group between the HQLE and LQHE subgroups (3.6 kg CO₂e per serve each), suggesting that, on average, there were no major differences in the choice of meats and alternatives eaten. The strongest association was between total energy intake and GHG emissions ($r=0.54$, $P<0.001$).

Comparing the average adult and recommended adult diets, the GHG emissions were 23% higher for the average male diet and 1% higher for the average female diet (Tables 3 and 4). On average, males consumed almost three times the recommended maximum number of serves per day of discretionary foods (8.5 serves per day compared to 3.0 serves per day). Though less pronounced, they also consumed above the recommended number of serves of fruits and meat (and alternatives) and less than the recommended number of serves of vegetables, grains and dairy products (and alternatives). Similarly, on average, females consumed discretionary foods in excessive quantities (5.9 serves per day compared to a recommended maximum of 2.5 serves per day). On average, they also consumed too few serves of vegetables, grains and dairy products (and alternatives). Considering adult males and females together, the average daily diet had GHG emissions 12% above the dietary scenario based on the Australian Dietary Guidelines.

4. Discussion

This study differs from most other studies of dietary GHG emissions by focusing on discretionary foods rather than livestock products. Obviously, if individuals seek to aggressively reduce dietary GHG emissions then vegetarian or vegan diets may be appropriate provided care is also taken to avoid the risks of personal micronutrient deficiency. However, suggestions to consume less livestock products are not generally well aligned with existing national dietary guidance in Australia and elsewhere, they pose risks in the case of population subgroups like children, pregnant women and the elderly (Temme et al. 2015a), and may not be culturally acceptable in many cases (Macdiarmid et al. 2016, Temme et al. 2015b). As such, strategies to reduce dietary GHG emissions based on livestock products may have limited effectiveness. In contrast, strategies based on limiting intake of noncore or discretionary foods have no known health risks, are fully coherent with existing national dietary guidance and also have outstanding potential to achieve dietary GHG emissions reduction. If the subgroup of Australian adult males with the lowest diet quality scores and highest dietary GHG emissions (LQHE) reduced their intake of discretionary foods from the current 14.8 serves per day to the recommended maximum of 3.0 serves per day, the GHG emissions savings would be almost 30% (Table 3). Similarly, if the subgroup of Australian adult females with the lowest diet quality scores and highest dietary GHG emissions reduced their intake of discretionary foods from the current 11.3 serves per day to the recommended maximum of 2.5 serves per day, the GHG emissions savings would be 25% (Table 4). If the discretionary food intake for these groups was further reduced to the levels reported by the HQLE subgroup, the GHG emissions benefits would be even greater again.

However, as described by Hendrie et al. (2014), the limitation of discretionary foods needs to be balanced by adequate intake of core foods in order to achieve a nutritionally complete diet. Not all diets which are lower in GHG emissions are necessarily healthy. Some lower GHG emission diets are reported to be higher in sugar and lower in micronutrient content (Payne et al. 2016). In the Australian context, even the subgroup with highest diet quality score (HQLE) was below the recommended level of intake for several food groups, including vegetables (Tables 3 and 4). It is therefore an important finding that a nutrient rich adult diet that meets the requirements of the Australian Guide to Health Eating (NHMRC 2013) is also lower in GHG emissions than the average adult daily diet. The 12% lower GHG emissions reported here, based on the 2011/2012 Australian dietary data, is less than the previously reported 25% based on the 1995 dietary data (Hendrie et al. 2014), but is nonetheless

sizeable. Other studies have also reported GHG emissions benefits of recommended diets relative to average diets of up to 17% (Green et al. 2015, van Dooren et al. 2014, Meier and Christen 2013). Taken together, these studies highlight the GHG emissions reduction potential of adopting healthy diets without reducing flexibility by narrowing healthy food options. Naturally, strict optimized diets can achieve much greater GHG emissions cuts (Perignon et al. 2016, van Dooren et al. 2015, Wilson et al. 2013), but the likelihood of any significant proportion of the population adopting such optimized diets must be questioned and the cumulative impact of a minor proportion of the population adopting these diets will be small in comparison to shifts toward sustainable healthy diets which are more mainstream.

5. Conclusions

In Australia and elsewhere, the escalating rates of overweight, obesity and diet-related chronic disease are major public health nutrition concerns. These issues are closely related to the interrelated problems of excess energy intake, excess consumption of energy-dense nutrient-poor noncore foods and overall nutrient deficiency which characterize typical diets. When higher quality and lower GHG emission diets in Australia were compared to lower quality and higher GHG emission diets, we found the major differentiating factors were the content of discretionary foods and total energy intake. Recommendations to limit discretionary food consumption are consistent with existing national dietary guidance, have no known health risks and can contribute large dietary GHG emission benefits.

6. Acknowledgement

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37. Modelling farm and field emissions in LCA of farming systems: the case of dairy farming

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ABSTRACT

The generic nature of LCA with a wide coverage of environmental processes and impacts, calls for a simplified environmental modelling. This is notably the case for agricultural and food systems, where often generic emission factors (EFs) are used such as those derived from the IPCC guidelines. These EFs reflect country averages or a global situation, but are not suited for comparing different farming systems, since they do not take into account specific management, climate, and soil parameters and therefore do not distinguish between different farming systems. Moreover, emission models in LCA tend to have varying degrees of detail, which means that climate, soil and farming practices are reflected for some emissions and processes, while ignored for others. In pasture-based livestock systems, direct emissions from animals and pastures tend to dominate many of the environmental impacts such as climate change, eutrophication or acidification. The objective of this study is to improve emission modelling for nitrogen (N) of pasture-based dairy systems in comparison to intensive dairy production using high amounts of concentrate feed.

The Swiss Agricultural Life Cycle Assessment (SALCA) method uses a nutrient balance model of a herd to calculate emissions from animal husbandry and nutrient excretion. The effects of feed intake, feed quality, and different levels of productivity on emissions and environmental impacts can thus be represented. To better consider the specific aspects of pasture-based dairy systems, the modelling of N excretion in urine and dung and subsequent N emissions is revised. The excretion of N in urine and dung is modelled as a function of N concentration in the diet. Subsequently different emission factors for N₂O, NH₃ and NO₃ are applied from urine and dung and for N₂O emissions from organic fertilisers. Furthermore, a correction factor for N₂O emissions after the application of N fertilisers is calculated as a quadratic function of the N rate. The revised emission models can result in changes of over 50% compared to use of generic EFs and should better reflect the N emissions from pasture-based dairy systems.

Keywords: dairy production, nitrogen emissions, nitrogen excretion, pasture

1. Introduction

The nitrogen and the carbon cycles are key drivers for the environmental impacts of agricultural systems (Williams *et al.*, 2010), since they are major determinants of several impact categories like e.g. the global warming, the eutrophication and acidification.

Grassland-based dairy systems are important examples of agricultural systems, where the nitrogen and carbon cycles play a key role (Ledgard, 2001). Methane from enteric fermentation and manure management, nitrous oxide from manure management, and deposition to soils, ammonia emissions related to animal husbandry and manure management as well as nitrate emissions from grazing and spreading manure are key drivers for the environmental impacts of dairy farming (Guerci *et al.*, 2013). The emissions of the greenhouse gases (GHG) methane and nitrous oxide are typically determined by simple EFs in LCA. EFs were developed for the use in national GHG inventories (IPCC, 2006), designed for the accounting at the national scale. Using the same EFs for specific farming systems might be inappropriate, since the specific conditions of the analysed systems are not taken into account (Peter *et al.*, 2016). Therefore, the EFs and models often used in LCA might not

adequately represent the emissions in real systems, since they do not reflect specific aspects of the investigated systems. Improving nitrogen emission models was identified as a top priority for the improvement of LCA of livestock systems (Cederberg *et al.*, 2013). The default EFs for nitrous oxide (N₂O) from agricultural soils from IPCC (2006) are unspecific and for example do not distinguish between different types of fertilisers or between excretion of urine and dung from grazing animals.

A further challenge is the varying level of detail for several emission pathways. This can be clearly seen for the different nitrogen losses in the form of ammonia, nitrous oxide and nitrate. While for ammonia (EEA, 2013; HAFL, 2013) and nitrate (Richner *et al.*, 2014) relatively detailed EFs and models are available taking into account the type of fertiliser, the timing of application, the soil and climate factors, nitrous oxide emissions from fertilisation are mainly calculated from the amount of N applied (IPCC, 2006). Therefore, variations in climate and soil conditions as well as mitigation measures cannot be evaluated properly for all emissions.

Pasture-based dairy systems differ from systems with high use of concentrate feed in many respects: animal husbandry (barn feeding vs. grazing), milk yield per cow, type of feed, manure management (manure spreading vs. excretion on pasture), and type of land use (grassland vs. arable land). Ideally, the EFs and models should reflect these aspects, in order to achieve a differentiated analysis. This paper presents the approach followed in the Swiss Agricultural Life Cycle Assessment (SALCA) method to model emissions from animal husbandry (section 2.1) and the recent improvements made in order to achieve a more adequate analysis of pasture-based dairy systems, with focus on the N modelling and emissions (sections 2.2 to 3.5).

2. Modelling emissions from animal husbandry

2.1 Overview of the concept

Calculations for emissions from animal husbandry and nutrients (N, P, and K) excreted by the animals are performed by a nutrient balance model of a herd in the SALCA method (Fig. 1). It takes into account the specific feed intake and quality, the export of animal products, changes in live weight, and emissions. The effects of feed intake, feed quality, and different levels of production on emissions and environmental impacts can thus be represented.

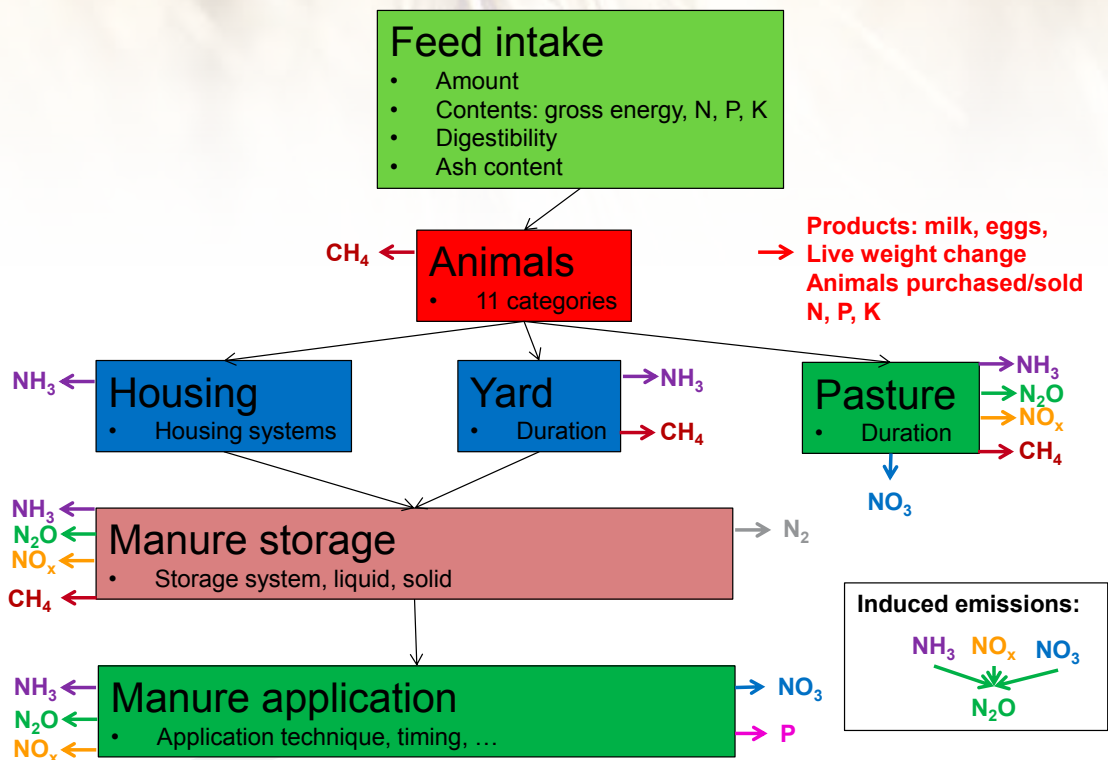


Figure 1: Flow diagram to model nutrient balances and emissions from animal husbandry (SALCA-Animal). Nutrient flows and emissions related to application of other fertilisers than animal manure are calculated separately and not shown in the figure.

The following EFs and models are used in SALCA to model emissions from animal husbandry: Agrammon (HAFL, 2013) or EEA (2013) Tier 2 for ammonia depending on the context of the study, IPCC (2006) Tier 2 factors for methane complemented by Kirchgessner *et al.* (1995) for dairy cows, IPCC (2006) Tier 2 factors for N_2O from manure management, EEA (2013) for NO_x and N_2 emissions, and Richner *et al.* (2014) for NO_3 leaching.

Details are given in Bystricky *et al.* (2014) and Nemecek *et al.* (2015). In the following sections the adaptations of the models are discussed to better account for the specific aspects on pasture-based systems. The focus in this paper is on the refinement of N emission modelling from grazing cow systems.

2.2 Nitrogen partitioning between urine and dung

Most emission models used in LCA do not take into account the partitioning of excreted N between urine and dung, e.g. EEA (2013) calculate N excretion in a balance model and assume a fixed proportion of TAN in total N of 60%. However, the proportions of N excreted in urine and dung vary considerably (Bracher and Menzi, 2015; Luo and Kelliher, 2014) and are influenced by many factors. Digestible N enters the cow's metabolism and is either used to produce milk or to form body tissue; the remaining N is excreted in urine. The non-digestible fraction of N is excreted in dung. The proportion of N excreted in the urine increases strongly with increasing N intake and increasing N concentration in the diet, while the proportion of N excreted in the dung changes only little in function of the above-mentioned parameters (Luo and Kelliher, 2014). Taking these relationships into account is crucial for emission calculations. N in urine is mostly composed of urea (73% according to Selbie *et al.* (2015)), the rest consists of various soluble N compounds. In fresh cattle dung, 99% of the N is organically bound, which is partly water soluble (fresh cattle dung contains

21.5% soluble N compounds (Kirchmann and Witter, 1992)). Due to these different compositions of urine and dung, the emissions of N_2O , NH_3 and NO_3 are much higher from urine than from dung.

Luo and Kelliher (2014) proposed relationships between N concentration in the diet and the N excretion in urine. A comparison of their numerical relationship to several experiments carried out in Switzerland however showed relatively large deviations particularly for low protein diets. Therefore the relationship was re-estimated for use in Swiss dairy systems using data from 14 studies published between 1993 and 2014 (total of 420 data points, personal communication of A. Bracher, Agroscope, May 2016 and Bracher and Menzi (2015), see Fig. 2).

A linear regression was calculated for lactating cows as

$$\%N_{urine} = 4.7 + 20.7 * \%N_{diet} \quad (n=389, r^2=0.65) \quad (eq. 1)$$

where

$\%N_{urine}$ = N excreted in urine [g N/day]/total N excreted in urine and dung [g N/day]*100

$\%N_{diet}$ = N concentration in the diet [g N/kg dry matter]*100.

A linear relationship is plausible only in a certain range. From the analysis of moving averages it was concluded that average values below 30% and above 70% are unlikely and the linear function was therefore truncated at these limits (Fig. 2). It should be noted that for individual cows and in specific circumstances higher and lower values can occur. Dry cows were not included in the regression. However, dry cows have generally low $\%N_{diet}$ and the corresponding values were around 30%; therefore the truncated function can also be applied to dry cows.

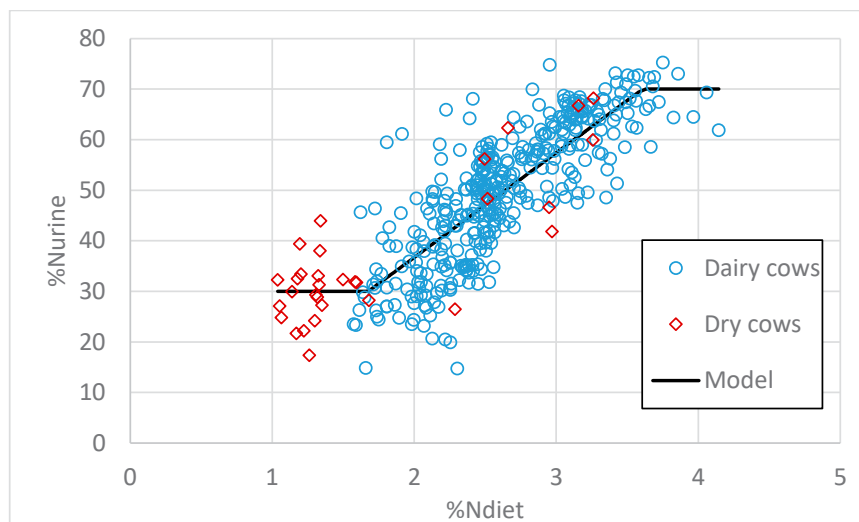


Figure 2: Proportion of N excreted in urine ($\%N_{urine}$) as a function of the N concentration in the diet ($\%N_{diet}$) on a dry matter basis.

This relationship is used for the subsequent modelling of N_2O (sections 3.1 and 3.3), NH_3 (section 3.4) and NO_3 (section 3.5) emissions. It is also used to estimate the proportion of TAN in the total N in the excrements as a starting point for the modelling of N dynamics (mineralisation, immobilisation) and N emissions during manure storage and subsequent manure spreading.

3. Modelling N emissions from manure management and fertiliser application

3.1 Direct nitrous oxide emissions from grazing cattle

N_2O emission rates from cattle excrements deposited on pasture are generally higher than N_2O emissions from applied fertilisers (Kelliher *et al.*, 2014). IPCC (2006) therefore proposed a default EF of 2.0% for grazing cattle excreta N compared to 1.0% for N fertilisers. The reason is that cattle excreta have a high N concentration, which corresponds on average to a local application rate of 613 kg N/ha within urine patches (Selbie *et al.*, 2015). The local N concentration is too high, so that N cannot be taken up by the vegetation quickly enough, which results in higher losses compared to the low N rate from an even distribution of N fertilisers.

Numerous experiments have shown that N_2O emissions from urine are considerably higher than from dung. N is organically bound in dung and needs to be mineralized first, before denitrification and nitrification can start, and N_2O can be formed. This leaves more time to the vegetation to use the N, resulting in lower emission rates. Kelliher *et al.* (2014) analysed data from 185 experiments carried out in New Zealand between 2000 and 2013. In the lowland soils the mean EFs of 1.16% were found for cattle urine and only 0.23% for cattle dung. For the hill areas (with slopes >12%), significantly lower EFs were found (Luo *et al.*, 2013). Higher EFs for urine N as compared to dung N were also found by Bell *et al.* (2015) for Scotland, Mori and Hojito (2015) in Japan, Sordi *et al.* (2014) in Brazil, and Rochette *et al.* (2014) in Canada.

In the New Zealand's GHG inventory, national Tier 3 EFs of 1.0% and 0.25% are used for cattle urine and cattle dung excreted on pasture, respectively (MfE, 2015). These values are considerably lower than the IPCC default value for grazing cattle, which is 2.0% for all excreted N. Lower EFs can be observed for urea, where the IPCC default EF is 1.0%, while the mean EF determined in New Zealand was only 0.48% (Kelliher *et al.*, 2014). The lower N_2O emissions in New Zealand are mainly explained by the different soil types, which are often of volcanic origin and tend to be coarse-textured and well-drained, where optimal conditions for the production of N_2O are less frequent. De Klein *et al.* (2014) found lower EFs on free draining soils as compared to poorly drained soils.

Therefore the average value of 2.0% for emissions from urine is used in SALCA from a meta-analysis (Selbie *et al.*, 2015), while for dung a value of 0.5%, i.e. 4x lower is applied, thus keeping the same relationship as in New Zealand's national GHG inventories. If these EFs are combined with the proportions excreted in urine and dung from eq. 1, depending on the diet the resulting EF for N_2O is 23-53% lower than the IPCC default, previously used in SALCA (Fig. 3).

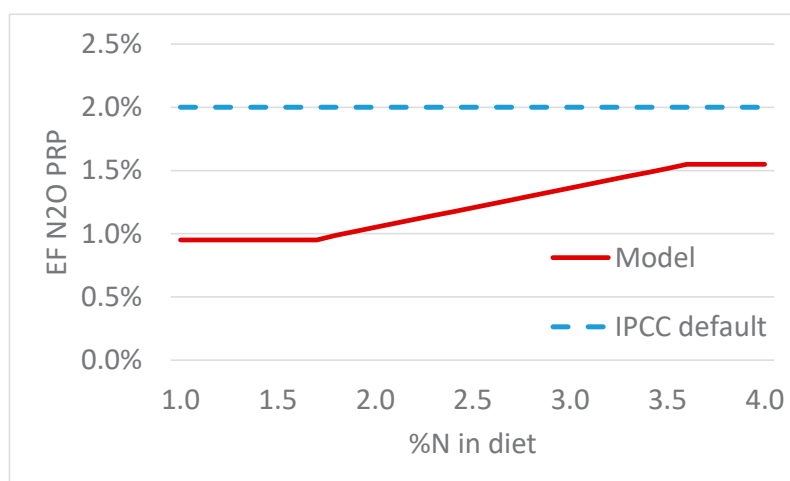


Figure 3: EF for N_2O on pasture (EF N_2O PRP) compared to the IPCC default (Tier 2).

3.2 Direct nitrous oxide emissions after application of N fertilisers at different rates

Higher rates of N application tend to result in higher emissions of N₂O (Bouwman *et al.*, 2002). At higher N rates, the vegetation might not be able to take up the N quickly enough. Shcherbak *et al.* (2014) analysed 78 studies with several levels on N application and found a non-linear response of N₂O emissions. Using the relationship derived for all experiments except four outliers, a correction factor (CF_{N₂O rate}) can be derived as a quadratic function of the N fertiliser rate:

$$CF_{N_2O\text{rate}} = 0.1 * (1.036/N_{\text{fert}} + 6.42 + 0.0244 * N_{\text{fert}}) \quad (\text{eq. 2})$$

where

N_{fert} = N fertiliser applied [kg N/ha/year]

The correction factor equals 1 at a rate of 146 kg N/ha/year, which corresponds to typical N doses. Lower N rates lead to reduced N₂O emissions as compared to the IPCC default value, while higher N rates give greater emissions (Fig. 4).

This correction factor is multiplied with the EFs for N₂O for mineral and organic fertilisers except for excreta from pasture (EF N₂O PRP, see section 3.1). The reason is that in the latter the N application rate is already taken into account in the higher EF for N in urine as compared to N fertilisers. Interestingly, the CF_{N₂O rate} for 613 kg N/ha/year (the average urine N rate, see section 3.1) yields a factor of 2.1, a value which is very close to the EF_{N₂O} for cattle urine of 2.0%.

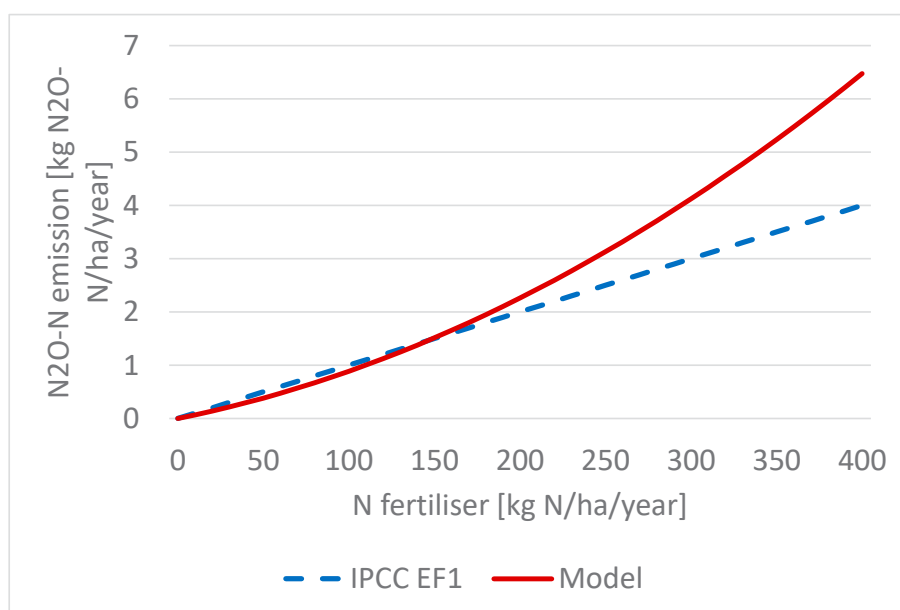


Figure 4: Emissions of N₂O from fertiliser applications with correction for the N application rate compared to the IPCC default (Tier 1).

3.3 Direct nitrous oxide emissions after application of organic fertilisers and manure

Since only a part of the N contained in organic fertilisers and manure is readily available, the emission rates can be expected to be lower than from mineral fertilisers (Bouwman *et al.*, 2002; Stehfest and Bouwman, 2006). However, the emissions depend on many factors such as type of organic fertiliser, soil, climate, conditions of application, etc. (Lesschen *et al.*, 2011) so that generic conclusions are difficult to draw (Thangarajan *et al.*, 2013). We opted therefore for the application of a simple robust model.

The TAN of organic fertilisers is known either from the N balance model (Fig. 1) in the case of farmyard manure, from standard tables or specific analyses. The default EF1 for N fertilisers (1.0%) will be used for the mineral part of N (TAN) as for the mineral fertilisers. For the organically bound fraction (total N - TAN), a factor of 0.25% will be used, i.e. a value 4x lower, thus keeping the same relationship as between urine and dung excreted on pasture. Note that the SALCA-animal model presented in Fig. 1 includes also mineralisation of N in liquid manure and immobilisation of N in solid manure.

3.4 Ammonia emissions from grazing cattle

Ammonia emissions from dung excreted on pasture can be considered as negligible, since almost all N is organically bound (Kirchmann and Witter, 1992). For urinary N, Selbie *et al.* (2015) give an average emission rate of 13% (median 12%; kg NH₃-N/kg N in urine) from a meta-analysis. Most of the measurements were done in summer, when ammonia volatilisation is highest. A seasonal variation can be derived as 8% in spring, 15% in summer, 9% in autumn and 7% in winter (Selbie *et al.*, 2015).

3.5 Nitrogen leaching from grazing cattle

Nitrate leaching shows a seasonal pattern, with highest leaching rates in autumn and winter and lowest in spring (Selbie *et al.*, 2015). From a meta-analysis the average leaching rates from urine patches were estimated as 17% in spring, 16% in summer, 24% in autumn and 20% in winter (Selbie *et al.*, 2015). N leaching from dung was measured only in a few studies and the results indicate much lower leaching rates than for urine, but do not allow a robust estimate of the leaching rate. As only 21% of the N is water soluble (Kirchmann and Witter, 1992), much lower leaching rates can be assumed. Therefore the above leaching rates for urine patches were extrapolated by multiplying by 0.2 in the case of N in dung. Most data on N leaching stem from lysimeter studies. In most of these experiments, the whole area of the lysimeter was wetted with urine. Buckthought *et al.* (2016) has shown that not only the plants growing in the wetted area use the N in the urine patch, but also the surrounding vegetation. 20% of the urinary N applied in spring was taken up by the surrounding vegetation. The authors conclude that many lysimeter studies might overestimate leaching under real conditions. Therefore, a correction factor of 0.8 will be multiplied by the calculated leaching rates for grazing in spring and 0.9 for grazing in summer to determine the effective N leaching, summarised in Table 1. No correction will be applied to autumn and winter application, since the recovery of N by the plants is low in these seasons. Higher stocking rates lead to more frequent overlaps between excreta, which increases the risk of N leaching. E.g. Ledgard *et al.* (2015) has shown increasing leaching rates with increasing N excretion from grazing animals. Such relationships could be taken into account in future model improvements.

Table 1: N leaching factors as a function of the season.

	Spring	Summer	Autumn	Winter
N leaching from urine [kg N leached/kg N excreted]	13.6%	14.4%	24.0%	20.0%
N leaching from dung [kg N leached/kg N excreted]	2.7%	2.9%	4.8%	4.0%

4. Discussion, conclusions and outlook

The proposed changes in the N emission modelling better take into account specific aspects of pasture-based dairy systems. In particular the effect of different diets on the emissions of N₂O, NH₃ and NO₃ is accounted for. Grazing dairy cows generally have diets with high protein contents, which leads to a higher proportion of N excreted in urine and subsequently higher N emissions. Furthermore, the N concentration of the diet depends on the composition of the feedstuffs and is related to the milk yield. In this context it is important to distinguish between diets during the grazing season and winter diets instead of using average annual diets. However, cows with low N concentrations could also have low milk yields, so that the emissions and impacts per kg milk have to be evaluated through the LCA of the whole dairy system in the next steps.

The adapted models will be tested in the project "Optimisation of grassland-based dairy systems with grass cutting and barn feeding" comparing three production strategies for dairy cows in Switzerland: 1) full grazing, 2) grass cutting and barn feeding low amounts of concentrates and 3) barn feeding using moderately high amounts of concentrates.

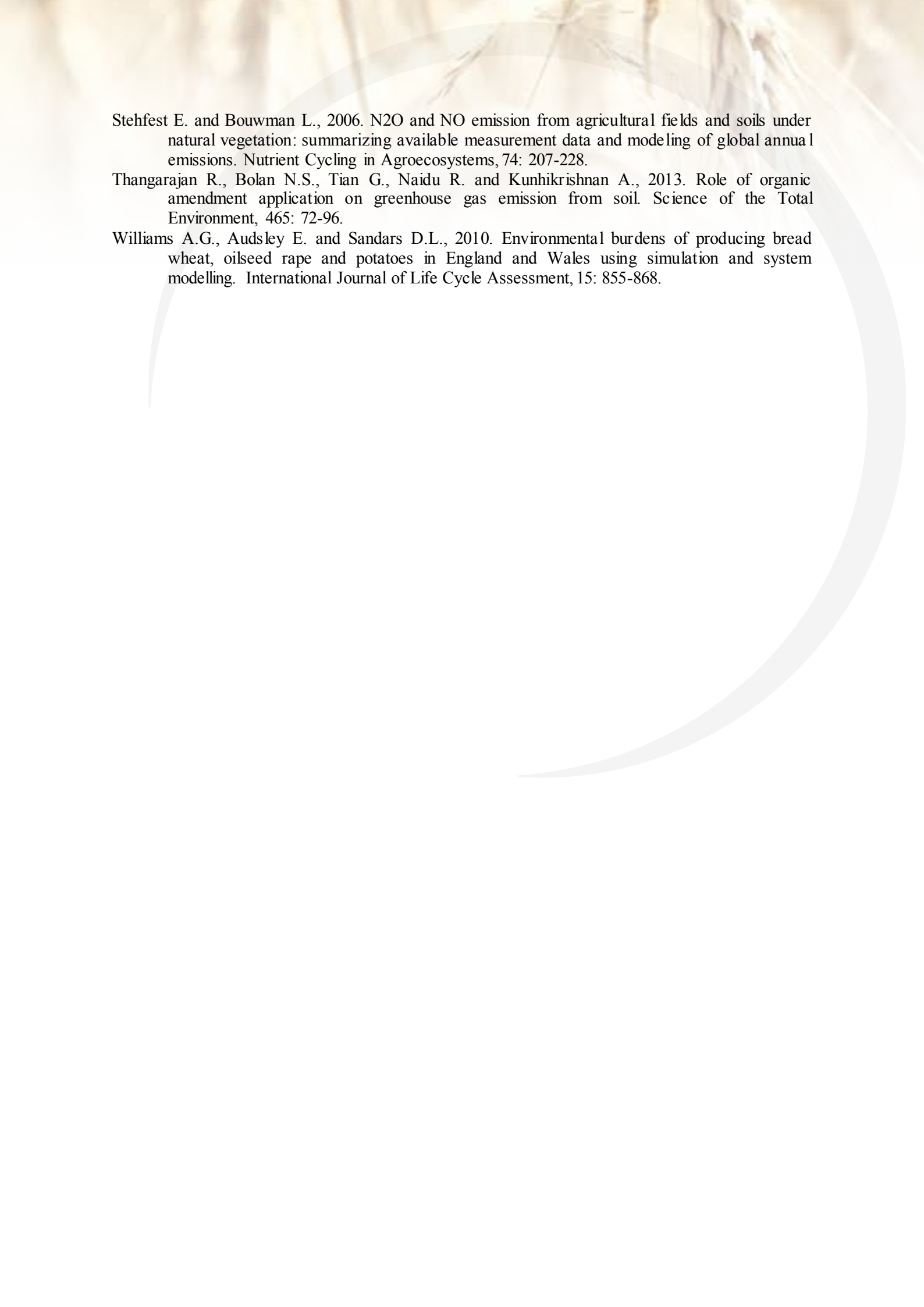
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166. Reducing environmental impact of dairy product through ECODESIGN

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ABSTRACT

The assessment of the environmental potential impact caused by the food industry is becoming crucial for most of the companies in Europe. European regulations are focusing on the environment, and specifically, some of them recommend the use of the Life Cycle Assessment (LCA) for measuring potential impacts of goods and services. However, most of the commercial LCA tools are focused to be used by experts, these are not friendly tools for the food industry, and there are no accurate data of every sector in the food industry. Within this framework, LIFE ECOLAC project has developed a software tool based in an LCA case study of a dairy company, connected to common ERP software, and suitable for the impact accounting. The main results obtained with the tool were compared with those obtained from SimaPro 8.2. and also with the average results from literature. The values obtained from the tool were aligned with the results from SimaPro 8.2 and with the averages from literature. Therefore, the conclusion is that the tool is valid for the environmental impact accounting.

Keywords: LCA, software, ERP, milk, climate change

1. Introduction

In 2006, the European Commission published the EIPRO (Tukker et al, 2006) Report which identified the products with the greatest environmental impact over their life cycle in the EU. The study concluded that the food and beverage sector accounts for between 20 and 30% of the various environmental impacts such as climate change, and increases to over 50% for eutrophication. This includes the entire chain of distribution and food production ('farm-to-fork'). Thus, the assessment of the environmental potential impact caused by the food industry is becoming crucial for most of the companies in Europe. Moreover, European directives and recommendations are focusing on the environment and they have recently developed the Product Environmental Footprint methodology which recommends the use of the Life Cycle Assessment (LCA) for measuring potential impacts of goods and services.

However, most of the commercial LCA tools, such as Simapro or Gabi, are focused to be used by experts and they are not directed to be easily utilized by the food industry. In fact, there are no accurate data of every sector in the food industry, and these are not friendly tools for people not familiarized with LCA.

It is important to emphasize that over 80 % of the environmental impact of a product is determined in the design phase (NRC, 1991). Ecodesign involves taking into account all environmental impacts of a product from the earliest stages of design, in order to avoid unexpected changes in the food production chain. European Commission, based on the Ecodesign Directive (DIRECTIVE 2005/32/EC), has sought to strengthen this aspect specifically in products consuming energy, and the food industry has been left apart.

Moreover, private approaches supporting ecodesign strategies have been mainly focused on the primary sector (due to its greater impact) and on the packaging stage but there are no approaches including the whole production or value chain (Zufia and Arana, 2008). Therefore, it is necessary to address this problem from holistic point of view, involving even consumers' decisions. However, there are few consumer studies focusing on the food industry and few research studies that reflect the relationship between sustainability and the impact on the behavior or attitude of consumers regarding the consumption of sustainable products.

Within this framework, the ECOLAC project funded by the European LIFE + Environment program, intends to promote products' environmental improvement through the development of an innovative software tool which allows companies to obtain eco-designed food products.

Within the food sector, meat products and dairy are the two sub-sectors producing greater environmental impacts. The contribution of milk, cheese and butter to global warming is estimated at 4%. In conclusion, to reduce the environmental impacts associated with the consumption of water and energy, improvements should be focused on the food sector that produces a greater burden on the environment. Therefore, the ECOLAC LIFE project focuses on the dairy sector which also has a wide range of degrees of freedom for the designing of new environmentally friendly products.

2. Methods

The project develops a software tool connected to common Enterprise Resource Planning (ERP) (SAP i.e.) software which allows companies to find new improvement ideas to reduce the impact of their product. The software is based on the results obtained in LCA of dairy products.

2.1. Life Cycle Assessment of dairy products

The goal of this life cycle assessment is to identify the main hotspots of the dairy processing industry in order to include them in the software tool. As agreed by the IDF (IDF, 2010) and in accordance with current Product environmental Footprint Category Rules for dairy products, the functional unit of the study is 1 kg of FPCM UHT milk including also the packaging. Moreover, in order to include various dairy products, the production of 1 kg of butter, 1 kg of UHT cream and 1 kg of packaged yogurt have been incorporated. The studied system includes production of the raw milk, raw milk transportation, dairy processing and final distribution (fig 1) of a dairy industry located in the North of Spain.

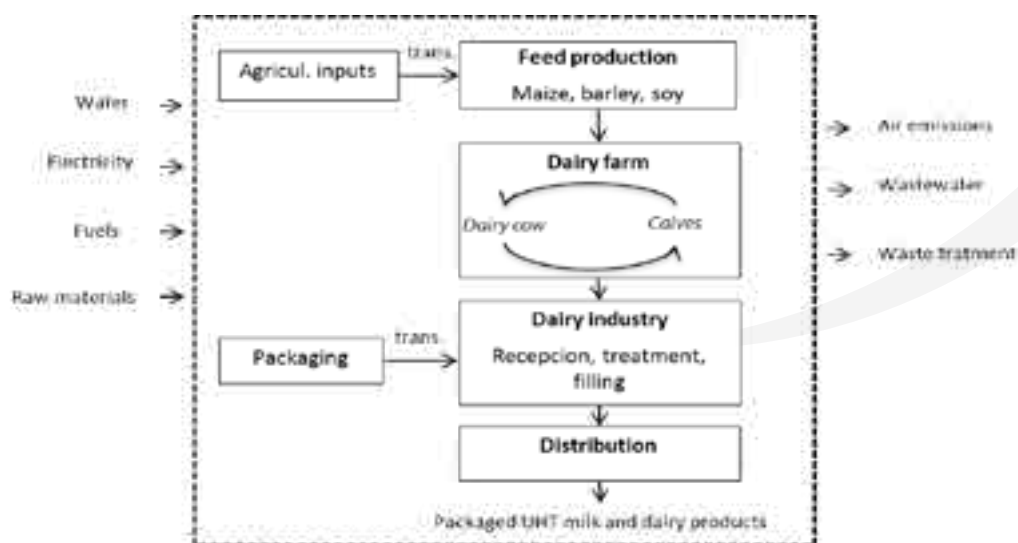


Figure 1: Life Cycle system boundaries of dairy production

The inventory analysis includes the main inputs and outputs of all the production stages. It is important to point out that one of the main objectives of this project is to improve efficiency in dairy processing, thus, the inventory has also been more exhaustive in this stage. However, it is widely known that raw milk production at farms is the main hot spot of almost all the dairy products, so the majority of farms suppliers have been inventoried. Data from 54 different farms have been collected for year 2013 and 2014. Almost all the milk suppliers' farms are part of CLAS (Central Lechera Asturiana) farmer association. This association collects data regarding feed, fuel use, water use, manure management, herbicides and pesticides use and productivity of all the farms using specific questionnaires for each year. Data regarding greenhouse gasses emission from enteric fermentation and manure management have been taken from IPCC guidelines (Dong et al., 2006)

In tables 1 and 2 the main inputs and outputs are detailed. It is important to note the production of livestock for those farms is on average 5000 L/head annually.

Table 1: Simplified inventory of key data of subsystem 1 livestock management

	Amount	Unit
Product		
Raw milk	1	t
Input from the technosphere		
Livestock	0,2	heads
Water	1,25	m ³
Diesel	12,10	kg
Electricity	45,93	Kwh
Feed	463	kg
Barley	70	kg
Maize	194,63	kg
Others	198.37	kg
Pasture	4900	kg
Output to the nature		
CH ₄ (enteric fermentation)	23,4	kg
CH ₄ (manure management)	6,8	kg

Table 2: Simplified inventory of key data of subsystem 2 dairy processing industry

	Amount	Unit
Product		
Dairy products*	1	kg / L
Input from the technosphere		
Raw milk	1,15	L
Water	3,87	L
Natural gas	0,10	Kwh HCV
Electricity	0,11	Kwh
Packaging materials:		
Packaging container	4,33	g
Aluminium	0,92	g
Paper/cardboard	0,07	g
Plastics HDPE	4,79	g
Plastics LDPE	2,44	g
Output to the nature		
R410A	0,015	mg
NH ₃	0,553	mg

* 1 L milk : 0.08 kg yogurt : 0.02 kg butter : 0,03 kg creaml

The following environmental impact categories have been selected for the impact assessment: climate change (Kg CO₂ eq from IPCC (Solomon, 2007)), human toxicity cancer effects (CTUh from Rosenbaum et al., 2008), human toxicity non-cancer effects (CTUh from Rosenbaum et al., 2008), acidification (molc H⁺ eq from Posch et al., 2008), terrestrial eutrophication (molc N eq from Posch et al., 2008), freshwater eutrophication (kg P eq from Struijs et al., 2009), freshwater ecotoxicity (CTUE from Rosenbaum et al., 2008), land use (kg C deficit from Milà i Canals 2007), abiotic resource

depletion (kg Sb eq from Guinée et al., 2002) and water resource depletion (m³ H₂O eq from Frischknecht et al., 2008). Mass allocation has been selected for the allocation.

The complete LCA of the selected dairy products has been carried out using SimaPro 8.0 and ecoinvent 3. database.

2.2. Development of the ECOLAC tool

The ECOLAC tool has been developed using Visual Basic .Net, on Visual Studio 2012. The database engine used is SQL Server 2008 R2, where all the application's information is stored. As far as the application imaging, both design and pictures implemented, were done using Gimp 2.8.

According to the result obtained from the LCA of dairy products, the tool accounts for the main inventory flows affecting the environmental sustainability of the dairy products. Moreover, since the main objective of the project is to facilitate environmental impact accounting, the tool is linked directly to the SAP[®] enterprise resource planning database. For that purpose, the tool is fed on a text format document exported from the SAP[®] with i) electricity, water and steam readings from the meters located along the production chain, ii) the delivery destinations of each product, iii) purchasing of packaging and auxiliary materials and, iv) waste and wastewater generation. Each reading or value is then allocated to the corresponding process or stage defined previously for the whole plant.

3. Results

The main results of the project are, on the one hand the environmental footprint of 4 different dairy products and, on the other hand, the environmental impact accounting ECOLAC tool.

3.1. Life Cycle Impact Assessment of dairy products

As stated above, the tool is based on an LCA case study of a dairy company, where the main inputs affecting environmental impact are defined. Overall, for the selected dairy products, the climate change potential varies between 1,5 kg CO₂ eq per kg of FPCM UHT milk and 9 kg CO₂ eq. per kg of butter (table 3).

Table 3: The main impact characterization results for the studied dairy product.

IC	Unit	UHT milk	Yogurt	Cream	Butter
CC	kg CO ₂ eq	1,56E+00	1,55E+00	1,65E+00	9,07E+00
HT _{ef}	CTUh	3,39E-08	2,43E-08	3,92E-08	1,17E-07
HT _{nef}	CTUh	9,04E-07	8,16E-07	1,75E-06	2,43E-06
A	molc H ⁺ eq	1,09E-02	9,64E-03	1,48E-02	4,64E-02
EuT	molc N eq	4,47E-02	4,05E-02	6,23E-02	1,19E-01
EuA	kg P eq	7,01E-05	5,64E-05	8,16E-05	3,51E-04
EcoT	CTUe	4,35E+00	3,84E+00	4,80E+00	1,72E+01
LU	kg C deficit	1,33E+01	1,16E+01	1,42E+01	3,47E+01
ARD	kg Sb eq	3,23E-03	2,64E-03	3,83E-03	4,79E-02
WRD	m ³ water eq	7,60E-03	6,71E-03	8,27E-03	1,84E-02

As expected, the overall environmental assessment suggests that the GHG emissions from the enteric fermentation and manure management emitted during raw milk production are responsible for the 90 % of the potential climate change impact. Other impacts such as human toxicity, acidification, eutrophication, land use and resource depletion follow the same trends as climate change impact.

It is important to highlight that the potential impact of water depletion presents different result, giving more emphasis to the environmental role of the dairy processing. For this impact category the sterilization of the milk has more importance than other inputs.

3.2. ECOLAC Tool: Environmental assessment linked to management software

The software tool is divided into four main sections as detailed below:

- *Configuration of the company:* The configuration is the most complex section of all the software, and all key aspects of the facility are defined in detail, i) the manufactured products, ii) primary and secondary packaging materials, iii) chemicals and refrigerants used, iv) established production processes, v) implanted meters and vi) waste types. In this section the company or the person in charge should link the meters with the different processes defined. This task is particularly sensitive because it can lead to large errors in the impact accounting. In principle, this section should not be modified unless significant changes are made in packaging, meters or production.
The tool includes the possibility to export to Excel file the whole relationship between meters and processes in order to facilitate the checking of the assignment.
Note that in the future when implementing the tool in another facility is likely that company will require external assistance from the administrator of the tool.
- *Process diagram:* This section of the tool shows a final image of the facility, taking into account all processes. It is also possible to visualize what products go through different processes and which elements have each process (figure 2).

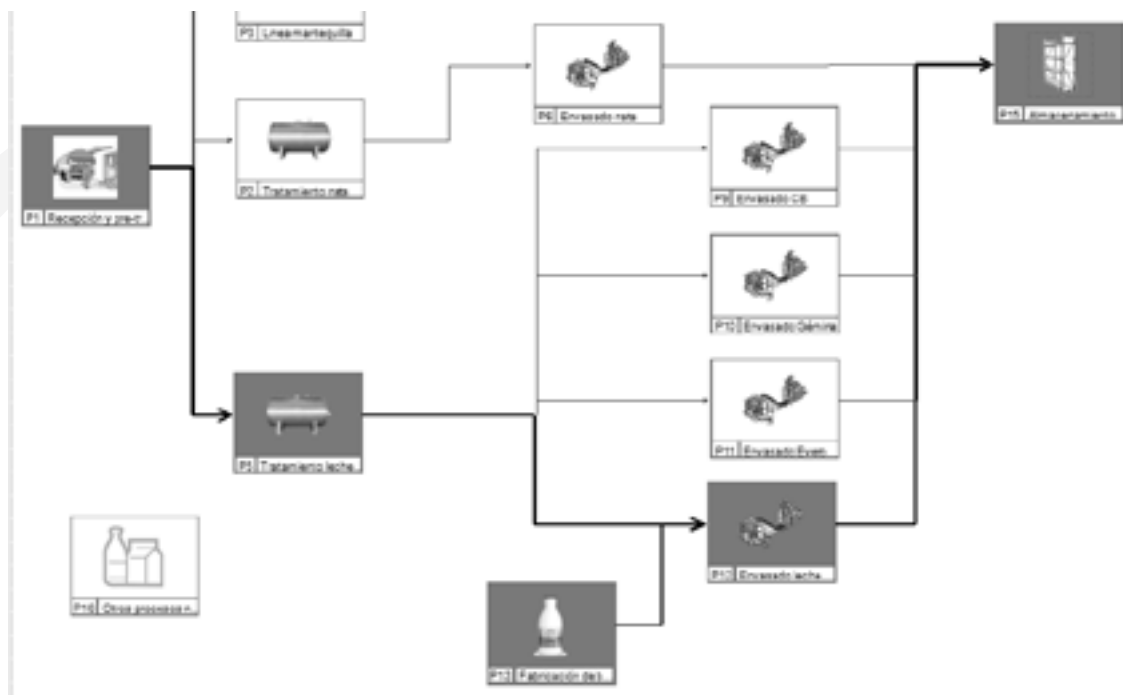


Figure 2: Product diagram flow in the ECOLAC tool

- *Monthly data:* In this section the company could import the readings of the meters and other required values (figure 3). Moreover, it has been recently implemented the option to include not only the values, but also the economic cost of each input in order to analyze the economic sustainability of the products.
As far as the raw milk is the main cause of most environmental impact categories, there is an option to change the main inputs regarding the raw milk production such as feed ingredients, water consumption, diesel consumption and productivity (n° of cow heads / L raw milk)

Codigo	Nombre
1001	Cortado LP (Mozz)
1002	Cortado V2 (Mozz)
1003	Cortado V3 (Mozz)
1004	Cortado V4 (Mozz)
1005	Cortado V5 (Mozz)
1006	Cortado V6 (Mozz)
1007	Cortado V7 (Mozz)
1008	Cortado V8 (Mozz)
1009	Cortado V9 (Mozz)
1010	Cortado V10 (Mozz)
1011	Cortado V11 (Mozz)
1012	Cortado V12 (Mozz)
1013	Cortado V13 (Mozz)
1014	Cortado V14 (Mozz)
1015	Cortado V15 (Mozz)
1016	Cortado V16 (Mozz)
1017	Cortado V17 (Mozz)
1018	Cortado V18 (Mozz)
1019	Cortado V19 (Mozz)
1020	Cortado V20 (Mozz)
1021	Cortado V21 (Mozz)
1022	Cortado V22 (Mozz)
1023	Cortado V23 (Mozz)
1024	Cortado V24 (Mozz)
1025	Cortado V25 (Mozz)
1026	Cortado V26 (Mozz)
1027	Cortado V27 (Mozz)
1028	Cortado V28 (Mozz)
1029	Cortado V29 (Mozz)
1030	Cortado V30 (Mozz)
1031	Cortado V31 (Mozz)
1032	Cortado V32 (Mozz)
1033	Cortado V33 (Mozz)
1034	Cortado V34 (Mozz)
1035	Cortado V35 (Mozz)
1036	Cortado V36 (Mozz)
1037	Cortado V37 (Mozz)
1038	Cortado V38 (Mozz)
1039	Cortado V39 (Mozz)
1040	Cortado V40 (Mozz)
1041	Cortado V41 (Mozz)
1042	Cortado V42 (Mozz)
1043	Cortado V43 (Mozz)
1044	Cortado V44 (Mozz)
1045	Cortado V45 (Mozz)
1046	Cortado V46 (Mozz)
1047	Cortado V47 (Mozz)
1048	Cortado V48 (Mozz)
1049	Cortado V49 (Mozz)
1050	Cortado V50 (Mozz)

Figure 3: Data introduction screenshot

- *Results*: The result section is divided into three main subsections in order to provide the company with reliable and harmonized environmental assessment. First, product environmental analysis is presented and the main causes of the selected environmental impact are suggested. Secondly, environmental impact comparison analysis is accessible for the organization and the company could compare environmental results of different products. Finally, “ecodesig” subsection has been developed. In this subsection the company could formulate different production scenarios in order to calculate the potential impact reduction, the widely known “what-if” analysis.

4. Discussion

4.1. Interpretation of dairy products LCA

Milk production at farm is the main responsible for environmental impact of the life cycle of dairy products. However dairy industry has also around 20 % of impact in some of the selected categories and thus, if efforts are focused on the dairy industry, minimization of milk losses could slightly improve the environmental profile, with reductions of up to 1.3 %, according to Gonzalez-Garcia et al. 2013. According to Berlin et al. (2008), energy efficiency in a dairy factory is closely related to the product being produced. However, in this case study, energy needs in the dairy factory is not the main cause of environmental impact. Therefore, environmental improvements resulting from this alternative measures are almost negligible (less than 1% in all categories), since the effects of the energy requirements in the environmental profile throughout the life cycle of dairy production are low.

Looking at production processes, reducing wastewater would be another of the variables to take into consideration. It could allow a significant reduction in the absolute values of impact associated with obtaining water resources, such as water depletion, as in impacts related to wastewater discharges, such as eutrophication.

Another interesting variable to consider is the distance covered by the distribution systems. Distribution of dairy products is one of the production stages with higher impact. Therefore, when

reducing the environmental impact of the products, measures to reduce the distribution of dairy products, as reducing distances or using transport with less impact on the environment should be considered. The studied dairy industry optimizes the filling the trucks, so the route optimization would be in this case the variable with the greatest potential for improvement.

With regard to the packaging, it should be noted that there are significant differences between the studied packaging, especially in the impact categories related to toxicity. Therefore, in order to minimize the impacts associated with packaging, promoting the container of HDPE bottle is suggested.

Furthermore, allocation of farm impact between dairy products and beef has also been considered in several dairy LCAs (Berlin et al, 2008; Berlin and Sonesson, 2008). Gonzalez-Garcia (2013) identifies the influence on the impact characterization when considering different perspective of allocation. Once all impacts of livestock are loaded to dairy products, they increase up to 8% the final impact. Note that in this study, 100% of the impact of the farm has been assigned to milk production, because the male calves were outside the limits of the study and the needs for their growth were not taken into account.

Although the improvement actions proposed in this discussion do not lead to a large reduction of impact, it is necessary to emphasize that all these improvement actions coming from the different stages of the product life-cycle management, combined, could lead to a significant reduction of the overall impact.

4.2. Using the ECOLAC tool in industrial environment

The functionality of the tool has been validated in a real facility of a dairy industry located in the North of Spain. The company receives daily more than 500 million liters of raw milk, produces more than 260 million liters of UHT milk and around 30 million kilograms of other dairy products such as yogurt cream and butter. Overall, there are 16 steam meters, 15 electricity meters and more than 50 water meters located along the dairy company.

The operator of the environment department of the company was the person in charge for using the tool. The configuration of the whole plant and the allocation of the entire meter to the different products was the most tedious part of the demonstration; however as far as the operator was aware of all the details of the plant, this barrier was solved easily.

The main results obtained with the tool were compared with those obtained from SimaPro 8.2. and also with the average results from literature. Finally, values obtained from the tool were aligned with the results from SimaPro 8.2 and with the averages from literature, and thus it could be concluded that the tool is valid for the impact accounting.

According to the Head of the environmental department of the dairy industry (Spain), the tool will be very useful in the future in order to create new ecodesigned dairy products (it is expected to launch a new product by the end of 2016). Moreover, new unexpected functionalities were also found such as the control of the global energy and water consumption of the facility, which could lead to a great cost saving strategies.

5. Conclusions

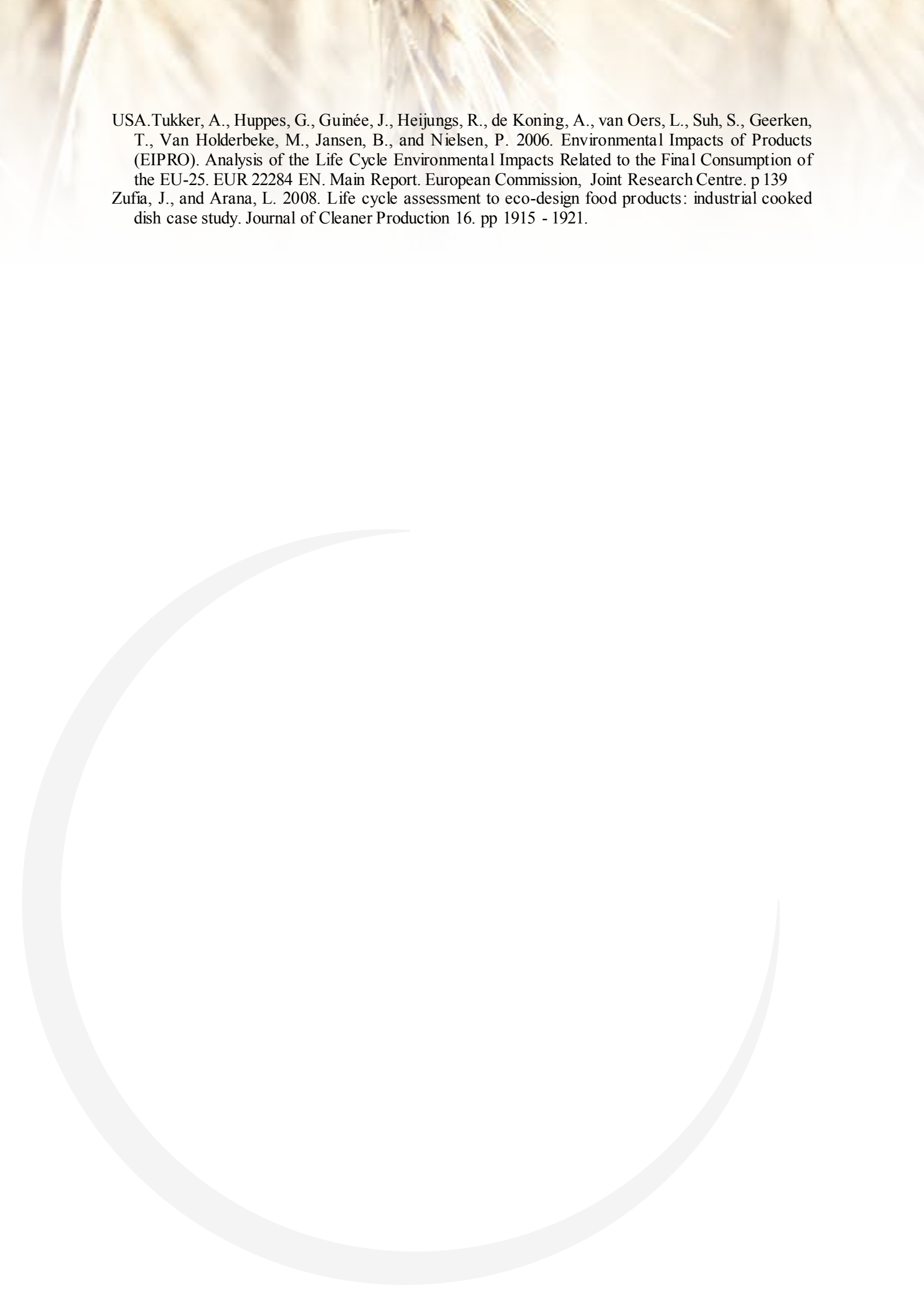
On the whole, the ECOLAC tool has been developed to be a suitable tool for the impact accounting of the dairy industries. Moreover, in order to be in accordance with the upcoming recommendation from the international regulations, this tool is aligned with the Single Market for Green Products initiative. This initiative is developing a new framework for measuring the “product environmental footprint” of all kind of good and services commercialized in the EU.

The integration of economic aspects in this tool is still not fully developed. Further research work is necessary in order to integrate all the incomings and costs assigned to a certain product.

Finally, as a recommendation, it is important to highlight that there is a need to encourage food companies to include the environmental issues in the decision-making processes by making the stakeholders of the food chains aware of the sustainability of their products.

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147. Reducing uncertainty at minimal cost: a method to identify important input parameters and prioritize data collection

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ABSTRACT

The study aims to illustrate a method to identify important input parameters that explain most of the output variance of environmental assessment models. The method is tested for the computation of life-cycle nitrogen (N) use efficiency indicators among mixed dairy production systems in Rwanda. We performed a global sensitivity analysis, and ranked the importance of parameters based on the squared standardized regression coefficients (SRC). First the probability distributions of 126 input parameters were defined, based on primary and secondary data, which were collected from feed processors, dairy farms, dairy processing plants and slaughterhouses, and literature. Second, squared SRCs were calculated to explain the output variance of the life-cycle nitrogen use efficiency, life-cycle net nitrogen balance, and nitrogen hotspot index indicators. Results show that input parameters considered can be classified into three categories. The first category (I) includes 115 input parameters with low squared SRCs (<0.01), which are less important and can be established with default or regional averages. The second category (II) includes 5 important input parameters, with squared SRCs between 0.01 and 0.1; that can be established with country specific data. The third category (III) includes 6 input parameters with squared SRCs >0.1 ; that contribute most to the output variance of at least one of the life-cycle nitrogen use efficiency indicators. These most important parameters need to be established with accuracy thus require high data quality. The input parameters of category II and III include emission factors and coefficients that are specific for a region as well as activity data that are specific to the livestock production system. By carrying out such analysis during the scoping analysis, any LCA study in food sector can cut on the cost of data collection phase by focusing on input parameters that can be fixed through good practices in data collection. Further work on global life-cycle nutrient use performance will benefit from these results to generate analysis at lesser data collection cost.

Keywords: Sensitivity analysis, uncertainty analysis, minimum data, data quality.

1. Introduction

Life-cycle assessment is known to be relatively data hungry, especially when applied to diverse and internationalized supply chains such as livestock. These data mainly include activity measurements, emission factors, coefficients and parameterized factors of the model. No dataset contains all these required data. The lack of a complete dataset or knowledge of the system is usually compensated with the use of default parameters, proxies or assumptions that can lead to large uncertainties around input parameters and, therefore, model output (Saltelli et al., 2008; Sarrazin et al., 2016). These uncertainties may arise from natural variability, such as seasonal variation, or from errors in measurement instruments, defined as epistemic uncertainty (Ascough et al., 2008). Uncertainties in input parameters may have an influence on the uncertainty of the model output, thus, the model output is not a true value, but it is accurately given as a range of values or a probability distribution. The uncertainties of the input parameters, however, do not contribute equally to the uncertainties of model outputs. Thus, in models with many input parameters, such as an LCA, only a few important input parameters can explain the output variance, and other input parameters are of less importance in explaining that variance (Groen et al., 2014; Heijungs and Lenzen, 2014; Saltelli et al., 2000).

The identification of important input parameters prior to any environmental study can help to prioritize the data collection efforts (Heijungs, 1996; Sin et al., 2011), to calibrate the model (Saltelli et al., 2008) as well as to produce accurate results (Oenema et al., 2015). This is much important as the results from environmental assessments of food systems do support product and policy decisions (Curran, 2013).

In environmental assessment studies, such as LCA, a global sensitivity analysis (GSA) is often carried out *a posteriori* to identify input parameters that contribute most to the variance of the model output, and thus to support the interpretation of the results (Groen et al., 2014; Saltelli et al., 2008).

However, the necessary steps to improve data collection for these important parameters are usually not carried out. In this paper, we argue that by performing GSA during a scoping analysis prior to the main study, it is possible to identify input parameters that should require high-quality data to reduce the epistemic uncertainty but also to get the accurate variability of the model output. This procedure, therefore, would reduce the cost of data collection by focussing efforts to the important parameters.

The aim of this study is to illustrate a method to identify important input parameters that explain most of the output variance of environmental assessment models. The method is tested for the computation of life-cycle nitrogen (N) use efficiency indicators among mixed dairy production systems in Rwanda in a case study. These indicators includes life-cycle nitrogen use efficiency (life-cycle-NUE_N) refers to the efficiency upon which N input are recovered in end-products, life-cycle net nitrogen balance (life-cycle-NNB_N) refers to amount of N that is lost along the supply chain and nitrogen hotspot index (NHI_N) which quantifies the evenness of hotspots of N losses.

2. Methods

2.1. Case study: Mixed intensive dairy system in Rwanda

In this study, we applied GSA to life-cycle nitrogen use efficiency framework describing a mixed crop-livestock dairy system in Rwanda, from cradle to the primary processing stage of animal products. This system is relatively intensive compared to average dairy production in the country and is characterized mainly by the purchase of feed, high stock density, and high milk production per ha. Moreover, it consists of three production processes: feed production, animal production and processing of milk and meat. Feed production includes on-farm grasses, legumes and purchased concentrates, including soybean meal, maize bran or cotton meal. Animal production includes dairy cows, young stocks, and replacement animals. In this stage, manure effluents are collected and stored in lagoons or are dried after being mixed with the litter. Manure is then applied to the on-farm cropland or sold to other crop farmers. Activity data were collected through the direct interview and bookkeeping from 15 dairy farms located in the peri-urban area of Kigali city. Data collection covered the feed production, purchased concentrates, herd structure, herd parameters, milk production, milk processing, slaughterhouse, etc. Additional data related to emissions factors and other coefficients were obtained from secondary resources: emission factors (IPCC, 2006), N content of crops (Feedpedia, 2012), runoff and leaching coefficients (Gerber et al., 2013; Velthof et al., 2007).

2.2. Description of life-cycle nitrogen use indicators

In the framework developed by Uwizeye et al. (2016), N flows in crop production, animal production, and processing and includes internal processes, loops and recycling of N are calculated. The three indicators are estimated based on the following matrixes: N uptake at each process (PROD), internal N supply to each process (INP), N imports from other food systems to each process (IMP), changes in N stocks at each process (SC) and N mobilized from the nature or other livestock supply chains to each stage (RES). Thus, at supply chain level, the life-cycle-NUE_N is calculated as follows:

$$RES^*_i = RES_i \cdot (PROD_i - INP_i - IMP_i + \widehat{SC}_i)^{-1} \quad (\text{Eq. 1})$$

$$\text{Life-cycle-NUE} = 1/RES^*_{processing} \quad (\text{Eq. 2})$$

where $RES^*_{processing}$ refers to the amount of N mobilized to produce 1 kg of N in end-products at the processing stage.

The life-cycle-NNB_N is expressed as kg N losses per area of land used (ha) and is calculated as follows:

$$\text{Life-cycle-NNB} = \frac{\sum NNB_i \times AF_i}{A} \quad (\text{Eq. 3})$$

where NNB_i refers to N losses at i-th stage, AF_i refers to the biophysical allocation factor between co-products at i-th stage, and A refers to the total land used at a supply chain level.

Finally, NHI_N is calculated as follows:

$$NHI = \frac{\sigma(NNB_i)}{\mu(NNB_i)} \times 100 \quad (\text{Eq. 4})$$

where σ is the standard deviation of NNB_N for all i-th stages of a supply chain, and μ is the corresponding average of NNB_N for all i-th stages of a supply chain.

2.3. Global sensitivity analysis

We performed GSA, which is an approach to estimate the contribution of the uncertainty in each input parameter to the variance of a model output (Saltelli et al., 2008), and ranked the importance of input parameters based on the squared standardized regression coefficient (SRC). First, the probability distribution of 126 input parameters was defined based on activity data, emission factors and other coefficients. For activity data, we selected a probability density function (PDF) that gives a better goodness-of-fit for each input parameter. For emissions factors, a uniform distribution was used based on their ranges. For the coefficients without any other knowledge on their range (e.g. runoff rate), we assumed a normal distribution with a coefficient of variation of 20% based on IPCC guidelines (2006). We denote the number of input parameters subjected to GSA as Z , the size of the sample as N , and a number of the model evaluations during GSA as n . Second, we performed a Monte Carlo simulation (MCS) which consists essentially of generating N random numbers from a specified PDF of each input parameter. During MCS the uncertainty of Z input parameters are propagated through the model that results in the uncertainty of the life-cycle NUE_N , life-cycle- NNB_N , and NHI_N . Third, we performed a GSA using the squared SRC method (Saltelli et al., 2008). Several examples of the application of squared SRC in LCA studies of food systems are reported in literature, e.g. in Groen et al. (2014). The squared SRCs (S_Z) are estimated as follows:

$$S_{Z_{ij}} = \frac{\text{Var}(Z_i)}{\text{Var}(Y_j)} \cdot b_{Z_i}^2 \quad (\text{Eq. 5})$$

where $\text{Var}(Z_{ij})$ refers to the variance of i-th input parameter (Z), $\text{Var}(Y_j)$ refers to the variance of each output indicator (Y) and $b_{Z_i}^2$ refers to the linear regression coefficient of the i-th input parameter (Z).

The squared SRCs take values between 0 and 1, and their sum refers to the coefficient of determination R^2 (Saltelli et al., 2008). The squared SRCs are calculated to explain the variance of life-cycle NUE_N , life- life-cycle- NNB_N , and NHI_N . Based on squared SRCs, we ranked the input parameters in three categories based on two thresholds 0.01 (Cosenza et al., 2013; Sin et al., 2011) and 0.10. The category I includes less important input parameters with squared SRC <0.01, and the category II includes the important input parameters with squared SRC between 0.01 and 0.10, whereas the category III includes the most important input parameters with squared SRC >0.10.

3. Results

The GSA results were classified into three categories. The category I consisted of 115 less important input parameters with low squared SRCs (<0.01) that did not contribute to the variance of life-cycle N use efficiency indicators. The category II and III included the ranking of sensitive input parameters that contribute to the output variance of at least one of the life-cycle N use efficiency indicators and were illustrated in Figure 1. The category II consisted of 5 important input parameters

(P7-P11) with squared SRCs between 0.01 and 0.1, whereas, the category III includes 6 most important input parameters (P1-P6) with squared SRCs >0.1 which explained more than 80% of the variance of the three indicators.

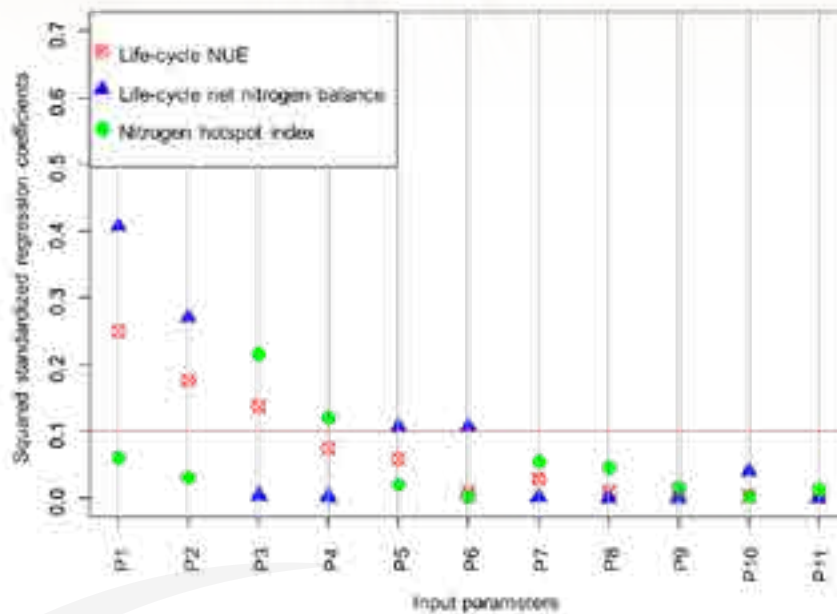


Figure 1. Results of the sensitivity analysis for life-cycle nitrogen use efficiency indicators. Only 11 sensitive input parameters that contribute more than 0.01 to the variance of at least one of the three indicators are shown. The red line is a proposed threshold that distinguishes the most important input parameters (P1 to P6 with squared SRC >0.1) and the important parameters (P7 to P11 with squared SRC between 0.01 and 0.1). Number of input parameters considered=126, N sample size=5000, n boots = 5000. The input parameters P1 to P11 are described in Table 1.

3. Discussion

The category I includes input parameters with low and high uncertainties. Given their low contribution to the variance of the three indicators, they would have low priority during the data collection phase and can be established with any value around their ranges such as regional averages or other secondary data including literature or regional database. This is comparable from the findings reported by Heijungs (1996) where input parameters with low sensitivity were assigned either no priority for those with low uncertainties or low priority for those with high uncertainties. For category II, important input parameters would have an intermediate priority and can be established with country specific data, whereas, those in category III with high priority, can be established with accuracy thus require high data quality. Our results are comparable to those of Heijungs (1996), where input parameters with high sensitivity were assigned high priority during the data collection stage.

By focusing effort to obtain accurate data on 11 out of 126 input parameters, it would help to cut on data collection cost and well as time. This approach may be used in any other life-cycle nitrogen use efficiency study for mixed crop-livestock dairy systems in other countries. The important input parameters of category II and III include emission factors and activity data. The latter should be established with primary data including on-site measurements, surveys or estimated from tier 3 process-based models.

Table 1: Description of the most important and important input parameters

Parameter	Description	Type of data	Unit	PDF ⁴	Range or Mean(sd)
P1	Indirect N emissions from applied manure to cropland ³	EF ¹	kg N. kg N ⁻¹ applied ¹	Uniform	0.05-0.5
P2	Indirect N emissions from synthetic fertilizer ³	EF	kg N. kg N ⁻¹ applied ¹	Uniform	0.03-0.3
P3	Share of solid manure at a farm level	Activity	%	Normal	50.3(27.4)
P4	Share of liquid manure at a farm level	Activity	%	Normal	17.3(32)
P5	Surface runoff rate at cropland ²	EF	%	Normal	25(5)
P6	N manure applied to cropland	Activity	kg N/ha	Normal	56.4(11.3)
P7	Indirect N emissions from solid manure ³	EF	kg N. kg N ⁻¹ excreted ¹	Uniform	0.1-0.4
P8	N content of a grass: <i>Pennisetum purpureum</i>	Activity	g N/kg DM	Normal	9.7(4.3)
P9	Digestible energy of feed ratio	Activity	%	Uniform	55-75
P10	N manure applied to grassland	Activity	kg N/ha	Normal	93.7(18.7)
P11	Maize gross yield	Activity	Kg/ha	Normal	8500(1420)

¹ EF refers to emission factor and other coefficients

² Estimated based on Velthof et al., (2007)

³ Uncertainty ranges obtained from IPCC guidelines (2006)

⁴ PDF refers to probability density function

These data should have a high score for data quality indicators such as those recommended by UNEP SETAC life cycle initiative (Sonnemann and Vigon, 2011) or Livestock Environmental Assessment Performance (LEAP) guidelines (FAO, 2016) to reduce the epistemic uncertainty and natural variability. For emissions factors, they are generally obtained from guidelines such as IPCC (2006) and are highly variable with a factor up to 10 (Table 1). Because of this high uncertainty, for example, input parameters P1 and P2 contributed to the variance decomposition of life-cycle NUE_N at 54%, life-cycle-NNB_N at 72% and NHI_N at 12%. This is similar to the findings reported by Groen et al. (2016), identifying N₂O and CH₄ emission factors as the main source of uncertainty in the computation of greenhouse gas emissions of pork production. The estimation of the emissions factors at the country level is essential and should be based on-site measurements or mathematical models; that take into consideration environmental conditions, such as climate, temperature, and management. These emissions factors would improve significantly the accuracy and reliability of the results. In case default values from IPCC guidelines are used, however, the choice of these values should be documented and supported with a global sensitivity analysis during the interpretation of results.

5. Conclusions

This study showed that carrying out a GSA during the scoping analysis of an LCA can help prioritizing data collection phase to reduce the epistemic uncertainty and to get a better grasp on natural variability of the model output by identifying important input parameters. These important parameters are either activity data that would be established with accurate data during the data collection phase or emissions factors and coefficient rates that would require additional data on environmental conditions or models to estimate them. By carrying out this analysis prior to the data collection, any LCA study in food sector can cut on the cost of data collection by focusing on input parameters that can be fixed through good practices in data collection. Further work on global life-cycle nutrient use performance will benefit from these results to generate analysis at lesser data collection cost.

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282. How LCA is used to define key performance indicators for food products

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ABSTRACT

This presentation will compare the PEF and TSC initiatives and their approaches, and will illustrate the different ways how environmental hotspots can be assessed. For the comparison between TSC and the PEF initiative, we focused on 1) the goal and scope of the initiatives, 2) the hotspot analysis and 3) the assessment of hotspots. The results indicate that the PEF and TSC initiatives especially differ in their main mechanism and their target audience while the hotspot analysis of the PEF dairy pilot and the TSC dairy category overlap strongly. Despite the hotspot overlap, they are assessed in the PEF pilots by calculating impact indicators for specific products, whereas TSC assesses these hotspots with a combination of KPIs focusing on mitigation activities, interventions and impact indicators. TSC steers sustainable development of a large number of food products by facilitating a data flow across multiple tiers of supply chains, while LCAs are done to identify environmental issues in specific situations and evaluate mitigation options. Despite the differences, the PEF initiative may stimulate the use of LCA in KPIs by focusing on harmonization of the LCA methodology to improve business to business and business to consumer communication of LCA results.

Keywords: key performance indicators; lifecycle assessment; environmental footprint; food.

1. Introduction

Retailers play an important role in sustainable consumption and production of products by the selection of products and cooperation with suppliers. To steer sustainable development of the product systems, highly efficient and thus concise product sustainability screening tools are needed. LCA is a tool that requires a large amount of data and effort, which makes it difficult for retailers to apply it to a large number of products or to activate suppliers to assess the environmental footprints of their products. To address this issue, existing LCAs and other types of sustainability assessment studies can be used to identify hotspots and improvement opportunities, which can then be translated into questionnaires with Key Performance Indicators (KPIs). This approach was adopted by The Sustainability Consortium (TSC, 2016). TSC is a global organization of manufacturers, retailers, suppliers, service providers, NGOs, civil society organizations, governmental agencies and academia. Together they build science-based screening and decision tools that address sustainability issues that are materially important throughout a product's supply chain and lifecycle.

The resulting tools contain KPIs that can be qualitative, asking whether certain actions have been taken, or quantitative, such as specific inventory data or LCA based impact indicators. There are important advantages of using LCA based impact indicators as KPIs, such as the objectiveness, transparency and a clear differentiation of the indicators. However, the number of impact indicator KPIs in the TSC system has been limited so far, due to several important drawbacks:

- 1) the expertise, effort and time required to report such indicators are often limited for suppliers;
- 2) the results are often variable due to lacking globally accepted methodology and different data sources;
- 3) the interpretation is in many cases difficult due to product and functional differences; and
- 4) the identification of improvement actions is generally problematic because of factors beyond the control of the company.

The recent Single Market for Green Products, also known as Product Environmental Footprint (PEF) initiative of the European Commission was initiated to address most of these issues (EC, 2016). This paper gives an example of how this development affects the way LCA is used to define KPIs for food products. It also compares the PEF and TSC initiatives and their approaches, and will illustrate the different ways how environmental hotspots can be assessed.

2. Methods

TSC and the PEF initiative have been compared by analyzing relevant reports, papers and other communications about the initiatives and interviews with various stakeholders. We focused in this comparison on the following aspects of the initiatives:

- 1) the goal and scope of the initiatives,
- 2) the hotspot analysis and
- 3) the assessment of hotspots.

The goal and scope of the initiatives has different aspects, such as target audience, sustainability themes, mechanism, indicators, use and quality criteria. The initiatives' websites were studied per aspect and researchers/consultants who are involved in the initiatives were interviewed to verify our interpretations.

As an important example of a food product group we chose the dairy products group to compare the hotspot analysis and assessment. This was based on the dairy KPI toolkit of TSC (only accessible to members and users) and the dairy pilot's draft Product Environmental Footprint Category Rules (PEFCR; EDA et al., 2015a) and PEF screening study (EDA et al., 2015b). We also followed the pilot by reading the draft documents and interviewing members of the technical secretariat (representatives of pilot member companies and sector organizations).

3. Goal and scope comparison

The results of the goal and scope comparison are shown in Table 1. The PEF and TSC initiatives especially differ in their main mechanism and their target audience. Whereas the PEF initiative aims to make Environmental Product Declarations (EPDs) more consistent to create a level playing field for companies and more clarity for consumers, PEF does not make explicit in which concrete situations (besides consumer-directed labeling through EPDs) the resulting improved LCA standard will be used, as also noted by Lehman et al. (2016). It could also be used for business-to-business (B2B) communication and policy measures. On the other hand, TSC's main mechanism and target audience have been defined and clarified over some years: the outcomes of the measurement system are used by retail buyers and their suppliers to draft sustainability improvement plans.

Because of the differences in main mechanism and target audience, different sustainability issues and indicators are selected: The PEF initiative focuses on LCA-based environmental issues through quantifying LCA impact category indicators. TSC uses LCA impact indicators in some cases but, in addition, it employs non-LCA-based (environmental and social) indicators that can be oriented to activities that determine or improve sustainability performance, with the aim to efficiently and effectively address all kinds of sustainability issues.

TSC and the PEF initiative also have a different focus in terms of life cycle phases, product classifications, and its basis for data needs. The PEF methodology is designed to assess the environmental impact of the entire life cycle of individual products (with a distinct function). On the other hand, the TSC system allows for product focused assessments as well as assessments of a more broadly defined product category (containing products with varying functions). The PEF category rules prescribe a selective data collection approach, substantiated by a new in-depth screening LCA study, whereas TSC focuses its measurement on specific hotspots in the life cycle, which are derived from pre-existing scientific literature (based on LCAs and other sustainability assessments).

The marked differences between the two initiatives can be explained by the clearly different goals and implementation levels of the initiatives. TSC has developed its measurement system's scalability and accessibility over the past years, and it is currently in use by more than 2000 reporting suppliers. It is expected that the PEF methodology will be used more intensively in the coming years. The two initiatives correspond in their focus on LCA, and this common ground will be a basis for methodology improvement.

Table 1: Goal and scope of the two initiatives: Product Environmental Footprint (PEF) and The Sustainability Consortium (TSC)

Aspect	PEF	TSC
Scope	Products sold on European market	Global consumer products
Target audience	Consumers / Business-to-Business	Retail buyers
Sustainability themes	LCA environmental	Social and Environmental
Main mechanism	Improve transparency to customers and business partners	Support retailer-buyer discussion on improvement plans
Indicators	14 Impact categories Communicate 3-5 to consumers?	10-15 activity and impact KPIs
Use	In development	+2000 reporting companies
Quality criteria	Data consistency & quality, method consistency	Answerability, Transparency, Differentiation, Actionability

4. Hotspot analysis

The hotspot analysis of the PEF dairy pilot and the TSC dairy category largely overlap, as illustrated in Table 2. The hotspots are assessed in the PEF pilot by calculating impact indicators for specific products, whereas TSC assesses these hotspots with a combination of KPIs focusing on mitigation activities, interventions and impact indicators. For example, TSC's fertilizer application KPI asks for nitrogen use efficiency and phosphorus surplus, whereas PEF requires more detailed data to calculate eutrophication and acidification potentials. Especially the hotspots related to eutrophication, acidification and climate change therefore have a strong overlap.

Social issues, such as workers health and safety, animal welfare, and antibiotics use in farm operations (risking resistance to antibiotics in humans) on the other hand are topics not covered by PEF. Antibiotics use is a topic that is very specific to animal products and therefore not addressed in LCA. Quantitative assessments of social issues and animal welfare have been applied in life cycle approaches (e.g. Dreyer et al, 2010; Blonk et al, 2010), but it involves value attribution to weight different qualitative aspects, which are not yet widely accepted and therefore prevent them from widespread adoption like environmental LCA.

Water scarcity in the dairy processing phases was not selected as hotspots by TSC. There may be several reasons behind this, one of them can be the high variability between dairy products, which can make it difficult to interpret the responses and identify improvement actions.

Land transformation on the other hand is often related to specific feed ingredients, so targeted actions can be defined to reduce related impacts. This gives an important reason to include a KPI for this hotspot.

Table 2: Illustration of overlap between PEF and TSC hotspot analysis

Phase	Impact category	Overlap
Feed cultivation	Climate change	Good
	Eutrophication	
	Acidification	
	Land transformation	
	Water scarcity	
	Social issues	None
Animal farm operations	Eutrophication	Good
	Acidification	
	Climate change	
	Water scarcity	
	Antibiotics use	None
	Animal welfare	
Social issues		
Dairy processing	Climate change	Good

	Eutrophication	
	Water scarcity	None

5. Assessing LCA based KPIs

TSC includes some LCA based KPIs. Most do not include impact assessment, because the inventory indicators are considered reliable enough to drive improvement; for example, nitrogen use efficiency, phosphorus surplus, and water use. The most important impact assessment based KPIs in TSC are based on the Global Warming Potential (GWP) 100 factors of the IPCC (2007), as applied widely in LCA (often referred to as carbon footprint). Generally, in lifecycle phases where climate change is a hotspot, there is a KPI that asks for the carbon footprint indicator from cradle-to-gate or gate-to-gate, depending on the lifecycle phase.

The dairy products hotspot climate change in the feed cultivation and animal farm phases is addressed by a KPI that asks for the carbon footprint of raw milk. Currently, we observe that there is a lot of diversity in the data quality and methods applied for assessing the raw milk carbon footprint, as noted before (Yan et al., 2011). We expect that the PEFCR will improve the guidance for calculation of this KPI, because it can bring methodological consistency when it is broadly accepted. However, large data gaps are likely to remain for dairy and the existing calculation tools are not yet adapted to the PEFCR.

The second contributor of dairy products to the climate impacts is dairy processing. This hotspot is covered by a KPI that asks for the gate to gate carbon footprint of dairy processing. The PEFCR gives important data and instructions to make the assessment easier and more consistent. It should be noted that the scores of different dairy products, such as cheese, butter, fresh milk, and yoghurt, are hard to compare. The KPI is therefore more suitable for identifying improvement options and benchmarking suppliers with historical performance than for benchmarking with average supplier scores.

LCA-based KPIs that focuses on the life cycle from cradle to grave are currently not included in TSC. Such a KPIs are not considered valuable, because it can be hard to derive actions from such KPIs, and aggregated results of different products can be difficult to interpret because of the wide variety of products. However, TSC does include KPIs that focus on improvement activities in a specific aspect, like packaging, from cradle to grave.

Besides LCA based KPIs on climate change, there is a potential for developing LCA based KPIs on other environmental impact categories, such as toxicity impacts of pesticides use, a water scarcity indicator, and a land use indicator. However, there are still large data gaps, lack of methodological consensus, and lack of operational tools due to the large uncertainty and complexity of assessing such indicators. The PEF initiative has clearly identified these issues and contributes to a solution. As the methods and tools for these impact categories continue to improve, it will become feasible in the future to ask a large number of suppliers to assess such indicators.

6. Conclusions

TSC steers sustainable development of a large number of food products by facilitating a data flow across multiple tiers of supply chains to stimulate and support sustainable development throughout the products' life cycles. LCAs are generally done to identify environmental issues in specific situations and evaluate mitigation options. This implies different demands on the efficiency and data intensity of how the hotspots are assessed (Dooley and Johnsson, 2015). Nevertheless, the PEF initiative has several important links with TSC:

- it produces valuable hotspot analyses that confirm or help improve TSC's hotspot analyses,
- it supports suppliers to answer the existing LCA based KPIs through methodological guidance, data and tool development, and
- it helps removing barriers for implementing LCA based KPIs on other categories than climate change.

So, the developments inside and around the PEF initiative seem to collectively lead to an improved quality standard for LCA in the coming years. Additionally, the initiative stimulates the development

of tools that make environmental footprint assessments more accessible to industries. Due to the increasing availability of tools, the improved harmonization and reduced reporting efforts, implementing LCA indicators as KPIs may pay off earlier than anticipated before. Therefore, intensive exchange and alignment between LCA initiatives and initiatives developing sustainability KPIs is crucial for making LCA based KPIs a success.

7. Acknowledgements

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POSTER PAPERS

WATER AND EUTROPHICATION

1. Water and Eutrophication

P01. Relevance of the Spatialized Territorial LCA method: case study of eutrophication in a French catchment

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To help local stakeholders manage agricultural activities, environmental impacts should be assessed at the territorial level by considering its biophysical characteristics. The Spatialized Territorial Life Cycle Assessment (STLCA) method (Nitschelm et al. 2015), integrates, (1) at the Life Cycle Inventory (LCI) stage, the locations of direct emissions as well as biophysical characteristics of the surroundings (e.g. soil type) into calculations of direct emissions, and (2) at the Life Cycle Impact Assessment (LCIA) stage, fate of pollutants as well as the territory's sensitivity (i.e. the biophysical context) to the impact(s). The objective of our study was to verify the utility of STLCA by applying it to the Lieue de Grève catchment, France. We present the results of LCA and STLCA of the territory. We focus on eutrophication since it is the major impact on this territory.

Non-spatialized emissions for crops and livestock were determined using emissions models of the French agricultural database Agri-BALYSE (Koch and Salou 2015). Spatialized nitrogen emissions (i.e. NO_3^- , NH_3 and N_2O) were predicted using Syst'N (Parnaudeau et al. 2012), a simulation model that includes agricultural practices and biophysical parameters. Non-spatialized eutrophication impacts were determined using the CML-IA characterization factors (Udo de Haes et al. 2002). Two spatialized characterization factors were derived from CML-IA for marine and freshwater eutrophication (Nitschelm et al., in prep.). Spatialized characterization factors include fate factors as well as a sensitivity factor specific to the territory. Fate factors for nitrogen and phosphorous were determined using the Nutting-N and Nutting-P models (Dupas et al. 2015). Sensitivity factors were derived using (1) the Håkanson method (Håkanson 2008) for marine eutrophication and (2) the French SYRAH method (Fabre and Pelte 2013) for freshwater eutrophication.

Three main results arose from this study. First, with spatialized results, a map can be generated that shows the locations of emission and/or impact hotspots within the territory. Second, at the territorial level, there was no difference in magnitude between non-spatialized impacts and impacts spatialized at the LCI stage. Including fate and sensitivity of surroundings in the characterization factor decreased direct eutrophication impacts (i.e. within the territory) by $10 \text{ kg PO}_4^{2-} \text{ eq. per ha}$ (Fig. 1). Given the variability in farm practices within the territory, the decrease in eutrophication from non-spatialized to spatialized LCIA is not significant. Third, at the farm level, eutrophication impacts per ha differed depending on whether they were non-spatialized, spatialized at the LCI stage, or spatialized at the LCI and LCIA stages (Fig. 2). These differences are explained by differences in farm practices and biophysical characteristics of the surroundings.

From these results, we can conclude that spatialization is not always necessary and depends on the goal and scope of the study. For a study at the territorial level, non-spatialized methods can be sufficient. For a study considering farms and their practices, spatialized methods can better represent farm diversity, as well as hotspot location, within a territory.

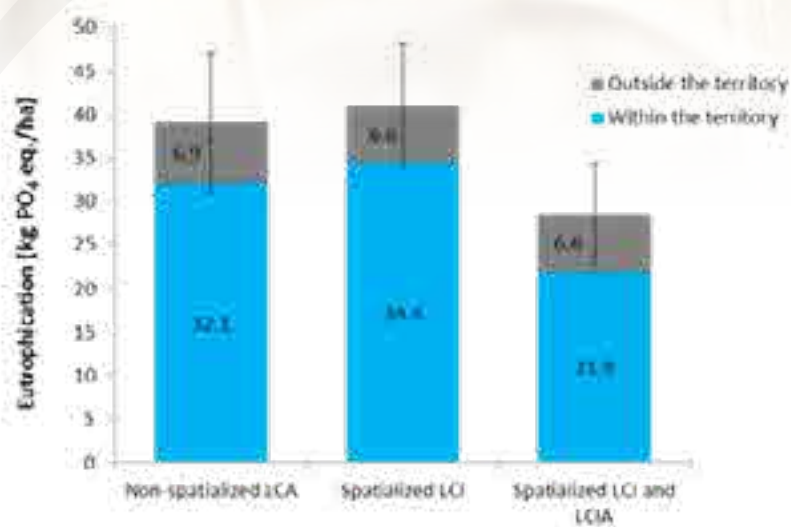


Figure 1: Mean eutrophication impacts (kg PO₄²⁻ eq.) per hectare of the Lieue de Grève territory. Non-spatialized LCA: Agri-BALYSE for emissions and CML-IA for eutrophication; Spatialized LCI: Syst’N for nitrogen emissions, Agri-BALYSE for other emissions and CML-IA for eutrophication; Spatialized LCI and LCIA: Syst’N for nitrogen emissions, Agri-BALYSE for other emissions and spatialized characterization factors for marine eutrophication in the Lieue de Grève catchment. Error bars represent maximum and minimum impacts of farm types in the territory.

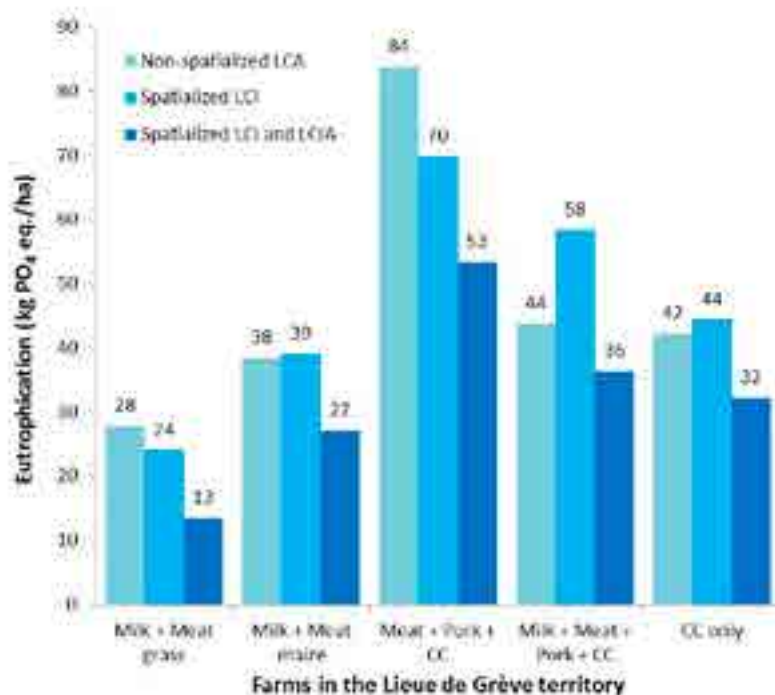


Figure 2: Eutrophication impacts (kg PO₄²⁻ eq.) per hectare of representative farm types. Milk: milk from dairy cows; Meat: from dairy or beef cows; CC: cash crops; grass: livestock feeding based on grass; maize: livestock feeding based on maize.

P02. Spatialized eutrophication characterization factors for more relevant agricultural LCAs

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Agriculture is one of the main contributors to aquatic eutrophication. In LCA, eutrophication has commonly been represented by placing the indicator early in the cause effect chain, at the system boundary between the technical system and the environment. Since eutrophication is a regional environmental impact, the magnitude of impact from emissions is highly dependent on the site of emission. In order to increase the environmental relevance of marine eutrophication assessment for nutrient leaching from soil for Sweden, regional characterisation factors (CFs) were derived at a high spatial resolution, with regards to the site specific nutrient transport and specific nutrient limitation in the Baltic Sea sub-basins. The derived CFs showed substantial variation, ranging from 0.056 to 0.986 kg N_{eq}/kg N and 0 to 7.226 kg N_{eq}/kg P respectively.

Applying these CFs to spring barley and grass ley cultivation at 8559 sites in Sweden gave eutrophication impacts of 1,2-58 kg N_{eq}/ha for spring barley and 0,31-32 kg N_{eq}/ha for grass ley. The analysis showed that N had a higher impact than P at the majority of the sites, in 98 % and 85 % of the sites, for spring barley and grass ley respectively.

Including spatial characteristics of the eutrophication cause effect chain was shown to highly affect the calculated impacts. A more environmentally relevant eutrophication impact assessment in LCA may improve the usefulness of LCA:s in eutrophication policy, as well as increase the importance of eutrophication when assessing overall environmental impact from agricultural goods.

P03. Environmental map towards water and food self-sufficiency: integrating rainwater harvesting and urban agriculture under different climatic conditions

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I. Objective

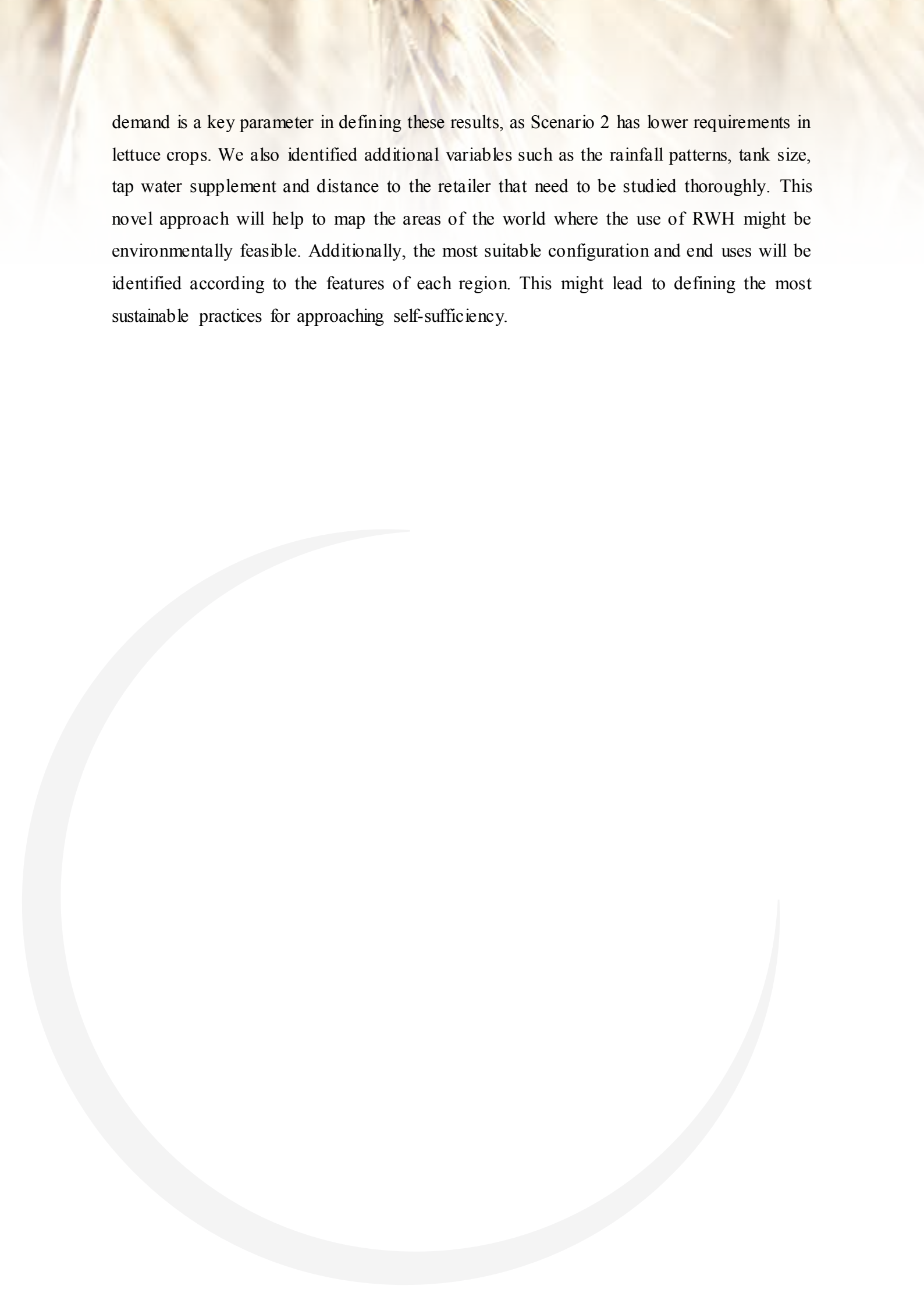
Population growth, urbanization and climate change stress the need for alternative water systems and food production areas. Fresh produce is imported from distant agricultural areas, whereas water availability is decreasing due to droughts or poses a threat due to sudden and intense rainfall events. In this context, the implementation of urban agriculture and rainwater harvesting (RWH) at a local scale might provide ecosystem services through the so-called water-energy-food nexus. The goal of this contribution is to assess the environmental performance of different RWH scenarios at a household level under different climatic conditions using Life Cycle Assessment (LCA).

II. Method

Cities that are representative of eleven major climatic regions were selected. These cities belong to Spain, the USA, India and Pakistan, which have different water consumption patterns. A RWH system was designed for a standard single-family house and its life cycle impacts were estimated considering three different scenarios (Figure 1). Rainwater end uses were toilet flushing and laundry (indoor demand; Scenario 1), irrigation of a small urban garden with lettuce crops (outdoor demand; Scenario 2) and a combination of outdoor and indoor water uses (Scenario 3). When rainwater was not able to meet the demand, it was supplemented with tap water. Avoided stormwater treatment and vegetable transportation were accounted for as benefits (Figure 2). The functional unit was 1 m³ of water demand supplied with rainwater and potable water.

III. Main result and implications

In general, the preliminary environmental impacts of Scenarios 1 and 3 were the lowest in all cities. In terms of Global Warming Potential, Scenario 2 resulted in 1.83 kg CO₂eq./m³, whereas we estimated an average 0.34 and 0.40 kg CO₂eq./m³ in Scenarios 1 and 3. Water



demand is a key parameter in defining these results, as Scenario 2 has lower requirements in lettuce crops. We also identified additional variables such as the rainfall patterns, tank size, tap water supplement and distance to the retailer that need to be studied thoroughly. This novel approach will help to map the areas of the world where the use of RWH might be environmentally feasible. Additionally, the most suitable configuration and end uses will be identified according to the features of each region. This might lead to defining the most sustainable practices for approaching self-sufficiency.

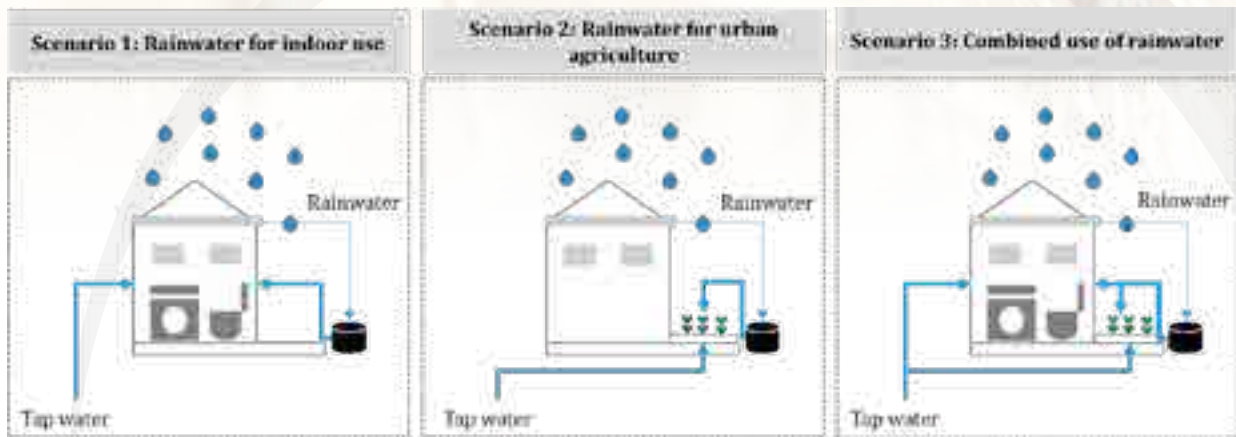


Figure 1 Rainwater use scenarios considering indoor water, urban agriculture and a combination of both

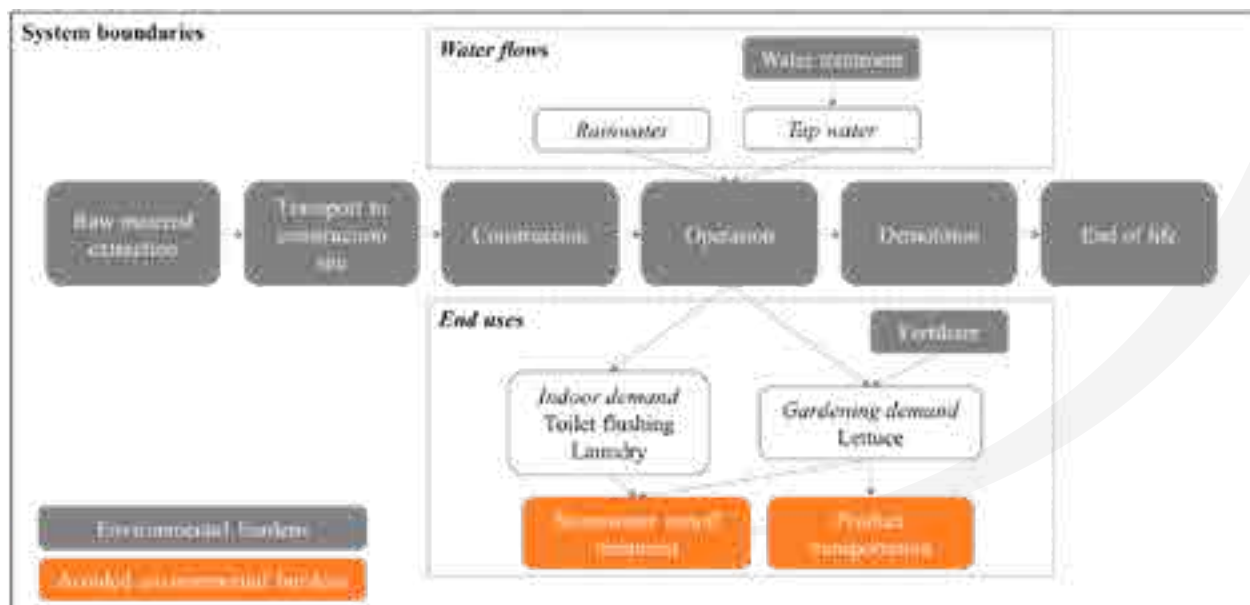


Figure 2 System boundaries of the LCA, including the environmental impacts and avoided burdens of the RWH scenarios

P04. The environmental impacts of silicon as an alternative to phosphorus fertilizers

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ABSTRACT

Phosphorus (P) availability in agricultural soils is reduced due to immobilization and inaccessibility to plants. Large amounts of inorganic P fertilizers are used by farmers, originated from the mining of phosphorite deposits. The exhaustion of this non-renewable resource combined with the increasing world population, transform P into a finite resource in the short term (next 40-400 years). Alternatives are thus needed. Here we study the environmental effects of switching from P to silicon (Si) fertilizers. Si is known to alleviate P deficiency by increasing P mobilization. Fertilization with Si is frequent in America and Asia, mainly in fields growing Si-accumulator crops i.e. rice, sugarcane, wheat, maize, with positive results. However, Si fertilizers are still neglected in Europe as an option. We performed a comparative Life Cycle Assessment (LCA) of the production and application of P and Si fertilizers. The functional unit is one kilogram of P fertilizer. We test different options for P fertilizer (e.g. single superphosphate, triple superphosphate, etc.). To establish an equivalence, we considered different amounts of Si fertilizer required to free enough P for plants to take up as the amount required by the functional unit of P fertilizer. We used the ILCD impact assessment methods adding characterization factors to depict the impacts of Si lost from fields in eutrophication and related impact categories. Our results show that Si fertilizers are, environmentally, a viable alternative to P fertilizers, even if the amount of necessary Si-fertilizer to mobilize P is higher than the P-fertilizer itself. The Si fertilizer with the lowest impacts in general is calcium silicate produced from lime, sand and water. Calcium silicate also show particularly very low impacts in the categories of human toxicity, freshwater ecotoxicity and water, mineral and fossil renewable resource.

Replacement of non-renewable P sources is an important challenge for modern agriculture. Here we concluded that Si fertilizers are an overall environmentally positive option especially in soils where P-fertilizers have been extensively used in the past and a high immobile P pool exists. This option should, however, be the focus of more technical agronomic viability studies to study biogeochemical competition between Si and P adsorption and the soil conditions that favor the process.

Keywords: Silicon fertilization, Environmental impacts, Life cycle Assessment, Phosphorus mobilization

1. Introduction

Phosphorus (P) availability in agricultural soils is reduced due to its constant immobilization and inaccessibility to plants despite their high requirement on this nutrient. Farmers usually add large amounts of inorganic P fertilizers to soils to supply the plants necessity, originated from the mining of phosphorite deposits. The exhaustion of this non-renewable resource, combined with the increasing world population, transform P into a finite resource in the short term (next 40-400 years) (*Obersteiner et al 2013*). Moreover, only a small part of the P-fertilizer eventually reaches the plants, since the majority is immobilized in several processes taking place along the soil-water interface and/or is lost through erosion and runoff. Also, when part of the available P is leached it might contaminate aquifers and coastal waters causing severe problems of eutrophication. Alternatives are thus needed to overcome this economic and environmental problem.

In this study we investigate the environmental effects of P re-solubilization using a competitor adsorber, namely silicon (Si) fertilizers. Si is not considered an essential element for plants (*Epstein 1999*), but studies from recent decades have proved that the availability of this element contributes to several beneficial effects, especially on Si-accumulator crops ($[Si] > 1\%$ and $[Si]/[Ca] > 1$) i.e. rice, maize, wheat, sugarcane, barley (etc) (*Guntzer et al. 2011*). The effects of fertilization with Si compounds on these cultures have been studied mainly in Asia and North and South America, with positive results in increasing resistance to pathogens and insects (*Rodrigues et al. 2003*), alleviation of drought and salt stress (*Ahmed et al. 2011; Ashraf et al. 2009*) and alleviation of Al, Cd and Zn (etc) toxicity (*Cocker et al. 1998; Feng et al. 2010; Gu et al. 2012*). However, a few studies also point out the effect that Si may have in increasing crop yield when P availability is low, thus contributing for the alleviation of P deficiency (*Brenchley & Makell 1927; Fischer 1929; Eneji et al. 2008;*) and avoiding P fertilization. Since the major processes that control solution P concentration in soils are adsorption and desorption from metal oxides - Al and Fe oxi-hydroxides (*Hinsinger et al. 2001*), the presence of available Si in soil might trigger competition between Si and P and eventually release P

into solution. This effect has been successfully identified in sediments in the Bay of Brest where the enrichment of Si in the sediment interface induced a high mobilization of P into solution (Tallberg *et al.* 2008). There is, at the moment, no follow up of this apparent alternative as a potential reducer of P-fertilization in croplands and grasslands where P is present but immobilized. However, in order to properly consider the full environmental effects of partial replacement of P-fertilization with Si-fertilization as a real *in-situ* solution, we have conducted a preliminary Life Cycle Assessment (LCA) study to evaluate the environment implications of such action.

2. Methods

2.1 Si-fertilizers considered

In this study we depicted the impact of Si-fertilization in Si-accumulator crops (Guntzer *et al.* 2011). We have thus considered the most important crops in Europe (depending on area harvested, yield and production): a) wheat; b) barley; c) sugarcane and d) sugar beet; for these crops, we identified the most often used Si fertilizers: sodium silicate ($\text{Na}_2\text{O}_3\text{Si}$) and calcium silicate (Ca_2SiO_4), based on a survey over more than 100 articles studying the Si-fertilization implications on different crops (Barão 2016, *in prep.*).

For sodium silicate (solid fertilizer) we used the “Sodium silicate, solid {RER}” Life Cycle Inventory (LCI) dataset from the database ecoinvent v3 with no additional considerations. For calcium silicate we used two alternative production processes in order to correctly use the software and identify the environmental impacts. First (“Method 1”), we directly used the ELCD v3 database dataset for “Calcium silicate, blocks and elements, production mix, at plant, density 1400 to 2000 kg/m^3 RER”. The processes depicted in this dataset are not directly applicable to fertilizer production, but they reveal a production process that could be used in fertilizer production. In it calcium silicate is obtained using lime, sand and water. Second (“Method 2”), we constructed a process for calcium silicate production using lime, hydrochloric acid and sodium silicate (Table 1). The datasets used from the ecoinvent v3 database for the first two materials were “Lime {CH}| production, milled, loose” and “Hydrochloric acid, without water, in 30% solution state {RER}”, respectively. Assuming an annual production of 69 ton/year, the request of raw materials are: 64 ton/year of lime, 24 ton/year of hydrochloric acid and 12 ton/year of sodium silicate (Ethiopian Embassy 2016).

Table 1: Raw material and quantity required to produce 69 ton of calcium silicate per year

Raw Material	Quantity (tons)
Lime	64
Hydrochloric acid	24
Sodium silicate	12

The amount of Si (atomic weight of $28.085 \text{ g mol}^{-1}$) in 12 tons of sodium silicate ($\text{Na}_2\text{O}_3\text{Si}$, molar weight $122.06 \text{ g mol}^{-1}$) is 2.76 tons. Assuming that the only source of Si in calcium silicate fertilizer is Si from sodium silicate, 2.76 tons is also the amount of Si present in each 69 tons of calcium silicate fertilizer. This means that there are $40 \text{ kg Si ton}^{-1}$ fertilizer, or conversely that each 25 tons of fertilizer provide 1 ton of Si to the soil. These data were used to convert results, which are obtained per mass unit of fertilizer, into impacts per mass unit of Si in each of the three cases.

2.2 P-fertilizers considered

We studied two frequently used P fertilizers, namely single superphosphate (SSP) and triple superphosphate (TSP) using the ecoinvent v3 datasets “Phosphate fertiliser, as P_2O_5 {RER}| single superphosphate production” and “Phosphate fertiliser, as P_2O_5 {RER}| triple superphosphate production”. In both cases, given the atomic mass of P (30.974) and the molar mass of P_2O_5 (283.89), we used a factor of $0.109 \text{ kg P kg}^{-1} \text{ P}_2\text{O}_5$ to depict results in terms of units of P.

2.3 System boundaries, functional unit and impact assessment

We performed a comparative analysis Si and P fertilizers by determining the difference of the impacts generated in producing one mass unit of Si and P in each fertilizer type. We place the system boundaries at factory gate and thus do not consider the impacts caused by the application of the fertilizers given the absence of data to create an LCI for Si application. In the future, we suggest to include this important part in LCA studies, which will be crucial to determine the real impact of Si in environment.

The comparison between impacts must be established using a substitution ratio of P with Si. However, there is no data to determine an actual equivalence that depicts how many units of Si are required to remobilize one unit of P. We define this quantity as the equivalence ratio. Ratios are presented as “kg Si : kg P”. A ratio of $n:1$ means that n kg of Si are required to free 1 kg of P. The lower the ratio, the more efficient Si is assumed to be in setting P free for uptake by plants. In our results, we define the equilibrium ratio as the ratio for which the impacts of producing n unit of Si in the Si fertilizer are equal to the impacts per unit of P in P fertilizer. Consequently, a higher equilibrium ration means that the production of Si fertilizer has lower impacts even if many units of Si are required to resolubilize each unit of P. The minimum plausible ratio is 1:1, assuming that one unit of Si cannot mobilize more than 1 unit of P. This fair assumption is based on research stating that metal oxides and other soil sorbents have higher affinity to P than for the most other competing inorganic ligands (*Hinsinger, 2001*).

We additionally studied the potential impacts of differences in transportation of both fertilizers, as the total outcome necessarily depends on the provenience of the materials and the location of the field where they are applied. To avoid making assumptions regarding both, we determined the number of kilometers by road or sea that Si fertilizer would need to travel (additionally to distances traveled by P fertilizer) to make the equilibrium ratio move from $n:1$ to $(n-1):1$. We used the road transportation LCI dataset “Transport, freight, lorry >32 metric ton, EURO5 {GLO}” from ecoinvent; for sea transportation, we used the “Transport, freight, sea, transoceanic ship {GLO}” dataset.

We used software SimaPro 8.0 to perform the impact assessment calculations for P and Si fertilizers using LCI processes constructed with the attributional and consequential methods (we were unable to use the latter for Method 1 of production of the calcium silicate fertilizer). The Life Cycle Impact Assessment (LCIA) framework used was ILCD midpoint (EC, 2011). We present results for all impact categories in ILCD next, analyzing in detail the climate change and freshwater eutrophication categories as demonstrative.

3. Results

Table 2 displays the results of the LCA study of all fertilizers considered in this paper. For SSP and TSP, results are shown per kg of P; for $\text{Na}_2\text{O}_3\text{Si}$ and Ca_2SiO_4 fertilizers, results are portrayed per kg of Si. SSP and TSP have much higher impacts than any Si fertilizer, but the difference between the two is minimal. For this reason, we excluded TSP from the rest of the results and present only results for SSP from hereon after. The Si fertilizer with the lowest impacts in general is calcium silicate produced from lime, sand and water, and the one with the highest impacts is also calcium silicate if produced using hydrochloric acid and sodium silicate.

A different way of looking at these results is shown in Table 3, which includes the equilibrium ratios for each combination of Si:P fertilizer. For some impact categories derived using attributional LCIs, such as human toxicity, freshwater ecotoxicity, water, mineral, fossil and renewable resource depletion, the equilibrium ratio using Ca_2SiO_4 is extremely high given the very low impacts involved in the production of this fertilizer. It should be noted, however, that the LCI dataset used for this fertilizer belongs to a different database (ELCD rather than ecoinvent), so there may be methodological issues that justify these striking results. To be conservative, we do not include Method 1 of Ca_2SiO_4 production in the rest of this Results section.

Table 3: Equilibrium ratios for combinations of fertilizers analyzed using attributional and consequential inventories (N/A – Not applicable; SSP – Single superphosphate)

Impact category	Attributional			Consequential	
	Na ₂ O ₃ Si: SSP	Ca ₂ SiO ₄ (1): SSP	Ca ₂ SiO ₄ (2): SSP	Na ₂ O ₃ Si: SSP	Ca ₂ SiO ₄ (2): SSP
Climate change	4:1	18:1	3:1	10:1	2:1
Ozone depletion	8:1	90:1	N/A	39:1	N/A
Human toxicity, cancer effects	15:1	4193:1	10:1	31:1	7:1
Human toxicity, non-cancer effects	13:1	355:1	8:1	8:1	9:1
Particulate matter	15:1	608:1	10:1	22:1	7:1
Ionizing radiation HH	27:1	356:1	6:1	N/A	3:1
Ionizing radiation E	19:1	62:1	5:1	N/A	3:1
Photochemical ozone formation	7:1	71:1	5:1	11:1	4:1
Acidification	10:1	198:1	7:1	15:1	8:1
Terrestrial eutrophication	5:1	63:1	4:1	3:1	N/A
Freshwater eutrophication	26:1	4906:1	13:1	110:1	7:1
Marine eutrophication	7:1	68:1	5:1	10:1	4:1
Freshwater ecotoxicity	12:1	25322:1	10:1	11:1	7:1
Land use	6:1	N/A	6:1	1:1	N/A
Water resource depletion	26:1	239126:1	5:1	N/A	2:1
Mineral, fossil & ren resource depletion	30:1	66622:1	28:1	8:1	10:1

In Figure 1 we plotted the difference of environmental impacts (impact of Si fertilizer minus impact of P fertilizer, both in kg of respective element) as a function of the substitution ratio, using the climate change and freshwater eutrophication impact categories as examples. The ratio where the line cut the x axis is the equilibrium rate.

In the climate change category, we observe that the attributional method places the interval up to which the impact of producing Si fertilizers is lower than using P at between 3:1 and 5:1. The consequential approach broadens the interval, placing it at approximately 2:1 to 11:1. In the eutrophication impact category, Si fertilizers avoid impacts even if the substitution rate is significantly higher (13:1-26:1, approximately, using an attributional approach – and more than 100:1 if a consequential approach is used).

Finally, Table 4 presents the role of transportation in shifting the ratio of substitution due to increased transportation requirements of one of the fertilizers (assumed Si). We observe that at minimum the additional transportation required is always in the order of the thousands, which means that transportation is unlikely to play a significant role in the sustainability of Si fertilizers as an alternative to P.

Table 2: Life Cycle Impact Assessment of all fertilizers analyzed using attributional and consequential inventories

Impact category	Units per kg P/Si	Attributional				Consequential				
		SSP	TSP	Na ₂ O ₃ Si	C ₂ SiO ₄ Method1	C ₂ SiO ₄ Method2	SSP	TSP	Na ₂ O ₃ Si	C ₂ SiO ₄ Method2
Climate change	kg CO2 eq	1.63E+01	1.44E+01	3.39E+00	9.01E-01	4.22E+00	1.62E+01	1.34E+01	1.49E+00	6.30E+00
Ozone depletion	kg CFC-11 eq	1.41E-06	1.05E-06	1.73E-07	1.56E-08	1.62E-06	8.68E-07	7.57E-07	2.19E-08	3.71E-06
Human toxicity, cancer effects	CTUh	2.77E-06	2.23E-06	1.78E-07	6.61E-10	2.58E-07	3.01E-06	2.07E-06	9.58E-08	4.06E-07
Human toxicity, non-cancer effects	CTUh	1.82E-05	1.36E-05	1.37E-06	5.13E-08	2.03E-06	2.20E-05	1.31E-05	2.64E-06	2.38E-06
Particulate matter	kg PM2.5 eq	3.66E-02	2.64E-02	2.30E-03	6.03E-05	3.61E-03	3.63E-02	2.44E-02	1.62E-03	4.92E-03
Ionizing radiation HH	kBq U235 eq	6.73E+00	3.93E+00	2.44E-01	1.89E-02	9.94E-01	6.09E+00	1.08E+00	1.10E+00	1.70E+00
Ionizing radiation E (interim)	CTUe	1.16E-05	7.23E-06	6.11E-07	1.86E-07	2.04E-06	1.06E-05	2.35E-06	-2.09E-06	3.49E-06
Photochemical ozone formation	kg NMVOC eq	7.17E-02	7.24E-02	8.97E-03	1.00E-03	1.33E-02	7.27E-02	6.67E-02	6.30E-03	1.46E-02
Acidification	molc H+ eq	2.45E-01	2.46E-01	2.35E-02	1.24E-03	3.28E-02	2.35E-01	2.24E-01	1.52E-02	2.69E-02
Terrestrial eutrophication	molc N eq	2.35E-01	2.35E-01	4.39E-02	3.71E-03	5.00E-02	2.44E-01	2.10E-01	7.53E-02	-2.08E-02
Freshwater eutrophication	kg P eq	2.87E-02	3.35E-02	1.09E-03	5.86E-06	2.09E-03	3.28E-02	3.24E-02	2.97E-04	4.27E-03
Marine eutrophication	kg N eq	2.34E-02	2.28E-02	3.05E-03	3.42E-04	4.66E-03	2.48E-02	2.02E-02	2.27E-03	5.30E-03
Freshwater ecotoxicity	CTUe	4.57E+02	3.30E+02	3.54E+01	1.81E-02	4.52E+01	5.33E+02	3.08E+02	4.69E+01	7.06E+01
Land use	kg C deficit	2.38E+01	8.36E+01	3.48E+00	0.00E+00	3.65E+00	3.09E+01	9.54E+01	2.40E+01	9.07E+00
Water resource depletion	m3 water eq	4.24E+01	2.58E+01	1.60E+00	1.77E-04	8.05E+00	6.27E+01	1.94E+01	4.69E+00	2.10E+01
Mineral, fossil & ren resource depletion	kg Sb eq	1.07E-02	1.05E-02	3.47E-04	1.60E-07	3.72E-04	4.17E-03	1.97E-03	4.94E-04	3.86E-04

SSP – Single superphosphate; TSP – Triple superphosphate

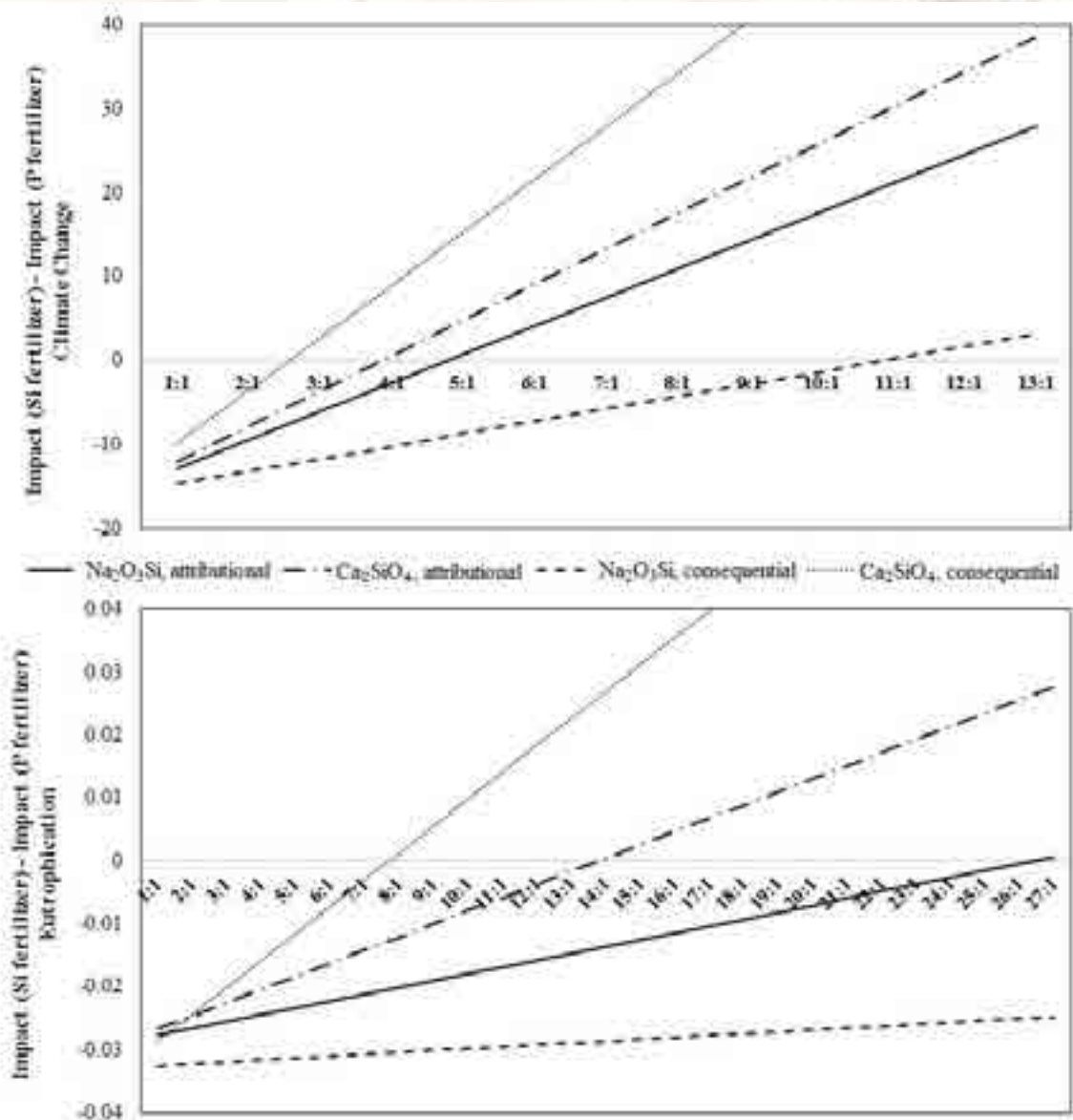


Figure 1: Difference in impacts, for the climate change (top) and freshwater eutrophication (below), between 1 kg of Si fertilizer and 1 kg of P fertilizer, as a function of the assumed substitution rate

Table 4: Number of kilometers of additional transportation required to change the ratio from (n+1):1 to n:1

Fertilizer	Transportation	Inventory	Number of km	
			Climate change	Freshwater eutrophication
Na ₂ O ₃ Si	Road	Attributional	6 974	24 103
		Consequential	3 295	7 825
	Sea	Attributional	67 582	205 406
		Consequential	30 591	47 191
Ca ₂ SiO ₄	Road	Attributional	6 151	32 894
		Consequential	9 902	79 620
	Sea	Attributional	59 607	280 328
		Consequential	91 947	480 174

4. Discussion

4.1 The impact of using Si-fertilizers

The results observed in Table 2 showed that generally the impact of using any Si-fertilizer is lower comparing with P-fertilizers. However, there are a number of issues that should be addressed before drawing any general conclusion. First, it is necessary to clarify that Si-fertilizers will not provide the soil with new P. Instead, the presence of mobile Si will adsorb to metal oxides, freeing immobile P and mobilizing it again for plant uptake. This means that Si-fertilization will have a bigger impact on soils where P-fertilizers have been extensively used in the past decades but where due to chemical properties (i.e. acid pH) the same P is largely unavailable (*Hinsinger 2001*). This also suggests that it is possible to use both fertilizers together, once all the immobile P has been mobilized. In that case, using the both fertilizers will decrease the necessity of using larger amounts of P fertilizers, since the mobilization is easier.

This analysis has also ignored the beneficial effects of Si-fertilization, especially in Si-accumulators such as some cereals and other vegetables. Recent studies report the increase of yield and/or quality in plants where Si-fertilizers have been added (*Guntzer et al. 2011, Rodrigues et al. 2003, Gu et al. 2012*). The most important effects relate to protection against pathogens and insects, reduction of water and salt stress and reduction of Al, Cd, Cu, Zn and As toxicity.

Another question that should be addressed is the absence of impacts for the application of both fertilizers. Inserting application into the analysis would, nevertheless, probably benefit Si fertilizers. However, at the moment, the novelty of this proposal means that there is no data available to build an inventory for Si application. Additionally, the contribution of Si for some impact categories in LCA, such as eutrophication, remains absent. Si abundance in fresh and coastal waters is known to decrease eutrophication effects, since in these conditions diatoms (i.e. microorganisms with Si-frustules) will grow in the place of toxic alga (*Cloern 2001, Verschuren et al. 2002*). Research is needed regarding the inclusion of the environmental effects of Si are introduced in LCIA. As such, it is possible that in our analysis the benefits of using the Si-fertilizers are underestimated. On the other hand it is also important to state that the most common P-fertilizers also contain some micronutrients, essential to the plants, that otherwise need to be provided. The need to supplement plants with these micronutrients together with Si fertilizers has been left out of the analysis, causing some minor underestimation of the impacts of the Si solution.

4.2 The replacement of P by Si in metal adsorption

To avoid skewing the analysis in favor of Si fertilizers, we took the conservative approach of ignoring the potential productivity boost Si has been observed to deliver for Si-accumulator crops. If we had considered this effect, then the Si-P substitution could be lower than 1:1 since, even if less P was re-solubilized, Si could increase productivity independently of its effect on the P cycle.

Studies that focus on the biogeochemical process of adsorption and competition between Si and P to metal oxides are needed. It is crucial to understand in which conditions the replacement of P is promoted by Si in order to add the Si-fertilizers together with a set of correct management practices that will help the objective. Currently it is known that P has more affinity to metal oxides than Si and therefore it is expectable to achieve a ratio of substitution higher than 1. However, our analysis has shown that even for higher substitution rates, Si-fertilizers can still be considered a solution.

5. Conclusions

The results of the LCA study analyzing the impact of replacing P-fertilizers with Si-fertilizers shows that generally the second one has a better environmental performance, even for substitution ratios of Si:P higher than 1, especially regarding calcium silicate fertilizer produced from lime, sand and water in categories such as human toxicity, freshwater ecotoxicity and water, mineral and fossil renewable resource.

Although this analysis has excluded both fertilizer applications, we expect that using Si-fertilizes has additional benefits such as the impact on eutrophication. Nevertheless, application of such

fertilizers would have a better impact on soils where P-immobile pools exist in large quantities in comparison to soils where P (mobile and immobile) content is low.

Finally, our analysis showed that even for high substitution ratios of Si:P the Si-fertilizers had lower impact compared to P-fertilizers. However, a new insight in the biogeochemical processes regarding Si and P competition are needed to establish the correct affinity of adsorption in the metal oxides surfaces.

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POSTER PAPERS

DAIRY PRODUCTION

2. Dairy Production

P05. Carbon footprint of milk from New Zealand dairy farm systems of varying intensification and the effects of use of different supplementary feed types

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Objective: To determine the carbon footprint (CF) or total greenhouse gas (GHG) emissions of milk from New Zealand (NZ) dairy farming systems varying in intensification level and the effect of type of brought-in feed used.

Methodology: Survey data were collected from >100 pasture-based dairy farms (for 2010/2011) in the Waikato region, a major dairying region of NZ. Farms varied in level of intensification associated largely with amount of brought-in feeds. They were grouped into low, medium or high categories according to level of brought-in feed (averaging 1, 2 and 4 t DM/ha/year, respectively). A life cycle assessment (LCA) approach was used to assess the CF of dairy farming systems per hectare and per kg fat- and protein-corrected milk (FPCM). Total GHG emissions (cradle-to-farm-gate) were estimated, based on International Dairy Federation guidelines. An LCA method was also used to determine the CF of thirteen different supplementary feed types. A scenario analysis was performed to investigate effects of different brought-in feed types on the CF of milk.

Results: The total GHG emissions per hectare from the dairy farm systems increased with intensification due to increased use of supplementary feeds (i.e. from low to high). However, there was little difference in the CF of milk between farm intensification levels, averaging 0.75 to 0.79 kg CO₂-eq/kg FPCM (Figure 1). Animal methane and excreta-related nitrous oxide emissions per kg FPCM generally decreased with increased intensification, coinciding with an increase in milk production per cow. However, this was countered by increased GHG emissions associated with the production of the increased amount of brought-in feeds. The CF of feeds varied widely (~0.02 to >0.5 kg CO₂-eq/kg dry matter) and was highest for palm kernel expeller from south-east Asia (based on economic allocation and including land use change). Sensitivity analysis showed a large effect of when different types of brought-in feed were used (Figure 2). When a low CF feed like Brewers grain was assumed to be used across all farm systems there was a decrease in CF of milk with intensification.

Implications: Use of brought-in feeds has been important for intensification of pasture-based dairy farms in NZ. The brought-in feeds have been used to increase milk production per-cow, which leads to increased milk production per-hectare. However, this needs to be associated with use of low-CF feeds in order to achieve an overall reduction in CF of milk.

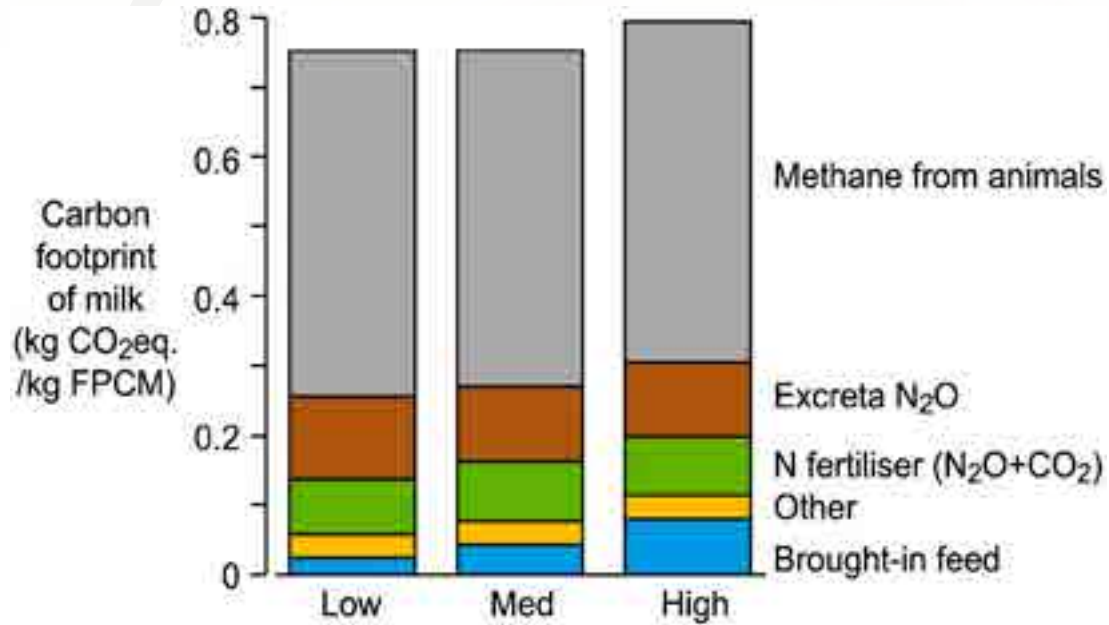
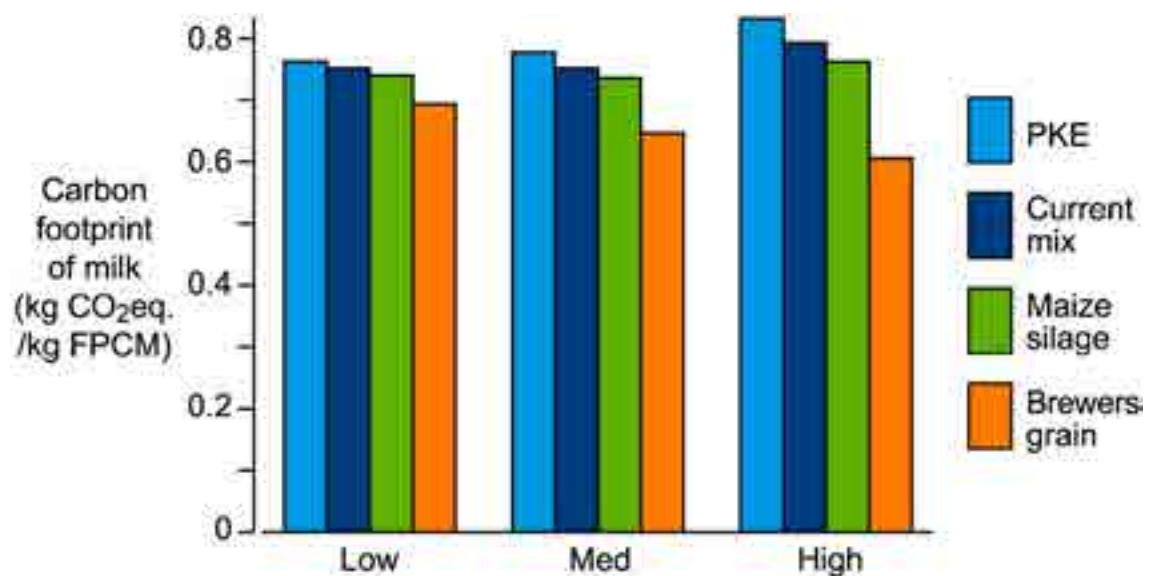


Figure 1: Carbon footprint of milk (cradle-to-farm gate) and main contributing sources for average pasture-based dairy farm systems in the Waikato region, New Zealand, that varied in amount of brought-in feed (1, 2 and 4 t dry matter/ha/year for low, medium and high, respectively). FPCM is fat- and protein-corrected milk.



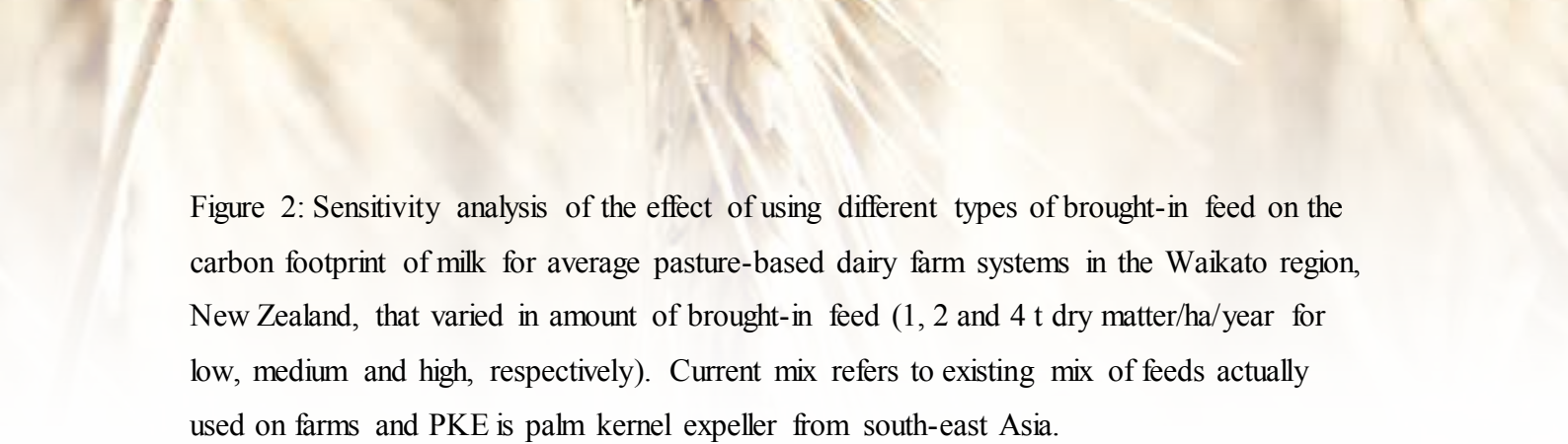



Figure 2: Sensitivity analysis of the effect of using different types of brought-in feed on the carbon footprint of milk for average pasture-based dairy farm systems in the Waikato region, New Zealand, that varied in amount of brought-in feed (1, 2 and 4 t dry matter/ha/year for low, medium and high, respectively). Current mix refers to existing mix of feeds actually used on farms and PKE is palm kernel expeller from south-east Asia.



P06. Combining Life Cycle Assessment and economic modelling to assess the environmental impacts of quota removal in the French dairy sector

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European dairy production faces great changes: dairy quota removal, increased farm size and increased world demand for dairy products are expected to affect the volume of raw milk produced, the share of the various dairy production systems in total production and the environmental impacts of dairy production. Following the Common Agricultural Policy Health Check in 2008, impacts of quota removal on prices and milk supply have been widely studied. However, impacts of this policy change on the milk production structure in Europe (i.e., on the share in production of the co-existing differentiated dairy production systems) and, in turn, on the environmental consequences of the European dairy production, have been less studied, and, to our knowledge, Consequential Life Cycle Assessment (CLCA) has never been used. LCA and economic models have been successfully combined to assess environmental impacts of public policies, mostly in the energy and biofuel sectors. This study combined LCA and economic modelling to assess the environmental impacts of quota removal in the French dairy sector using CLCA.

MATSIM-LUCA, a partial equilibrium model, was used to simulate the world and national agricultural market situations in 2030 under different scenarios allowing to assess the market and trade impacts of the European Union dairy quota removal (Table 1 and Figure 1). For each scenario, MATSIM-LUCA provides: i) the share in production of seven production systems representing the diversity of French dairy systems, ii) the total quantity produced, consumed, traded and the prices for milk and a range of agricultural goods, iii) the area of crops and grasslands, in 17 geographical zones and at the world level. These outputs were then used to feed a CLCA model, through scenario comparison (Figure 1).

Our main findings were that dairy quota removal: i) led to increased dairy production, by a larger dairy herd (data not shown); ii) did not significantly affect the share of French dairy systems in total production, even if they evolved differently (data not shown); iii) led to additional environmental impacts (Table 2). A contribution analysis (Figure 2) showed that for global warming including land-use change and impacts on ecosystems, additional impacts were dominated by emissions from land-use change. For the other impact categories

additional impacts were mostly due to additional bovine meat from both dairy and suckler production.

Table 1: Scenarios tested to assess effects of public policy. Timeframe: 2030. FR: France; E26: European Union minus France.

Scenario	Scenario description
S0	Dairy production quota in FR and E26
S1	No dairy production quota in FR and E26

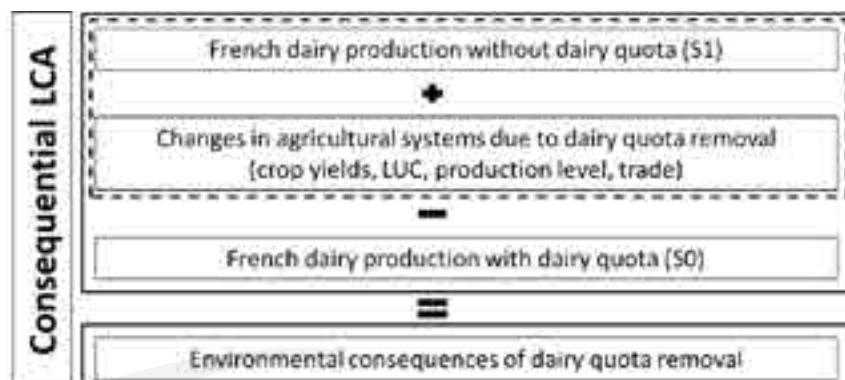


Figure 1: Conceptual framework used to perform CLCA of dairy sector, inspired by Vázquez-Rowe et al. (2013). LUC: land use change.

Table 2: Consequences of quota removal on impacts of French milk. Impacts were expressed per kg of Fat and Protein Corrected Milk. GWP: Global Warming Potential; GWP-LUC: GWP including land-use change; AC: Acidification; EU: Eutrophication; CED: Cumulative Energy Demand; EcoTox; Freshwater ecotoxicity; LC: Land Competition; EcoSys: Impact on ecosystems.

Impact categories	GWP-LUC	GWP	AC	EU	CED	EcoTo		EcoSys	
						x	LC		
Units	kg CO ₂ eq	kg CO ₂ eq	g SO ₂ eq	g PO ₄ ³⁻ eq	MJ	CTUe	m ² a	species.y	
S0 (Dairy quota)	0.855	0.854	8.6	4.4	2.86	1.16			
					9	0.315	4	3.87E-08	
					2.92	1.19			
S1 (No quota)	1.123	0.868	8.8	4.4	6	0.320	1	4.16E-08	
Additional impacts	per kg				0.05	0.02			
		0.268	0.014	0.2	0.1	8	0.005	6	2.87E-09
Variation	%	31.4	1.6	1.9	2.2	2	1.7	2.3	7.4

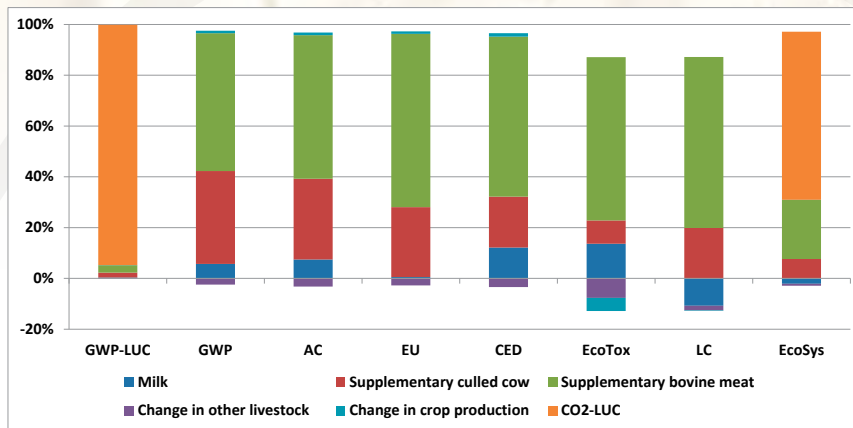


Figure 2: Contribution analysis of additional impacts of French milk due to quota removal. CO₂-LUC: Carbon dioxide emissions from land-use change.

P07. Synergies and trade-offs between local and global farm environmental performance of dairying: a case study of the Swiss mountain region

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Complying with the carrying capacity of the local and global ecosystem is a prerequisite to ensure sustainable development. In terms of environmental performance assessment at farm level, this implies separate implementation of local and global farm environmental performance indicators (Repar et al., 2016). In order to improve the environmental sustainability of farming, a better understanding of the link between these two dimensions of farm environmental performance is necessary. The aim of our work is to investigate the possible synergies and trade-offs between the local and global environmental performance of dairying in the Swiss alpine region.

Our analysis relies on a sample of 56 dairy farms in the Swiss alpine region for which detailed and comprehensive cradle-to-farm gate LCAs have been estimated using the SALCA (Swiss Agricultural Life Cycle Assessment) approach and the quantified environmental impacts decomposed into their on- and off-farm parts. We define global environmental performance as the on- and off-farm environmental impacts generated in the cradle-to-farm gate link per MJ digestible energy for humans. We assess local environmental performance using the indicator on-farm environmental impact generation per unit usable agricultural area.

The results of the Spearman's rank correlation analysis between the global and local environmental performance indicators show a quite complex picture (see Table 1). Depending on the environmental impact category considered, both synergies and trade-offs can be observed, nevertheless trade-offs clearly predominate.

Our findings imply that the improvement of the environmental sustainability of dairy farming is a highly complex endeavour, for which no one size fits all solutions may exist. To avoid that any improvement in one dimension of environmental performance happens at the expense of the other, both local and global dimensions have to be accounted for. Our results furthermore imply that existing agri-environmental policy measures that exclusively focus on the local dimension of environmental performance may lead to a deterioration of global environmental performance.

POSTER PAPERS

CROPS AND FRUITS

3. Crops and Fruits

P08. Conservation *versus* conventional agriculture: a case study in Wallonia (Belgium)

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Farming systems implementing conservation agriculture have been developing on the American continent for almost a century and their use has been increasing under European latitudes for about 20 years. Tillage in particular and conventional farming in general are frequently criticized for decreasing soil fertility and biodiversity. In this framework, more and more farmers are turning to conservation agriculture. Based on three main principles (no-till farming, maximum and permanent soil cover, and long and diversified rotations), conservation agriculture, also called ecologically intensive agriculture, aims at maintaining high crop yields while conserving or even improving soil quality and biodiversity.

The increasing development of ecologically intensive agriculture brings up many questions from farmers and scientists regarding its environmental impacts and assumed benefits. In Wallonia (Belgium), most field observations and studies carried on so far are based on empirical methods and not on thorough scientific methodologies.

Using Life Cycle Assessment (LCA) combining environmental and socio-economic LCA, this work aims at answering the following question: What are the impacts of conservation agriculture compared to conventional agriculture in the Walloon situation? The study focuses on entire crop rotations in order to include all farming system parameters. Analysed data are based on environmental (*e.g.* chemical and biological indicators of soil quality) and socio-economic (*e.g.* costs, working hours) data collected from Walloon farms on a time span of at least three years. These farms have been carefully selected according to criteria presented in this work in order to ensure results comparability. This systematic study also aims at identifying the most impacting cropping steps and practices in order to inform farmers, stakeholders and decision-makers on practices best suited to mitigate environmental and socio-economic pressures from cropping systems.

P09. Potato production in Wallonia (Belgium): impacts of various cropping and storage systems

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As the fourth most important crop in Wallonia (Belgium), potatoes were cropped on 34,634 ha of the region in 2015, corresponding to 8.6% of the arable lands.

In Wallonia, ware potatoes are the most commonly produced potatoes (97.22% of the surfaces). These potatoes can be stored from a few weeks to several months, according to the variety and storage facility (ventilation, refrigeration, in bulk, etc.).

Potato is a high demanding crop with respect to plant protection products (PPP). In particular, fungicides are largely used to protect the crop against the late blight caused by the pathogen *Phytophthora infestans* (Mont.) de Bary.

As the most cultivated variety by far, the “Bintje” potato offers several advantages: high yields, cheap young plants, high quality and multiple outlets (processing, export, fresh market, etc.). However this variety is very sensitive to the late blight and requires large amounts of energy for its storage. Farmers are more and more aware of these concerns.

Moreover, various stakeholders from the fresh market are calling for other varieties in order to distinguish themselves from the competition. Alternative varieties, such as the high yielding and less sensitive “Fontane”, are therefore gaining ground. Furthermore, organic potato production is currently slowly but steadily increasing in Wallonia, with farmers producing potatoes for the fresh market, the processing industry or for direct sale on the farm. Considering the high diversity among potato cropping systems in Wallonia, this work aims at evaluating the impacts of the potato production and storage using Life Cycle Assessment (LCA) combining environmental and socio-economic LCA. The objectives of this study are therefore multiple: (1) conduct an LCA for the major or promising potato cropping systems in Wallonia (Bintje, Fontane, organic potato), considering various storage options according to the targeted value chain; (2) identify the most impacting steps in the chain production-storage; (3) propose actions to mitigate the impacts of this highly demanding crop in energy and PPP.

P10. Assessing the greenhouse gas emissions mitigation potential for cereal-based cropping sequences through changes to management of synthetic and biologically-fixed N inputs

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LCA was used to compare pre-farm plus on-farm greenhouse gas (GHG) emissions profiles of four cropping sequences on a Vertosol in the sub-tropical grains region of Australia. We examined the mitigation effects of split fertiliser applications, use of ENTEC®; and substitution of synthetic with legume-fixed N.

Foreground data were obtained from direct field measurement using automated chambers (Schwenke et al. 2014; 2015; 2016) and complimented with internationally recognised background data (Weidema et al. 2013; Life Cycle Strategies 2013). Analysis was conducted in SimaPro with formulae and approaches taken from the IPCC reports and the Australian National Inventory Report (NIR, 2015). System boundaries, inputs and impact assessment for GHG emissions followed Brock et al. (2012; 2016). The sequences were: canola with 80 kg/ha fertiliser N (80N)–wheat 85N–barley 65N, chickpea 0N–wheat 85N–barley 5N, chickpea 0N–wheat 5N–chickpea 5N and chickpea 0N–sorghum 45N. Yields were wheat: 3.1t/ha, barley: 3.8t/ha, canola: 1.8t/ha, chickpea: 1.6t/ha and sorghum: 7.1t/ha, with results presented per ha due to crop type variation.

The 3-year direct N₂O emissions ranged from 0.489 kg N₂O-N/ha for the third sequence to 1.336 kg N₂O-N/ha for the first sequence (Fig.1). The range for CH₄ fluxes was smaller (i.e. -1.13 to -1.02 kg CH₄-C/ha). Total pre-farm plus on-farm GHG emissions for the 3-year sequences ranged from 919 kg CO₂-e/ha to 1893 kg CO₂-e/ha (Fig. 2). Soil N₂O accounted for 24–44%, and fertiliser production accounted for 20% to 30% of total emissions. The on-farm measured 53% and 85% reductions in N₂O emissions from the 50:50 split fertiliser N application and ENTEC®, for the first sequence, resulted in total emissions reductions 18 and 29%, respectively (Fig. 3).

Potential is evident to achieve emissions reduction through these strategies. However, this analysis was based on an assumption that soil C levels had reached a steady state, as per the NIR (2015). Preliminary analysis, building on this study, supports an hypothesis that once the C cost of N₂ fixation is considered, crops such as N-fertilised canola, with a low harvest index may be preferable to legumes from a total GHG perspective (Brock et al. 2016). This

issue and an identified discrepancy with the default emission factor for crop residues are currently being pursued for future versions of the NIR (2015).

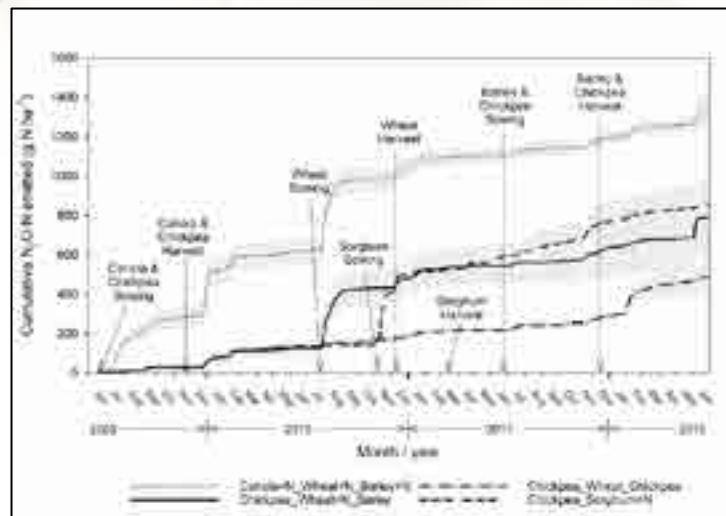


Figure 1. Nitrous oxide (N₂O) emissions, measured using automated chambers, for the four 3-year cropping sequences at Tamworth in the sub-tropical grains region of eastern Australia. Shading around lines indicates ± standard error of mean

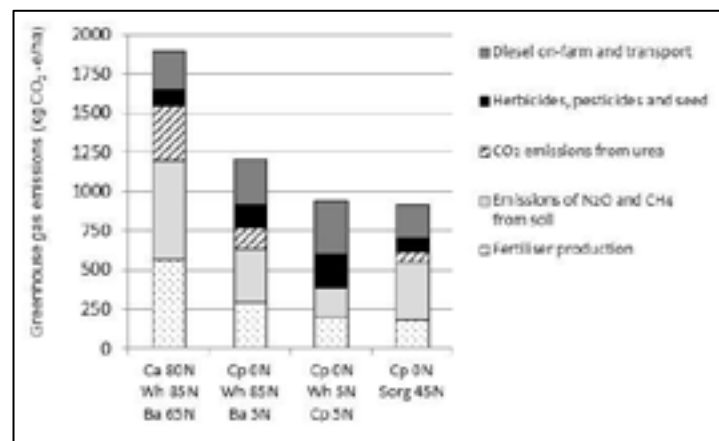
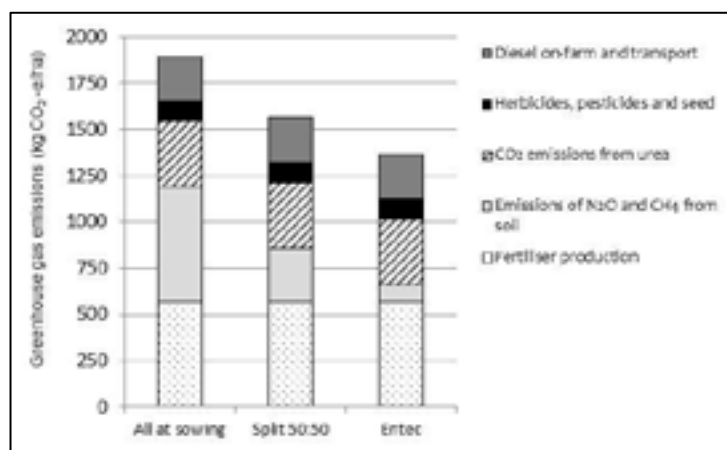


Figure 2. Cradle-to-farm gate greenhouse gas emissions (kg CO₂-e/ha) for the four 3-year cropping sequences at Tamworth in the sub-tropical grains region of eastern Australia; Ca – canola, Wh = wheat, Ba = barley, Cp = chickpea and Sorg = sorghum



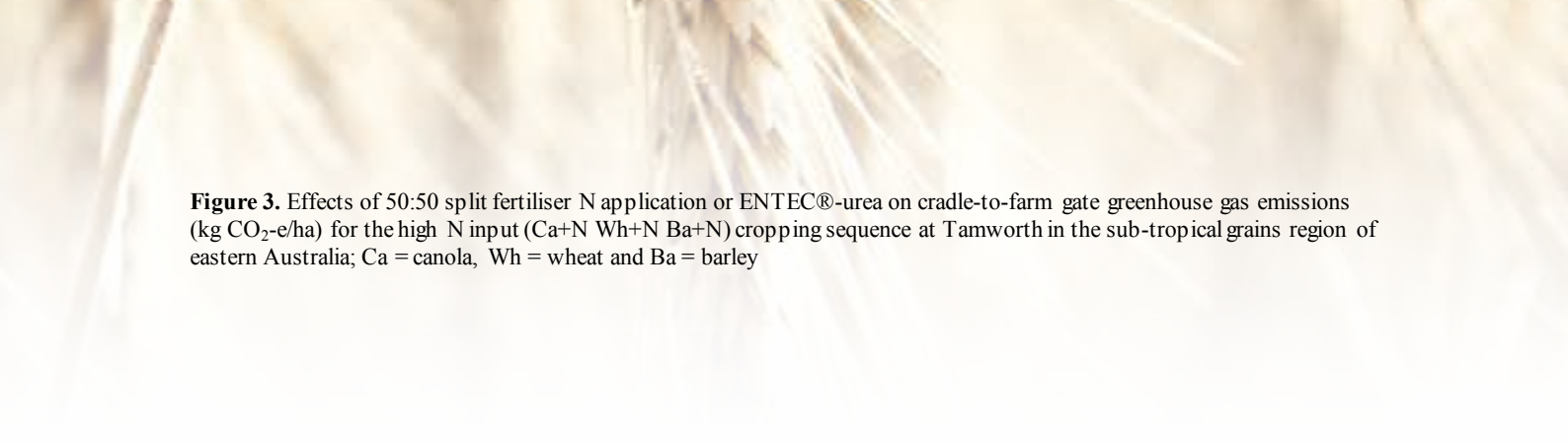



Figure 3. Effects of 50:50 split fertiliser N application or ENTEC®-urea on cradle-to-farm gate greenhouse gas emissions (kg CO₂-e/ha) for the high N input (Ca+N Wh+N Ba+N) cropping sequence at Tamworth in the sub-tropical grains region of eastern Australia; Ca = canola, Wh = wheat and Ba = barley



P11. An approach for a Social Life Cycle Assessment of the citrus production in the region of Valencia (Spain)

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This study is a first attempt to assess the social and socio-economic performance of the citrus production in *Comunidad Valenciana* (CV), the leading producing region in Spain, defined by small farms and family agriculture. A Social Life Cycle Assessment (SLCA) is carried out by following UNEP/SETAC (2009), initially focused on the agricultural stage alone, the function of which is to produce high-quality citrus for fresh consumption.

Impact subcategories were defined at the stakeholder level (UNEP/SETAC, 2013), considering “landowners” as an independent group, subject to different social conditions.

The inventory analysis was based on survey data collected from 280 farms (both ecological and conventional) disseminated across 105 municipalities of CV for the season 2012/2013, together with official reports and statistics, and personal communication with a labor union specialized in the sector. A performance reference point (PRP) approach was proposed, by comparing indicators at farm and sector levels with country-based indicators, used as a threshold of minimal compliance. A scoring system was then applied to measure if the social performance of the farms and sector is better or worse relative to average levels in Spain and minimal legal requirements, in quantitative terms. Table 1 is the outcome from this characterization procedure.

Pickers, who are mostly immigrant men, hired externally by the company responsible for the postharvest treatment, represented the group of workers with higher social risks. They were subject to the greatest impacts (“very high”) in terms of *fair salary*, *hours of work*, *equal opportunities/discrimination*, and *health and safety*. Nevertheless, the average score for each subcategory showed that the highest social impact (“very high”) corresponded to *landowners* in terms of *dependence/reliance on cooperative organization* and *job satisfaction*. Further harmonization in characterization methods is needed to allow for the social performance of different systems to be compared, while upstream processes should be equally included for a hotspots analysis from a life cycle perspective.

Table 1. SLCA results for the citrus production in CV under the PRP approach proposed.

Stakeholder category	Subcategory	Impact score	Baseline data source
Workers	Freedom of association and collective bargaining		
	Share of unionized workers	0	Ministerio de Empleo (2015), 2010 data
	Freedom of association	2	BOE (2001); Union CCOO (personal communication)
	Fair salary		
	Workers earn less than 1,000 euros per month	3	Ministerio de Empleo (2015), 2010 data
	Monthly salary of hired permanent workers	0	BOPCS n°60 (2015); Eurostat (2015)
	Monthly salary of hired temporary workers	0	BOPCS n°60 (2015); Eurostat (2015)
	Salary of citrus pickers hired through cooperatives	0	DOCV n°7533 (2015); Eurostat (2015)
	Salary of citrus pickers hired through temporary agencies	3	DOCV n°7533 (2015); Union CCOO (personal communication)
	Presence of suspicious deductions in wages of pickers when hired through temporary agencies	4	Union CCOO (personal communication)
	Hours of work		
	Number of hours worked per week by hired permanent workers	2	BOPCS n°60 (2015); BOV n°223 (2015); INE (2016)
	Number of hours worked per week by hired temporary workers	1	BOPCS n°60 (2015); BOV n°223 (2015); INE (2016)
	Number of hours worked per week by pickers	4	DOCV n°7533 (2015); INE (2016)
	Unpaid overtime	2	INE (2016a), average 2012-2015
	Forced labor		
	Risk of forced labor among hired (permanent and temporary) workers	0	Union CCOO (personal communication)
	Risk of forced labor among pickers	2	Union CCOO (personal communication)
	Equal opportunities/ Discrimination		
	Job satisfaction of women to men agricultural workers	2	Ministerio de Empleo (2015), 2010 data
	Discrimination towards women hired workers	3	ILOstat (2016); INE (2016a), average 2010-2013
	Discrimination towards women pickers	4	ILOstat (2016); Union CCOO (personal communication)
	Irregular migrant workers among the citrus pickers	4	Union CCOO (personal communication)
	Health and safety		
	Risk for accidents at work in agriculture	1	Ministerio de Empleo (2015), average 2010-2014
	Risk for fatal accidents at work in agriculture	1	Ministerio de Empleo (2015), average 2010-2015
	Farming equipment (machinery and tools) conform the standards	0	BOV n°223 (2015); Union CCOO (personal communication)
	Adequate general occupational safety measures are taken	0	Union CCOO (personal communication)
	Favorable conditions for pickers to find (livable) accommodation	4	Union CCOO (personal communication)
	Social security		
Hired workers pay social security contributions according to the working hours	0	Union CCOO (personal communication)	
Pickers pay social security contributions according to the working hours	3	Union CCOO (personal communication)	
Job satisfaction			
Own perception of job satisfaction of agricultural workers	2	Ministerio de Empleo (2015), 2010 data	
Landowners	Hours of work		
	Number of hours worked per week by landowners (self-employed agricultural workers)	2	INE (2016)
	Equal opportunities/ Discrimination		
	Share of women landowners	1	INE (2016), average 2012-2014; surveys
	Share of agricultural workers with poor educational attainment	4	Ministerio de Empleo (2015), 2010 data; surveys
	Self-sufficiency		
	Share of landowners with a complementary job (conventional agriculture)	3	INE (2016), average 2012-2014; surveys
	Share of landowners with a complementary job (organic agriculture)	2	INE (2016), average 2012-2014; surveys
	Dependence/reliance on cooperative organization		
	Share of landowners who belong to an association/cooperative	4	INE (2016), average 2012-2014; surveys
Job satisfaction			
Confidence that the sons/daughters will continue with the family business	4	Union CCOO (personal communication); surveys	
Local community	Delocalization and migration		
	The sector contributes to delocalization, migration or "involuntary resettlement"	3	Union CCOO (personal communication)
	Absence of organizational procedures for integrating migrant workers into the community	3	Union CCOO (personal communication)
Consumers	Health and safety		
Risk of pesticide residues in citrus fruits	0	IVIA (2005)	

P12. Life Cycle Assessment of a Wheat Cultivation Process: Effects of Allocation Criteria

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1. Objective of the work

The purpose of this work is to assess the environmental impacts of an integrated wheat cultivation process located in the Italian province of Modena. In particular, besides a proper systematic description of wheat production, different ways of allocating environmental burden between wheat and its by-product were examined in order to deeply explore unequal issues of our planet, as the right to food and the environmental pollution.

2. Materials and methods

To assess the environmental impact, the analysis was conducted using the SimaPro 8.0.4 software and IMPACT 2002+ evaluation method. The functional unit is the wheat production during the three-year rotation of crops, like wheat, beet and sorghum that means a total production of 6090 kg of wheat per hectare. Data related to cultivation, management and disposal were directly collected from the producer. Whenever possible, emissions to air, water and soil from pesticide considered in the present study were calculated using the Mackay model. In this study, a multioutput model was applied considering straw as a coproduct and allocating 1/3 of the shared operations between the different crops. Additionally to existing allocation approaches, such as mass allocation, energy allocation and economic allocation, different allocation criteria, based on carbon content and nutritional value, were applied.

3. Results

The results show a total damage of 3.226Pt caused by 64.5% to Ecosystem quality, 21.54% to Human health, 7.07% to Climate change and 6.89% to Resources. In particular the main environmental burdens are primarily due to direct emissions of fertilizers and pesticides and to land use. The Pirimicarb, a selective carbamate insecticide used to control aphids on wheat cultivation, produces a damage of $2,8857E-3Pt$ in Aquatic ecotoxicity and $1.6147E-3Pt$ in Soil ecotoxicity. The mass allocation provides the minimum damage of 2.307 Pt with respect to the highest environmental impact obtained with the economic criterion.

Damage categories	Pt	Substances
Human Health	21.54	Particulates <2.5mm in air and Cadmium to soil
Ecosystem quality	64.5	Transformation, to arable, non-irrigated
Climate change	7.07	Dinitrogen monoxide
Resources	6.89	Oil, crude

Table 1. Evaluation of the wheat cultivation life cycle

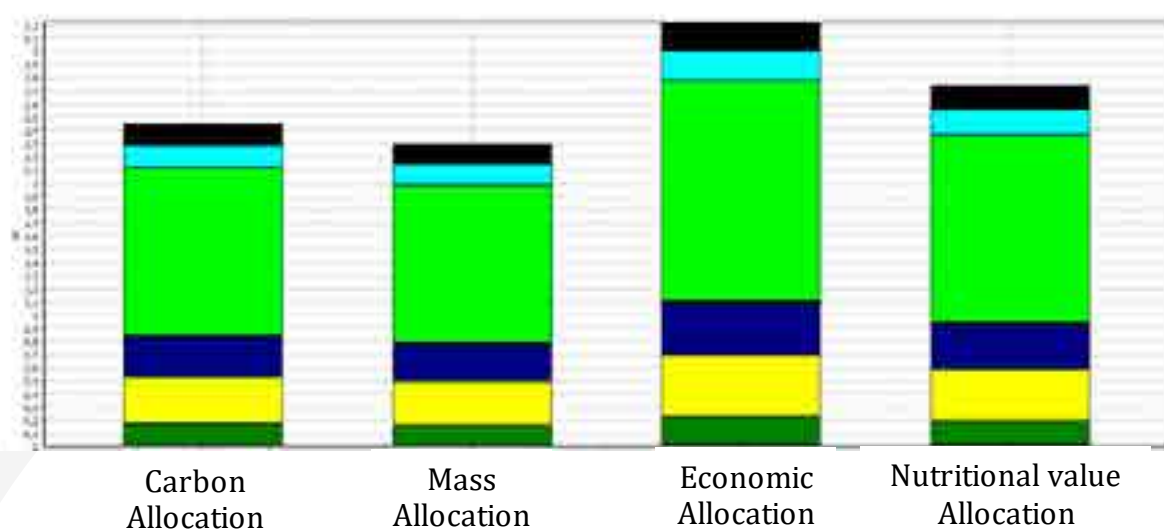


Figure 1. Effect of allocation criteria on the environmental impacts of the wheat cultivation life cycle

P13. A comparative systems description of maize production in Southern and Eastern Africa

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Life-cycle based knowledge of grain production and usage is needed to weave together issues such as agricultural GHG emissions, wise use of scarce agricultural lands for food, materials and energy production, and healthy nutrition. In Southern and Eastern Africa, however, maize is associated with few of these systemic sustainability issues, but instead remains central to an understanding of poverty and human security, as it is a staple food for hundreds of millions in the region. To date, there have been isolated attempts to use the life cycle perspective to study maize in South Africa.

This paper aims to explore maize production systems in Zimbabwe, Zambia and Kenya, and compare these, from a life cycle perspective, to each other, and also to South African maize production. Life cycle thinking was used to model the system from maize cultivation, via harvesting and post-harvest processing to storage. The SimaPro (version 8.05) farm-to-silo model was used to explore variations in a range of environmental impacts relative to the key performance parameter, yield per hectare, and key input parameters, viz. fertiliser input and irrigation. Variations in these parameters were obtained through a search of FAO datasets and the open literature on maize production in the countries of interest. The impact categories considered include greenhouse gas (GHG) emissions, acidification, eutrophication and ecotoxicity. The results below have been scaled so that the highest impact in each impact category equals 100 %. The global warming potential (GWP) of maize grain varies by country from 0.6 to 0.8 kg CO₂-eq/kg.

Relative environmental impacts of the maize grains

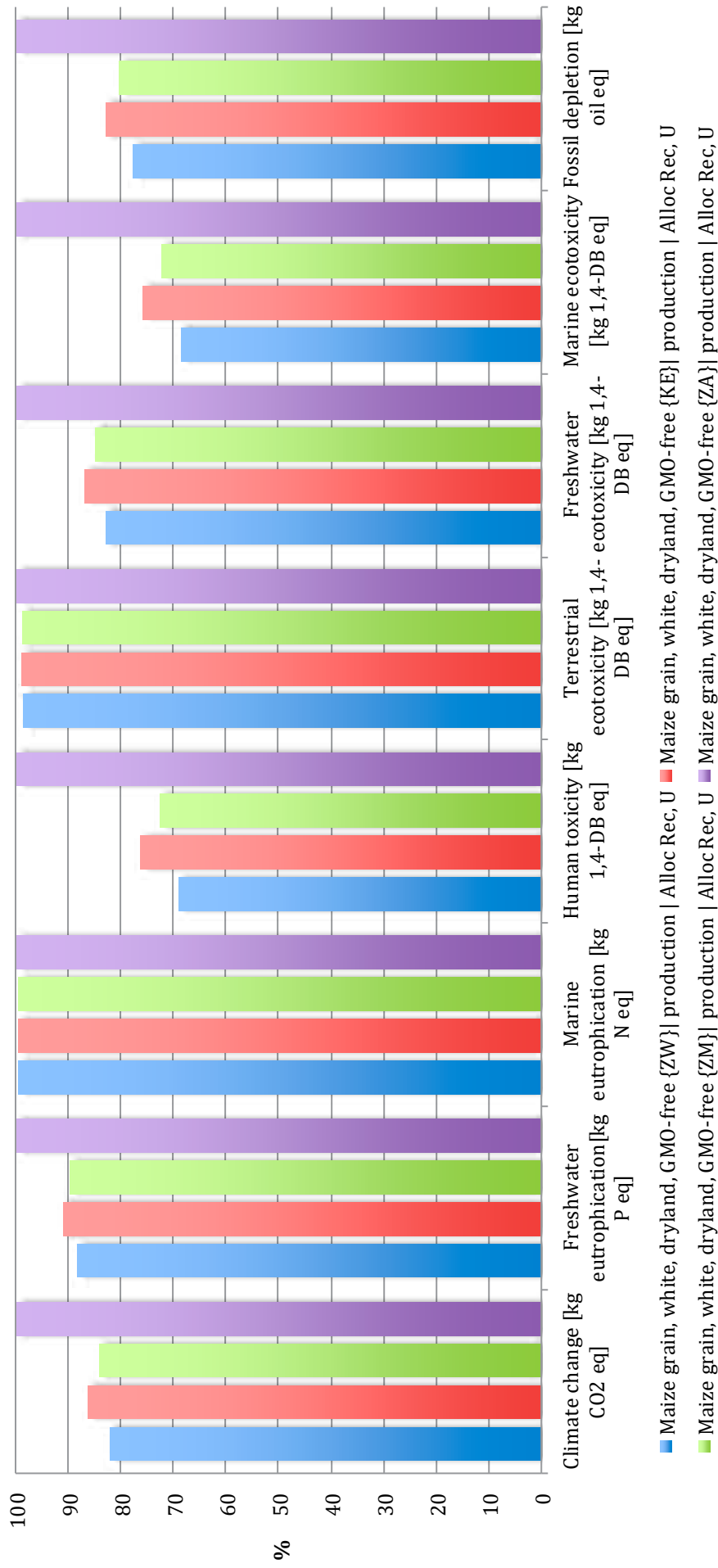


Figure 1: Relative environmental impacts of 1kg maize grain in Zimbabwe, Kenya, Zambia and South Africa

P14. Comparative analysis of regionalized inventories: Life Cycle Assessment of Portuguese maize, wheat, barley and oat

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ABSTRACT

Objective: Regionalization is one of the current priorities in Life Cycle Assessment (LCA). To enable accurate regional-level studies, it is essential to improve Life Cycle Inventories (LCI). Increasingly new methods and initiatives for regionalized LCI are being proposed, as for example the Agribalyse project in France or the guidelines by the World Food LCA Database. Consequently, food databases are progressively moving towards including more regional and country-specific records. So far, however, there is an absence of studies that test regionalized inventories to check for inclusiveness and representability. In this paper we used Portugal as an example and compared the inventory interventions included in the Agri-Footprint database v2 for maize, wheat, barley and oat with scale-consistent regional statistics from Portugal.

Methods: We considered only life cycle stages occurring at the farm, i.e. a cradle-to-gate approach. The functional unit used was 1 kg of product. The Portuguese LCI interventions adapted were obtained in Morais et al (2016a), who adapted some inventory interventions for agri-food products in Portugal consistently with Agribalyse rules. Main adaptations are in fertilizers and pesticides applied, crop yield, land use and transformation. For impact assessment (LCIA), we used the impact category Global Warming Potential (GWP), measured in kg CO₂ eq. We used SimaPro v8.1 to perform calculations and analyses.

Results: Results show that fertilization was the key contributor to differences between results obtained using national statistics and Agri-Footprint. These differences in fertilizer use explain most of the differences between the records. Land use and respective farming activities in the LCI are also particularly relevant to explain the differences among the interventions.

Conclusions: The comparison of results from using a secondary database (Agri-footprint) with the same inventory adapted using highly specific, regionalized data for Portuguese products, yielded significant differences that illustrate the need for a better understanding of when secondary data can be used to represent the impacts of each product.

Keywords: Life Cycle Inventory; Regionalization; Agri-food; Inventories comparison

1. Introduction

Life cycle assessment (LCA) is a method used to measure the performance of a product or service in every stage of its life cycle (Hellweg and Milà i Canals, 2014). Life cycle inventory (LCI) is the stage where most time and effort is dispended, as it is the phase where data is compiled to characterize the system. The LCI is often separated in three components: background data (processes deep within the supply chain), foreground processes (first tier of processes required in production) and activity data (process data measured in situ for the target of the study).

Regionalization in agri-food LCA studies is a relevant issue, due to the fact that the variance of results for agri-food products is relatively high (Haas et al., 2000; Teixeira, 2015). Nevertheless, standard databases may be insufficient to grasp accurate regional data (Reap et al., 2008), as location is a critical aspect in agriculture stages (Roy et al., 2009). Thus, local data in inventories is required to capture the regional features of the agricultural operations and the processes it depicts. Nevertheless, LCI regionalization should be led using a coherent and consistent approach, rather than study-specific updates, resorting to international frameworks to ensure comparability with international LCA studies (Yang, 2016). At country scale, Agribalyse (ADEME, 2013) was a pioneer project conducted to produce a regionalized agricultural inventory, mandated by the French government. The World Food LCA Database (WFLDB) (Nemecek et al., 2014) is a project that aims to produce directives that support the establishment of regional databases for global agri-food products, developed by Quantis and the Swiss Institute for Research in Agriculture, Nutrition and the Environment, Agroscope. Recently, Morais et al. (2016b) assessed progress in Portugal towards regionalization in agri-food sector and Morais et al. (2016a) started to produce a national, consistent inventory for Portugal. The Blonk Agri Footprint BV also produce a new version (v2) of Agri-Footprint database (Blonk Agri-footprint BV, 2015), covering Portugal. These attempts highlight the importance of regionalization as the next step in the evolution of accurate inventories.

This study aims to compare results using an LCI adaptation drawn from Morais et al. (2016a) and the Agri-Footprint database v2, for the all four Portuguese products included in Agri-Footprint (Blonk Agri-footprint BV, 2015). We assessed how the two approaches affect results for the agricultural systems mentioned.

2. Methods

The objective of this study was to compare the inventory flows obtained using the method laid out by Morais et al. (2016a) and the flows in the Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015). We performed the comparison for all Portuguese products present in Agri-Footprint (Blonk Agri-footprint BV, 2015), which are: maize, wheat, barley and oat. We opted for the complete list of products in order to enable a thorough comparison between these two inventories for one country (Portugal).

Regarding the approach of Morais et al. (2016a), the LCI interventions adapted are fertilizers and pesticides applied, crop yield, land use and transformation, soil loss and greenhouse gases (GHG) emissions (including carbon dioxide after urea or lime applications), i.e. it is not a complete inventory. These interventions are completed with background data from ecoinvent v3 (Weidema et al., 2013). The main intervention adapted by Morais et al. (2016a) was the type and amount of fertilizers applied by crop. Fertilization, in agri-food products, is particularly important as the main source of direct and/or indirect GHG emissions. Fertilizers application was adapted to Portugal using regional data, at Agrarian region level, using official data from Gabinete de Planeamento, Políticas e Administração Geral (GPP), Portuguese Agriculture Ministry. Next, these regional fact sheets (GPP, 2001) for each crop were corrected using the total national consumption obtained from the Portuguese statistical office, INE (INE, 2015). This process guarantees scale consistency, in the sense that the sum of all fertilizer consumption from all crops produced in all regions is equal to the amount of fertilizers used in the country.

Table 1 presents the correspondence between product processes from Morais et al. (2016a) and Agri-Footprint database v2. For Agri-Footprint database v2 products we opted to use mass allocation (rather than economic or energy allocation). Oat is missing from ecoinvent v3 (which is necessary as background data for processes in Morais et al., 2016a), and thus the comparison between inventories was realized only for direct emissions associated with fertilization and crop residues.

Table 1: SimaPro processes used in this study from ecoinvent v3 adapted with Morais et al. (2016a) and Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015)

Morais et al. (2016a)	Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015)
Maize grain {PT} production Alloc Def, U	Maize, at farm/PT Mass
Barley grain {PT} barley production Alloc Def, U	Barley grain, at farm/PT Mass
Wheat grain {PT} wheat production Alloc Def, U	Wheat grain, at farm/PT Mass
Oat (does not have ecoinvent process)	Oat grain, at farm/PT Mass

Regarding system boundaries, we used a cradle-to-gate approach, adapted to the maximum level of detailed allowed by the Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015), e.g. indirect machinery emissions related to product transport inside the farm. Figure 1 presents the stages considered (only stages occurring in the farm). To simplify the comparison between inventories we used a mass basis functional unit (FU), i.e. 1 kg of product.

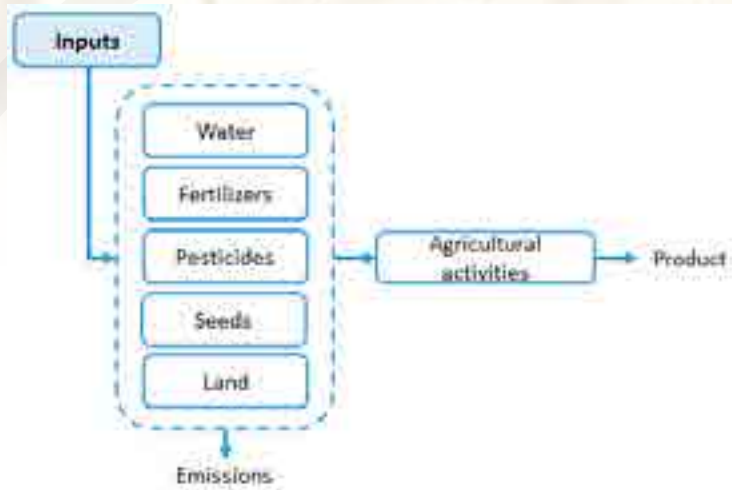


Figure 1: System boundaries considered in the study

Regarding Life cycle impact assessment (LCIA), we used only Global Warming Potential as an impact category due the fact that interventions covered by Morais et al. (2016a) have an implication mostly on GHG emissions. We used GWP potentials from Intergovernmental Panel on Climate Change (IPCC), with a time horizon of 100 years, and recommended also by the European Union (EC - JRC, 2011). We used SimaPro v8.1 to perform calculations and the analysis.

3. Results

Figure 2 shows results for the GWP impact category for the four products assessed. The main contribution to GWP is fertilization. GHG emissions from machinery used in agricultural activities are also relevant to explain the differences in results, due to the fact that these activities depend on land use occupation (expressed in SimaPro as “Occupation”) and respective activities areas. This means that higher land use area typically leads to higher GHG emissions per functional unit.

Results also show that wheat is the only product where the method put forth by Morais et al. (2016a) led to higher GHG emissions. This fact is justified by the assumed fertilizer quantity applied, but also due to high land use. In fact, wheat is the only product where fertilizer quantity applied and land use (i.e. “Occupation” process) according to Morais et al. (2016a) is higher than what is obtained when using the Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015).

However, the differences between inventories are relatively low, approximately 0.31 kg CO₂eq. The highest difference is found in wheat, about 0.61 kg CO₂eq, and the lowest difference is in maize, about 0.03 kg CO₂eq.

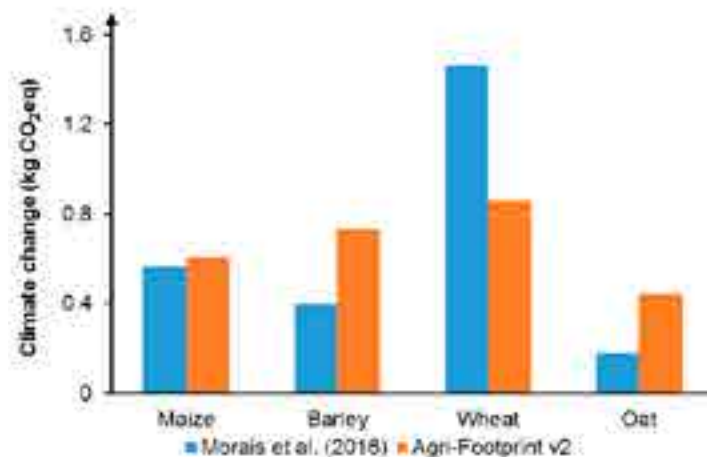


Figure 2: Difference between in the climate change impact category, according to the ILCD method.

4. Discussion

When conducting an LCA study, LCI is the stage where most effort is required. Secondary databases reduce this effort, but they should be accurately regionalized if they are intended to depict the processes accurately. In agri-food studies regionalization is particularly relevant due to the locally specific processes and activities involved in the agricultural stage of food products. However, this cannot come at the expense of comparability between studies, which is a risk if inventories and models are built independently for very specific situations. There is thus a pressing need for regionalization methods that are built consistently with international frameworks, as the WFLDB (Nemecek et al., 2014). ADEME (2013) and Morais et al. (2016a) are two cases of international frameworks application in a concrete cases, France and Portugal, respectively.

The comparisons between inventories performed in Morais et al. (2016a) between their interventions and ecoinvent v 3 (Weidema et al., 2013), and in this study between Morais et al. (2016a) and Agri-Footprint v2 (Blonk Agri-footprint BV, 2015) reveal significant differences in the outcomes of impact assessment models depending on the method used to draw regionalized inventories.

System boundaries considered in this study is the same that Agri-Footprint v2 (Blonk Agri-footprint BV, 2015). Therefore, we disregarded the emissions from transportation inside the farm and after the farm. These simplifications do not influence results, since the study area was constricted and is assumed to be the same for all products (all are representative of the same geographic area).

Besides different data sources and the scale adaptation carried out by one of the references of this study (Morais et al., 2016a), an additional important difference observed in the comparison presented in this study was the model used for GHG emissions. Both inventories use the same underlying method to calculate GHG emissions, i.e. IPCC (2006). However, Morais et al. (2016a) used it indirectly by resorting to the Portuguese National Inventory Report (NIR) (APA, 2014). The NIR, for some emission factors (e.g. N₂O emission factor due synthetic fertilizers application) suggests a different reference value compared to the IPCC (2006). Since Agri-Footprint v2 (Blonk Agri-footprint BV, 2015) used the reference value of IPCC (2006), this aspect can be relevance in GHG emissions and the GWP impact category.

5. Conclusions

LCI is the life cycle stage where more efforts are dispended in LCA studies. Regionalized inventories, which depict locally specific processes, facilitate the work of LCA practitioners, and are necessary to ensure inter-study comparability and accuracy. In this study we performed a comparison between a generic inventory (Agri-footprint v2) with other inventory adapted using highly specific and regionalized data for Portuguese products (Morais et al., 2016a), in agri-food sector. We concluded that, even in the GWP impact category, which is highly standardized, there are significant differences if the inventories are built according to different rules.

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P15. Greenhouse Gas Emissions of Sweet Cherry Production: Identification of its Key Factors in the Farm Stage

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Objective Nowadays, several organizations and governments have created a demand for information about the carbon footprint (CF) of agricultural products. Today, the export oriented fruit sector in Chile is being challenged to quantify and reduce their CF. In this sector, Chile is the largest southern hemisphere producer and exporter of sweet cherry fruit. There are scant peer-review studies that examine the CF of sweet cherry production. Within this context, the main objectives of this study are to evaluate the CF of conventional sweet cherry production under Chilean representative practices and to identify key factors that contribute significantly to the GHG emissions in the farm stage.

Methods This study follows the ISO/TS 14067 (1) framework and the main recommendations for CF of horticulture systems of PAS 2050-1 guide. Figure 1 shows the system under study. The data from the agricultural inputs are based on technical studies published by Chilean governmental agencies. The previous data are complemented by field information obtained from sweet cherry orchards. These data give a set of values without the range of variation. To estimate the level of uncertainty, the approach of the GHG Protocol standard (2) is employed. We include a pedigree matrix and the Taylor series expansion for to determine the overall system uncertainty. The CF calculations are conducted with the CCaL V3.0 software. Additionally, the Ecoinvent 2.2 database is used.

Results The average CF of the Chilean sweet cherry production is 0.41 kg CO₂-e/kg of harvest fruit, with a 95% confidence interval between 0.36 and 0.47 kg CO₂-e/kg. The diesel for field operations and fertilizers are the most important contributors to the CF causing, on average, 41 and 32% of total emissions, respectively (see Table 1). Under average European conditions, Audsley et al. (3) indicated a CF for sweet cherry of 0.43 kg CO₂-eq/kg from the agricultural production up to a UK regional distribution center.

Conclusions This study is one of the first assessments of the CF of sweet cherry production worldwide. According to our evaluation of improvement scenarios, the CF of Chilean sweet cherry production could be reduced by a diminution of diesel use and optimizing fertilizer application. New field data or evaluation of CF on other life cycle stages could further improve the knowledge on the CF of sheet cherry. Additionally, the determination of other

impact categories, such as eutrophication, acidification and impacts related to water use could be added to develop a more complete life cycle assessment.

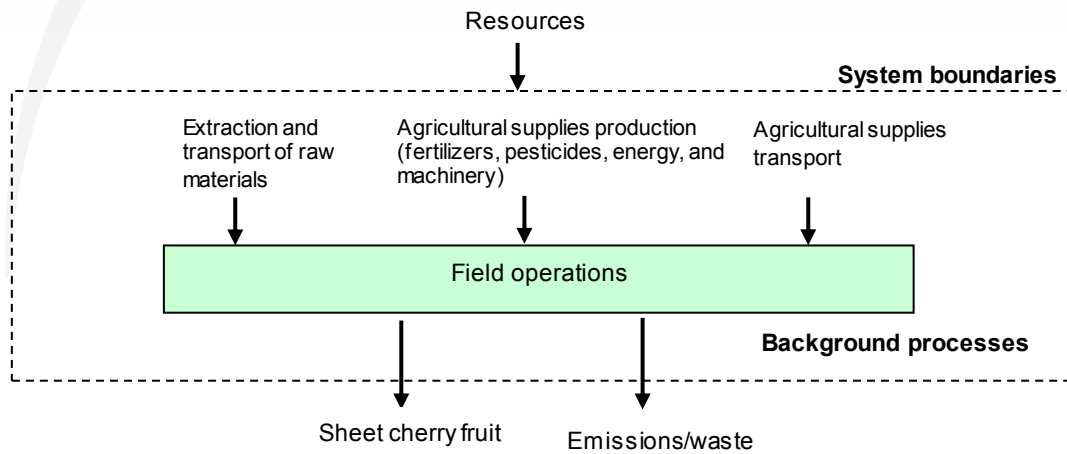


Figure 1. System boundaries of sweet cherry production system.

Table 1. Contribution of agricultural factors to the CF of Chilean sweet cherry production under study.

Agricultural factor	Average (kg CO ₂ -e/FU ^a)	Lower value ^b (kg CO ₂ -e/FU)	Upper value ^b (kg CO ₂ -e/FU)
Fertilizers	0.130	0.116	0.162
Pesticides	0.010	0.009	0.012
Diesel	0.170	0.132	0.218
Machinery (tractors, tools)	0.020	0.016	0.025
Electricity for irrigation	0.080	0.067	0.095
Total carbon footprint	0.410	0.360	0.467

^a Functional Unit = 1 kg of harvest sweet cherry.

^b Lower and upper limits of the 95% confidence interval on the uncertainty distribution.

Acknowledgments The present work was supported by CONICYT (Chile) - FONDECYT Project N° 11140765.

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P16. LCA of Apple Cultivation: from a Focus on Full Productive Years to the Entire Orchard Cycle

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While most agricultural LCAs tend to focus on full productive years of fruit orchards, fruits for sale in shops may come from both highly and less productive trees. Using apple as a case study, the present research contributes to improving the scientific knowledge on the impacts associated with the less productive years within an orchard cycle, and on how these affect the impact associated with the entire fruit growing cycle.

The data inventory comprises a large dataset of 512 apple orchard records spread over 70 farmers in Flanders (Belgium) and covering an eight year period to account for input and yield variability. The 512 orchard data records were categorized into three production methods and three orchard production phases. The product environmental footprint (PEF) and related ILCD impact assessment method was used to calculate impacts per tonne apples. The annual median impacts per ton are then used to describe a “typical” orchard for each orchard production phase and production system. The impacts associated with an entire orchard cycle were subsequently quantified using these annual median impacts and using weighting factors based on the yield contribution of each orchard phase to the total yield obtained throughout the orchard lifespan.

A large variability amongst the 512 orchard record impacts can be observed. Nevertheless, looking at median impacts, young trees generally perform worse than full productive trees. This is particularly true for organic farming, due to the FU chosen (1t), as young trees are associated with low yields. Old trees on the other hand, are associated with lower impacts in integrated and organic production orchards, while being higher than their full productive counterparts in conventional farming. Calculated impacts for the entire orchard cycle are, on average, higher than full production impacts in conventional and integrated farming, while lower in the case of organic farming. A mere focus on full productive trees is in the majority of the impact categories an underestimation of the entire orchard impacts for conventional and integrated farming, while it is an overestimation for organic farming.

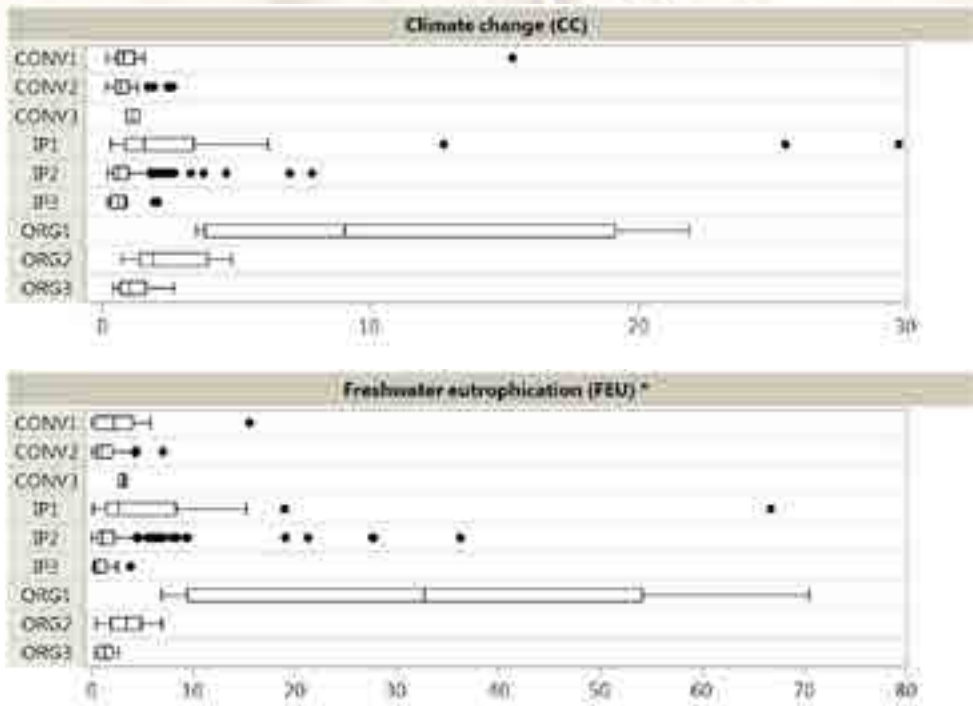


Fig 1 Normalised impacts per tonne of apples for each production system and orchard production phase, expressed in % of the annual impact of one reference person living in Europe: boxplots showing median value, range and outliers for climate change (CC) and freshwater eutrophication (FEU). For FEU, an outlier was removed for IP1 at 62.7% to enhance readability of the graph (impact category marked with an *).

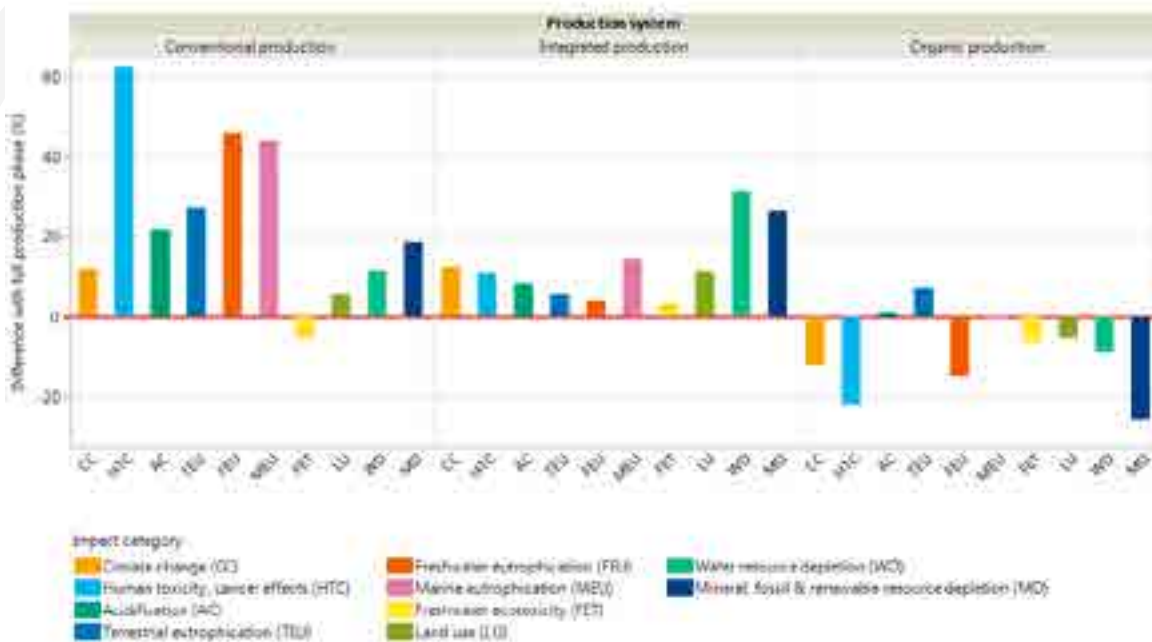


Fig 2 Impacts per tonne of apple based on the entire orchard cycle expressed in % increase or decrease compared with impacts of the full production phase, as indicated with a red reference line. Positive bars show the extent to which the orchard cycle impact is an increase compared with the full production phase and thus represent cases where a mere focus on the full production phase would be an underestimation of the orchard cycle impact.

P17. Eco-design in fruit and vegetable farming systems: moving forward in the AGRIBALYSE program

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A large number of LCA studies for agricultural products have been carried out over the last few decades, producing benchmark values and feeding databases such as AGRIBALYSE (www.ademe.fr/agribalyse). These studies enabled the identification of food production, highlighting the importance of sustainable farming systems, and showing more specifically the high impact of some agricultural operations. AGRIBALYSE partners now feel that the focus should move further towards defining and assessing with LCA innovative agricultural practices/systems, which would ensure both environmental and economic performances.

Several projects on fruits and vegetables are ongoing, following a similar approach based on AGRIBALYSE methodology. Innovative practices from agronomic research and farmers are identified, focusing on the ones targeting environmental hotspots. These practices are then assessed (and possibly optimized), and potential environmental gains communicated to the stakeholders. At the end, the “Eco-designed product“ LCIs will be included in the AGRIBALYSE database, in unit format.

For now, analysis is ongoing to provide a benchmark for the 10 most consumed fruits and 10 most consumed vegetables in France (table 1). In addition, analyses of eco-efficient production systems are being carried out. Work on apples and tomatoes is the most advanced, testing new heating systems and orchard disease management. In depth work with pineapple production on Réunion Island (France) is also ongoing. Such projects could be replicated for many different kinds of products and conditions. We hope that these projects will contribute significantly to more sustainable practices, and to a European market for green products.

Table 1. Main fruits and vegetables consumed in France and available dataset

Vegetables	LCI source	Fruits	LCI source
tomatoe	AGRIBALYSE v1	Apple	AGRIBALYSE v1
Carrot	AGRIBALYSE v1	Banana	WFLDB
Melon	AGRIBALYSE v2	Orange	WFLDB
Salad	AGRIBALYSE v2	Clementine	AGRIBALYSE v1
Onion	AGRIBALYSE v2	Peach/Nectarine	AGRIBALYSE v1
Zucchini	AGRIBALYSE v2	Pear	AGRIBALYSE v2
Cauliflower	AGRIBALYSE v2	Table grapes	-
Cucumber	AGRIBALYSE v2	Strawberry	AGRIBALYSE v2
Endive	AGRIBALYSE	Apricot	WFLDB

	v2		
Leek	AGRIBALYSE v2	Lemon	WFLDB



P18. How precision agriculture enhance the environmental profile of a pear orchard

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Objective: Precision Agriculture (PA) is a crop site-specific management system that aims to enhance sustainability. It adopts environmental friendly agricultural practices, like variable rate application (VRA) i.e. applying inputs rates according to the requirements of the crop. Life Cycle Assessment (LCA) methodology was used to evaluate the environmental impact of nitrogen (N) VRA in a pear orchard and compare it to conventional uniform N application.

Methods: A Cradle to Gate system with functional unit 1 kg of export quality pears was analysed. High quality primary field data and all emissions during pear growing and the supply chains of all inputs were included, except for the tree nursery production process. A methodology was adopted for modelling individual years and averaging over the orchards life time, rather than calculating the impact results for each year and then averaging. Since primary data were available for a few production years and yields were highly fluctuating, the involved farmers' and agronomists' experience/expert judgment was used for creating the necessary data time series.

Results: The important impact categories, climate change and particulates are largely determined by CO₂, N₂O, NO_x, and NH₃ emissions to air that resulted from fertiliser production and application, and CO₂ emission from tractor use (absolute numbers in Figure 1). Results showed that more efficient use of fertilisers with VRA combined with higher yield significantly reduced the overall environmental impact especially in the case of climate change and particulates, despite the higher overall fuel consumption the VRA year (Figure 2).

Implications/conclusions: LCA has proven to be a useful tool to evaluate alternative fertiliser management systems in a Greek pear orchard, indicating the environmental impact reduction potential of the precision agriculture application.

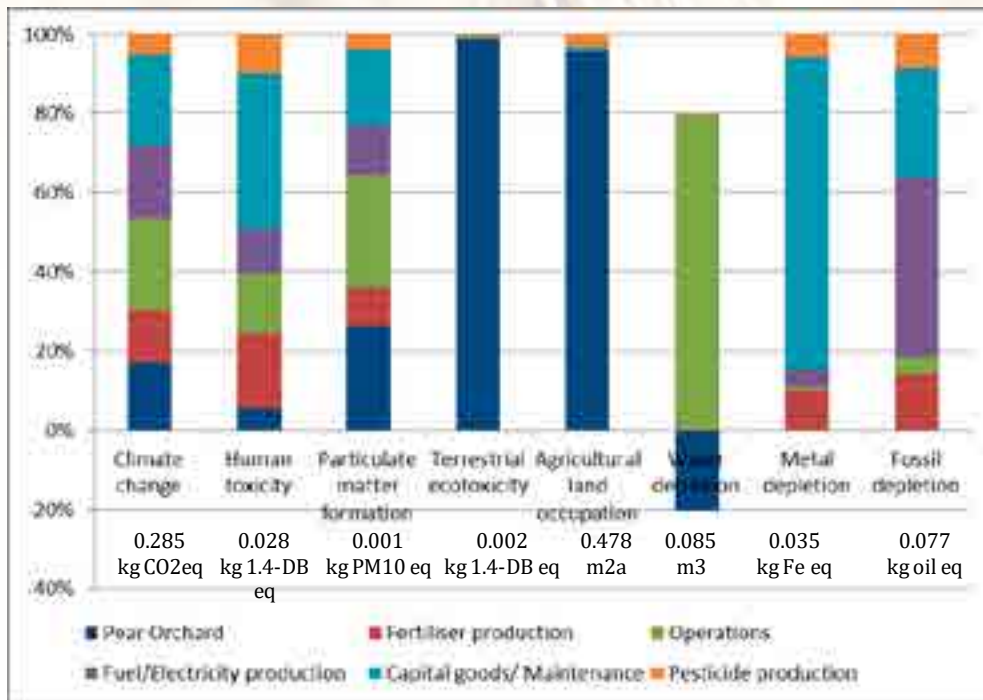


Figure 1: Processes contribution to the most significant impact categories in the characterisation phase

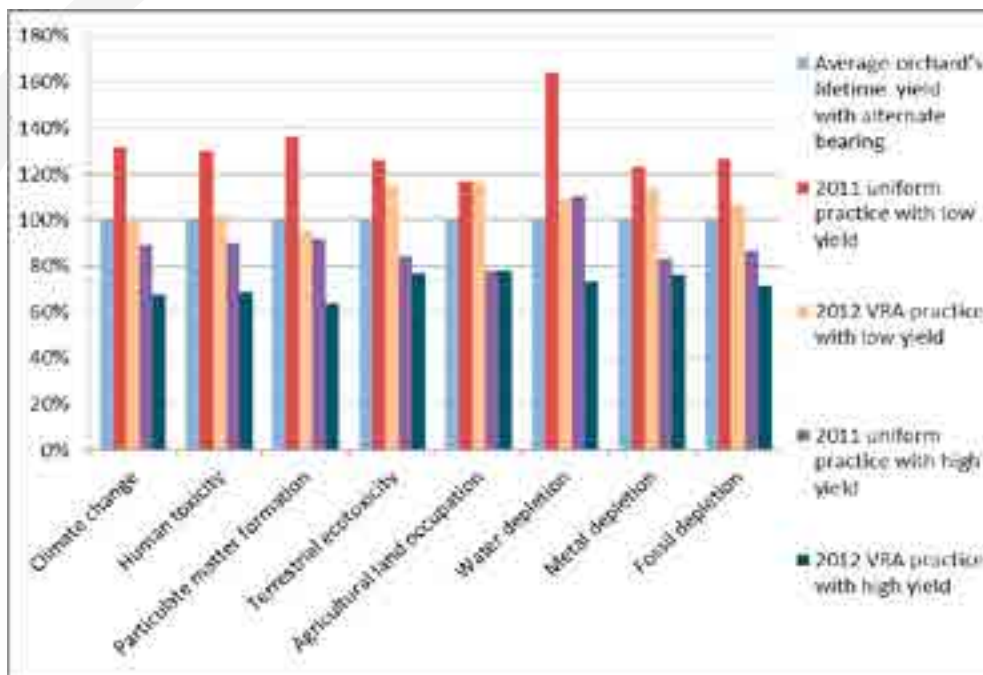


Figure 2: Yield comparisons of the Average pear orchard's yield with alternate bearing¹; the single years with low yields: 2011 uniform practice and 2012 VRA practice; the single years with high yields: 2011 uniform practice and 2012 VRA practice

¹ When a year of high yield is followed by a year of low yield due to inherent plant factors

P19. Life Cycle Assessment Of Cherry Production In Southern Italy

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OBJECTIVE

The study aims at quantifying and evaluating via LCA the environmental sustainability of cherry production in the southern Italian Apulia region.

METHOD

An LCA approach is used that considers different lifecycle phases, namely: the agricultural operations and the transformation system which gives two intermediate products for the food manufacturing industries, namely cherries in SO₂ and cherries in alcohol. The functional unit is one tonne of processed cherries. The primary data was collected on site from a local Apulian farm (Fig. 1) and from the processing plant (Fig.2) (which is responsible for more than 50% of the regional production of SO₂/alcohol cherries). Secondary data was selected from commercial databases.

MAIN RESULTS

The results highlight that, as in other food systems, the agricultural phase scores worst in almost all the impact categories compared to the transformation phase. However, it is interesting to note that, in particular for the cherries in alcohol system, the overall environmental burden of the transformation phase is not much lower than the one due to the agricultural operations. This is in contrast with the results shown in studies of other food products of the Apulia region such as wine, extra virgin olive oil and pasta.

Overall the cherries in SO₂ system has a better environmental profile compared to that of the cherries in alcohol.

IMPLICATIONS

The results of the research show that different environmental improvements could be achieved for this cherry product system by implementing more efficient transportation and by recycling some of the solutions (such as the hydro-alcoholic one).

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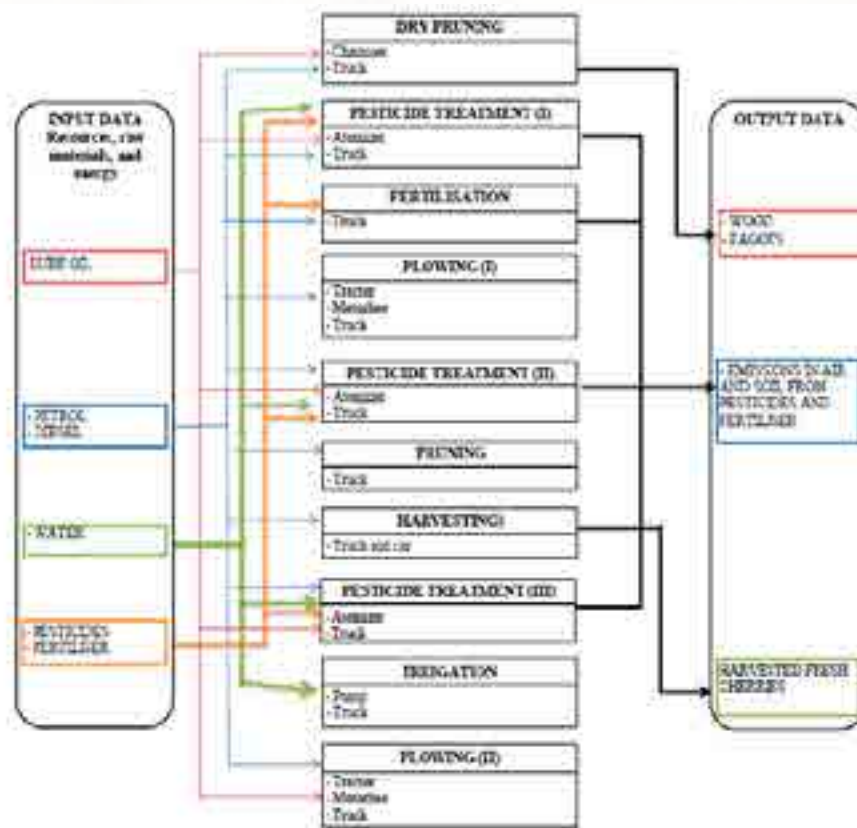


Fig 1 - Flow chart of the agricultural phase

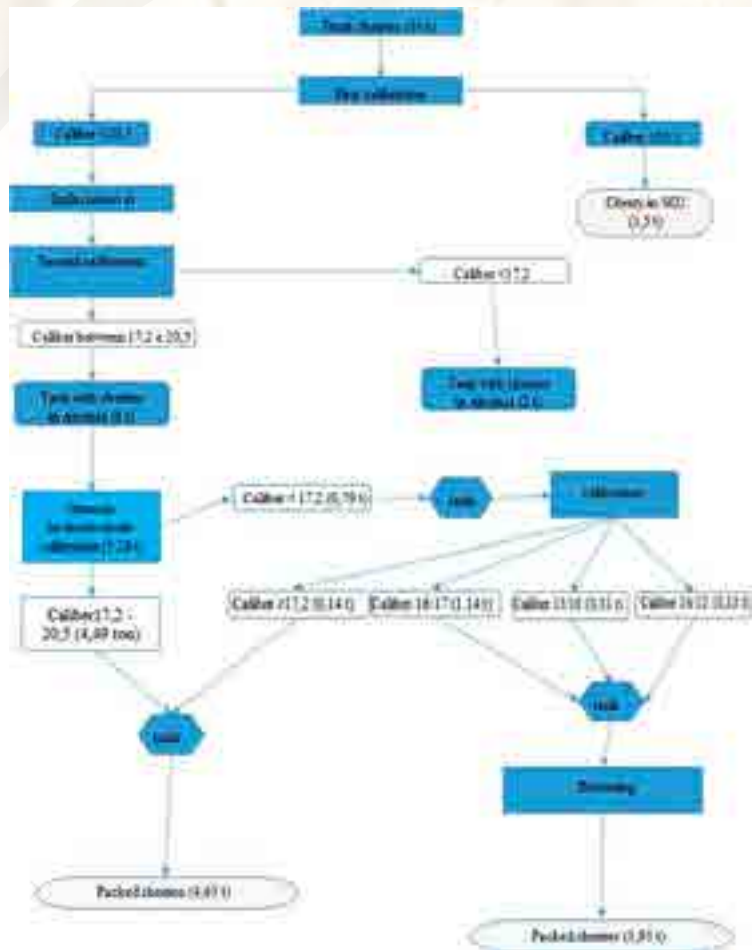


Fig 2 - Flow chart of the cherry transformation system

P20. Comparative Lca Of Maize Crop With Conventional And Transgenic Seeds

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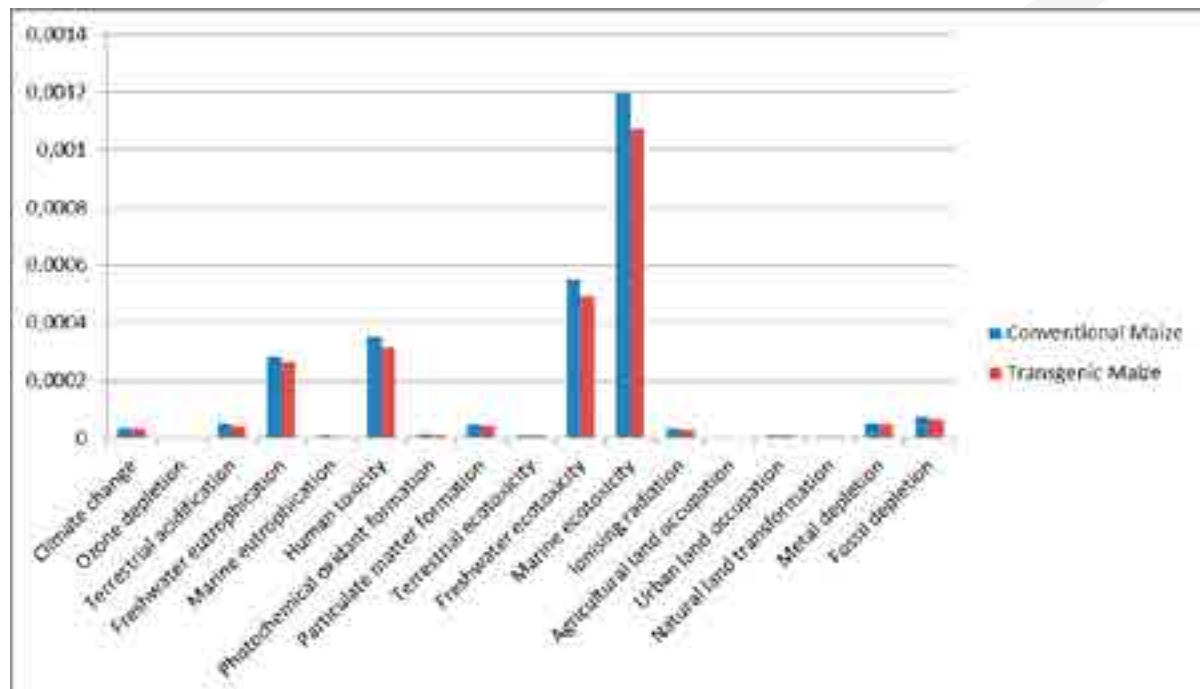
2. Federal Institute Catarinense (IFC – Blumenau)

The environmental aspect is an important issue for decision-making in the agricultural sector due to the high impacts associated with food production. The sustainability of the sector can be defined through a Life Cycle Assessment (LCA). Therefore, the aim of this study is to compare the maize crop production in Brazil, considering two scenarios, (1) with conventional and (2) transgenic seeds. The adopted functional unit was 1 kg of corn produced at the farm gate. The case study is based on crop cultivation in southern Brazil. At this region, it can be identified two kinds of production system: with 100% of conventional seeds, which sets scenario (1) and with 90% of transgenic seeds and 10% conventional, setting the current scenario (2). The latter are placed in the city of Campo Belo do Sul, state of Santa Catarina, Brazil. The LCI was built with primary data and modeled in SimaPro 8.1 software. Emissions to air, water and soil due to the use of chemical fertilizer, pesticides and herbicides were not included in the system boundaries. In both cases, the herbicides management per hectare is the same, unlike the insecticide management that is used only in conventional seeds crops. However, it should be noticed that the productivity are different for both scenarios (see Table 1). The life cycle impact assessment (LCIA) method used was the ReCiPe considering the impact categories at midpoint and normalized values for "World ReCiPe H". From the results presented in the study, the ACV points to producers that the three categories that impact most are Marine ecotoxicity, Freshwater ecotoxicity, and Human toxicity, but the worst case scenario is with cultivation of conventional seeds. We conclude that the use of transgenic seeds it can be a good option for this economic sector.

Table 1. Life Cycle Inventory of the maize cultivation

Stage/Process/Substance	Unit	Sc. 1	Sc. 2
Maize cultivation			
<i>Inputs</i>			
Limestone	kg	6,48E-01	5,83E-01
Urea	kg	4,63E-02	4,17E-02
P2O5	kg	1,39E-02	1,25E-02
K2O	kg	9,26E-03	8,33E-03
Tractor	kg	5,56E-04	5,00E-04
Harvester machine	kg	9,26E-05	8,33E-05
Agricultural machinery	kg	9,26E-04	8,33E-04
Diesel	kg	5,56E-03	5,00E-03
Gliphosate	kg	0	2,50E-04
Atrazine	kg	4,63E-04	4,17E-04
Triazine compounds	kg	0	0
Carbofuran	kg	9,26E-05	0
[sulfonyl]urea-compounds	kg	2,78E-05	0
Pyretroid compounds	kg	2,78E-04	0
Pesticide unspecified	kg	1,85E-04	0
<i>Outputs</i>			
Maize	kg	1	1

Figure 2. Comparison of the impact categories of maize production



POSTER PAPERS

METHODS AND DATA 1

4. Methods and data (1)

P21. Dynamic LCA modelling of agriculture by integrating biological models into LCA – case Finnish beef production

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The livestock sector is responsible for the largest share of environmental impacts of agriculture. Improving beef and dairy production systems plays an important role in climate change mitigation. Holistic and dynamic system modelling is needed to avoid sub-optimization and to find the best solutions.

In FootprintBeef-project (2012-2014) biological models of animal growth, and feed production were integrated into LCA model to create dynamic system models. Current dairy and suckler beef production systems were assessed, and automatic scenario studies were conducted for bulls and heifers of both systems to find out the best ways to minimize the environmental impacts of those systems. Also new greenhouse gas (GHG) emission and eutrophication (Saarinen 2010) assessment methods were applied.

As expected, beef production which uses resources in a balanced and efficient way is performing the best in all impact categories. The reduction potential of current production is nearly one fourth. Still, many improvement options cause conflicting changes in different impact categories. Thus, it is valuable that scenarios can be built combining multiple improvement options simultaneously (see Table 1). Applying the new GHG-emission models cause large changes to emissions from different sources, even if the total change for Finnish animals is not very large (see Table 1.).

The system model integrating animal production, manure production, feed cultivation and their environmental impacts is able to give holistic view of the system and its hot spots. The developed model allowed the running of multiple scenarios on beef production (see examples in Table 2.). Sub-optimisation of particular stages of production (enteric fermentation, feed production) can be avoided. Modelling environmental impacts of a single animal can give new insights for extension services and authorities, but farmers still need simplified tools to be applied at farm level, which would be the next step of the current study.

		Base- line, grain share 40%, D660	D660 -> D690	Great grass yield, D690	Grain share 20%	Grain share - 20%, D690	Best combi- nation, grain share 40%	Best combi- nation, grain share 20%
Dairy bull								
Daily weight gain (6-19 months)	kg/ day	1,02	7%	7%	-15%	-8%	24%	5%
Dry matter intake (6-19 months)	kgDM/ day	8,53	1%	1%	-2%	-3%	11%	3%
Efficiency	kgCW/ ha	373	6%	33%	-6%	1%	34%	40%
Livestock units	LU/ha	1,10	0%	26%	6%	8%	14%	35%
GWP	kgCO₂ -eq./ kgCW	17,1	-4%	-7%	6%	0%	-19%	-15%
Land use	ha	0,89	-5%	-25%	7%	-1%	-25%	-28%
Eutrophication	gPO₄- eq./ kgCW	17,5	-3%	-23%	14%	8%	-23%	-17%
Acidification	gPO₄- eq./ kgCW	49,9	0%	-6%	25%	23%	-8%	12%

Table 1. Example of changes in key parameters and environmental impact categories of different analysed scenarios.

	Dairy		Suckler	
	Bull	Heifer	Bull	Heifer
Barley cultivation modelled as annual production	9 %	4 %	28 %	17 %
Grass cultivation modelled as perennial production	-32 %	-32 %	-34 %	-35 %
Methane from enteric fermentation	14 %	11 %	12 %	8 %
Total change	0 %	-3 %	1 %	-5 %

Table 2. Difference in percentages when applying Regina (2013) method for direct nitrous oxide emissions from soils and Ramin & Huhtanen (2012) method for methane from enteric fermentation compared to baseline (IPCC 2006 Tier 2 method).

P22. How to conduct a LCA study for multi-product integrated systems

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Integrated systems can combine crop, livestock and forest cultivation, supporting the production of at least three types of product from the same land area over a defined period – for instance soy, silage, sorghum, cattle, eucalyptus, among others. These systems are based on intercropping, succession or rotation, and can optimize biological cycling of nutrients between plants and animals, improving production efficiency and maintaining long-term soil fertility in an environmentally sustainable manner. In Brazil, there are plans to increase the production area of integrated systems to four million hectares by the year 2020. However, benchmarking the environmental efficiency of integrated systems against conventional through quantitative methods such as Life Cycle Assessment (LCA) can be challenging, especially due to synergic effects between the system components and need for allocation across multiple products.

LCA studies conducted for different purposes or audiences may involve different allocation decisions (e.g. based on economic, energy or mass outputs), leading to very different conclusions. Soybean produced for biodiesel in integrated systems is just one example where such conclusions become relevant.

The aim of this paper is to review how LCA studies on multi-product integrated system have been conducted to date, with a focus on the main approaches, definition of functional unit and allocation procedures. Recurring limitations and challenges will be highlighted.

This paper will then propose recommendations for application of LCA to integrated systems, and put forward a case for system expansion and consequential LCA as a way to avoid limitations associated with allocation. We explore how wider ecosystems services not well represented in LCA may be accounted for, either using separate metrics or e.g. as co-products with the LCA framework.

P23. Global Guidance for Life Cycle Impact Assessment Indicators and Methods and their relevance for food LCAs

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The attention to environmental issues and globalized markets call for harmonized guidelines to quantify environmental impacts in a consistent way over the entire supply chain of product. In late January 2016 a Pellston Workshop™ on „Global Guidance for Life Cycle Impact Assessment Indicators and Methods“ was held in Valencia, Spain to meet these needs. The goal of the workshop was to reach consensus on recommended environmental indicators and characterization factors in the four agriculture relevant areas of global warming, particulate matter emissions, water use impacts (both scarcity and human health impacts), land use impacts as well as overall LCIA framework and crosscutting issues. The 40 participants represented a well-balanced mixture of LCA experts and domain experts, with LCA users from industry, governments and NGOs from different regions in the world. The one week workshop was characterized by intensive discussions, exchange of arguments and positions, passion and at the same time openness and fairness. Finally, the participants agreed on tangible and practical recommendations on environmental indicators, including substantial innovations. Each group also provided tables of recommended characterization factors to operationalize the application of the recommended indicators. This will bring LCA and LCIA in particular a big step ahead. Several of the main recommendations are directly relevant for agriculture (see Table 1).

Additionally, all recommended indicators and characterization factors were tested on common food LCA case study on the production, distribution and cooking of rice (Frischknecht et al., 2016). Three distinctly different scenarios of cooking rice were defined and supported with life cycle inventory data. The case study results illustrate well the practicality of the finally recommended impact category indicators.

This workshop has demonstrated that such a well-prepared science-based consensus finding process does not freeze scientific knowledge but rather promotes progress in science and at the same time fosters the practicality and robustness of the recommended indicators. It was also mutually agreed to profit from the momentum created to install a structure which provides stewardship for both recommended characterization factors as well as for their regular update in the future, under the flagship project 1b on LCIA guidance of the Life cycle

Initiative. The official launch of the Valencia Guidance on LCIA is scheduled for the Eco-balance conference 2016.

Table 1: Main recommendations for the considered Impact categories and food LCAs

For climate change:

It is recommended to use two separate impact categories for climate change:

- GWP100 as the indicator for shorter-term and
- GTP100 as the best proxy for longer-term impacts, including climate-carbon cycle feedbacks for all climate forcers.

For human health impacts of fine particulate matter (PM2.5):

Intake fractions and exposure response functions based on the Global Burden of Disease are recommended a) for an urban archetype or more than 3000 specific cities of the world, b) for rural archetypes in 19 world regions, c) for secondary PM with interim factors including NH₃ agricultural emissions, and d) for indoor PM2.5 exposure, in particular for cooking.

For water use:

- The AWARE scarcity indicator based on the Available Water REmaining is recommended, emphasizing the need to perform sensitivity analyses with conceptually different methods and to further ground truth the approach through 10 test case studies.
- Endpoints impacts on human health are determined based on the influence of 1m³ water consumed on the malnutrition damage due to food deficiency of the undernourished population. This accounts for a) the influence of 1m³ water consumed on the water available for agricultural use, b) the food production losses resulting from reduced irrigation (kcal / m³) modified by trade adaptation and c) the average value of malnutrition damage per kcal food deficiency of the undernourished population as determined in the global burden of disease.

For land use impacts on biodiversity:

The recommended characterization factors (CFs) developed by Chaudhary et al. (2015) are provisionally recommended for hotspot analysis in LCA only. It is recommended to use ecoregional CFs for foreground systems rather than country averages. The reasoning for these recommendations includes a) the global coverage of six major land use types covering most products' life cycles, b) the consideration based on empirical observations of many important aspects: species richness; local effect of different land uses; links between land use and species loss through the Countryside-SAR model; the relative scarcity of affected ecosystems; the threat level of species, c) the youth of the method and the need to further test it in a wide range of product systems, regions or application areas and , d) the limitations in land use types, management intensities, and the substantial uncertainties.

For cross cutting issues, the main novelties were:

- An updated framework distinguishing intrinsic, instrumental and cultural values including human health, ecosystem quality, as well as natural resources and ecosystem services.
- More transparent reporting is needed regarding the impact pathway, units, consistently defined reference states, the original and aggregated spatial scales, uncertainties and variability, modelling and data choices, and consistent global normalization references.
- The spatial scale of regionalized models needs to reflect the nature of impact, and CFs need to be reported for two different timeframes (till 100 years and longer term whenever relevant,

and if possible in an additive way), with marginal and average CFs.

- We recommend to characterize ecosystems and/or species in a way that takes resilience, rarity and recoverability into account

P24. CAP'2ER[®], the environmental footprint calculator and decision making for ruminants production systems

Sindy Moreau¹, Catherine Brocas², Jean-Baptiste Dollé³

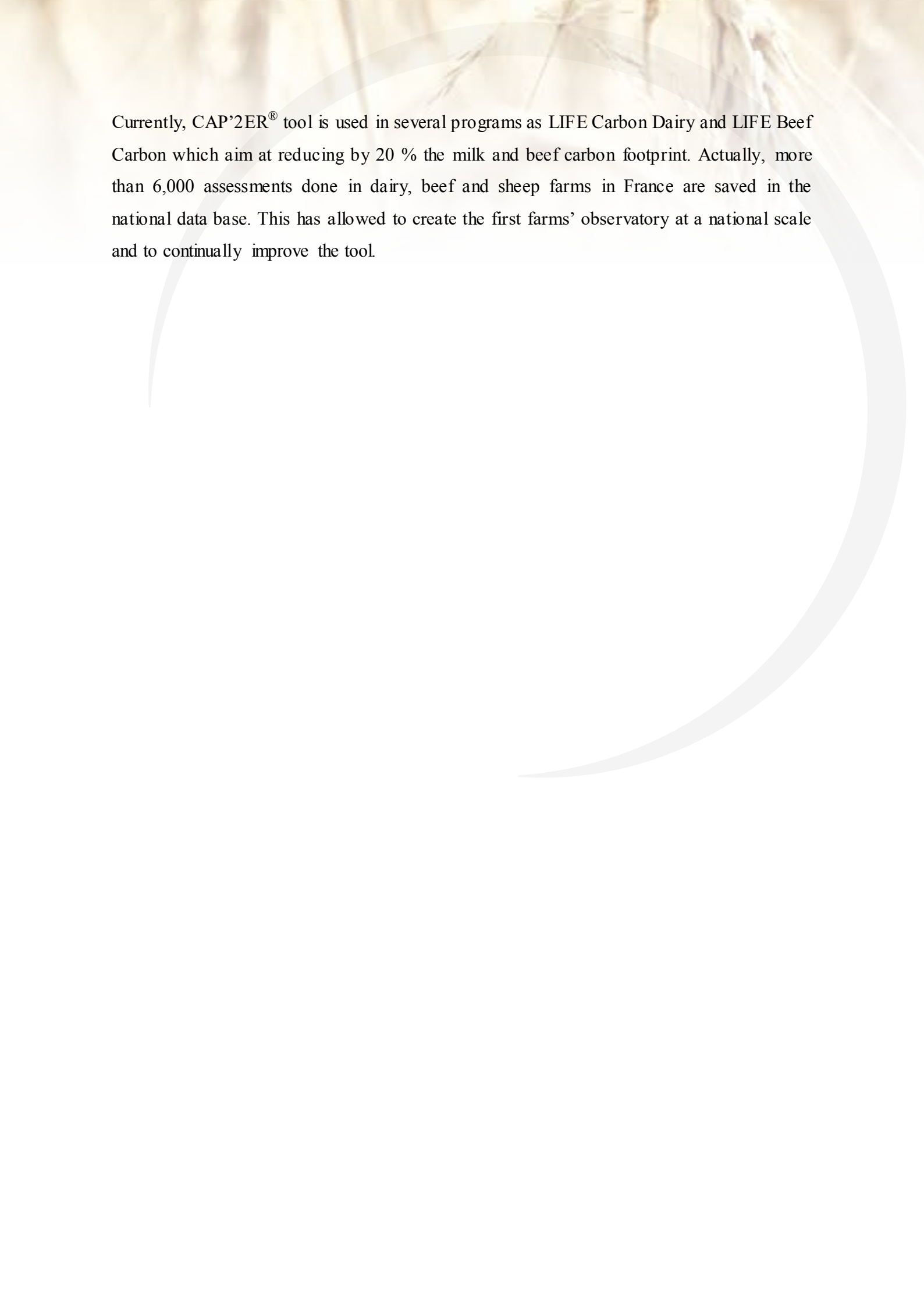
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CAP'2ER[®] is an environmental assessment tool developed in France for dairy, beef and sheep farms (Figure 1). Answering to Life Cycle Assessment (LCA) standards, the system boundaries covered by CAP'2ER[®] represent 'cradle-to-farm-gate' (on-farm impacts and embodied impacts from inputs used on the farm). The methodology is a multicriteria assessment (GHG emissions, eutrophication, acidification, and energy use) based on international methodologies (IPCC, CML). The tool also evaluates positive contributions like carbon sequestration, biodiversity and food performances. Carbon storage is assessed by French standard values depending on the type of surfaces. The counting of agro-ecological elements (hedges, trees...) allows to evaluate biodiversity. Finally, food performances are evaluated according to the methodology PerfAlim[®]. The functional unit is the quantity of product in liter of fat and protein corrected milk and kg live weight leaving the farms (Figure 2). The livestock unit (LU) or area mobilized (ha) are also used. By this way, items allow to explain farmers the link between practices and environmental impacts. At each step, farm results are compared to a reference according to the system.

The first objective of the tool is to sensitive stakeholders on the link between environmental impacts and breeding, and the second one is to allow the development of mitigation action plans on farms. By this way, the CAP'2ER[®] level 1 (Table 1) is a simplified tool which requires 30 technical data. Free on line, it aims at educating farmers and livestock advisers to consider environmental issues in order to do a quick evaluation of the environmental footprint, to get farmers into position compared to references and to create a national observatory. The CAP'2ER[®] level 2 aims at including environmental assessment in the advising approach and creating the link with technical aspects. Reserved to specialists in breeding production systems, the level 2 allows to set up mitigation action plans in order to reduce impacts and to increase positive contributions.



Currently, CAP'2ER[®] tool is used in several programs as LIFE Carbon Dairy and LIFE Beef Carbon which aim at reducing by 20 % the milk and beef carbon footprint. Actually, more than 6,000 assessments done in dairy, beef and sheep farms in France are saved in the national data base. This has allowed to create the first farms' observatory at a national scale and to continually improve the tool.

Figure 1: CAP'2ER tool free on line: www.cap2er.fr/Cap2er



Figure 2: Example of output data of the CAP'2ER tool



Table 1: Impacts environmental assessment according two levels

Objectives	Sensitization	Decision making
Levels	CAP'2ER® level 1	CAP'2ER® level 2
Public	Farmers, students...	Advisers
System boundaries	Dairy, beef or sheep units / Products	
Primary data needed	= 30	= 150
Time to collect data	30 min	3 h.

P25. The need for co-product allocation in the Life Cycle Assessment of agricultural systems – is “biophysical” allocation progress?

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Several new “biophysical” co-product allocation methodologies have been developed for LCA studies of agricultural systems based on proposed physical or causal relationships between inputs and outputs (i.e. co-products). These methodologies are thus meant to be preferable to established allocation methods, such as economic allocation, under the ISO 14044 guidelines. The aim here was to examine whether these methodologies really represent underlying physical relationships between the material and energy flows and the co-products in such systems.

To meet the requirements of Step 2 in the ISO hierarchy, an allocation methodology must be based on causal relationships within the system, and for practical reasons these are usually quantified through mathematical modelling. In order to establish physical causality between co-products and inputs to the system, it must be possible to change the value or delivery of any co-product independently of other functions delivered by the system. Two systems utilizing agricultural LCAs involving co-product allocation were used to provide examples of current methodological practices and issues, namely 1) crop production and 2) multiple co-products produced by livestock. The premise of many biophysical allocation methodologies has been to define relationships which describe how the energy input to agricultural systems is partitioned between co-products. However, as represented in Figures 1 & 2, none of the outputs from animal or crop production can be considered independently from the rest on the basis of the inputs to the system. For example, is not possible to produce milk without the feed energy input needed to raise heifers into adulthood.

As a conclusion, the proposed “biophysical” allocation methodologies for various aspects of agricultural systems will not be able to adequately explain how the physical parameters chosen in each case represent causal physical mechanisms in these systems. Allocation methodologies which are based on these shared (but not causal) physical properties between co-products are not preferable to allocation based on non-physical properties within the ISO hierarchy on allocation methodologies.

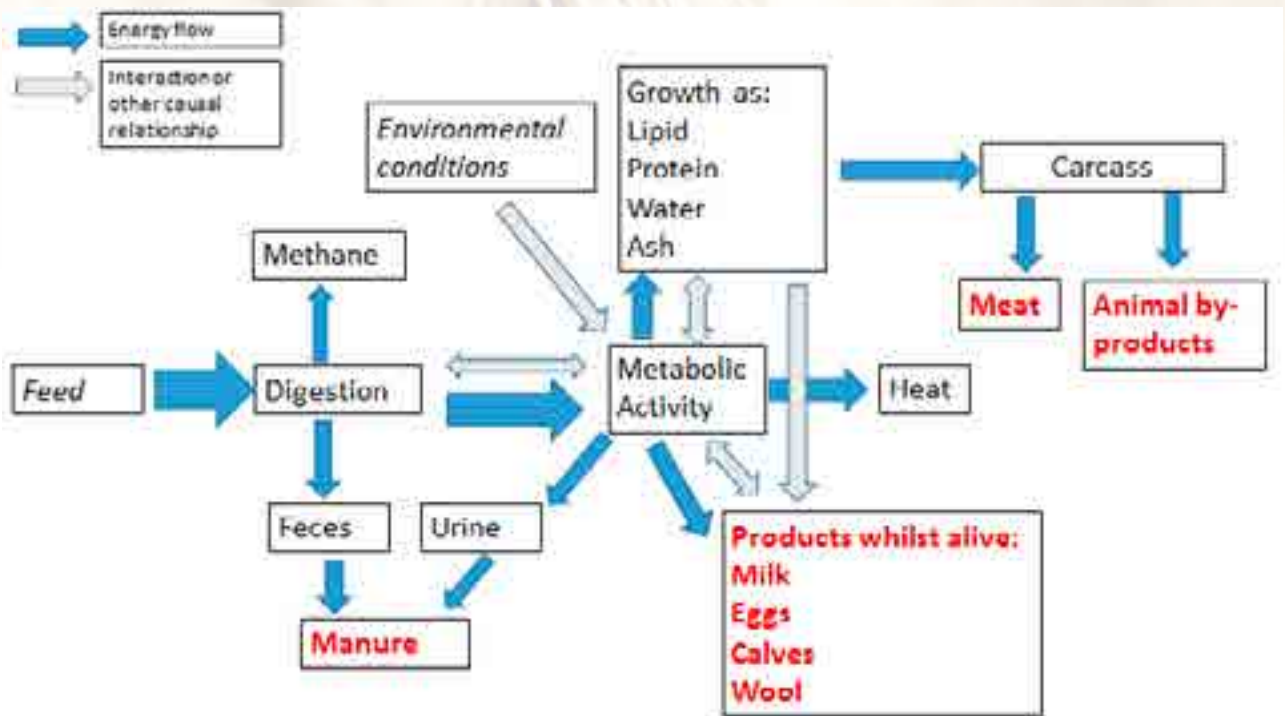


Figure 1 A simplified illustration of energy flow and other causal relationships in animals in livestock production systems. Inputs to the system are indicated in italics while potential co-products are in red bold font

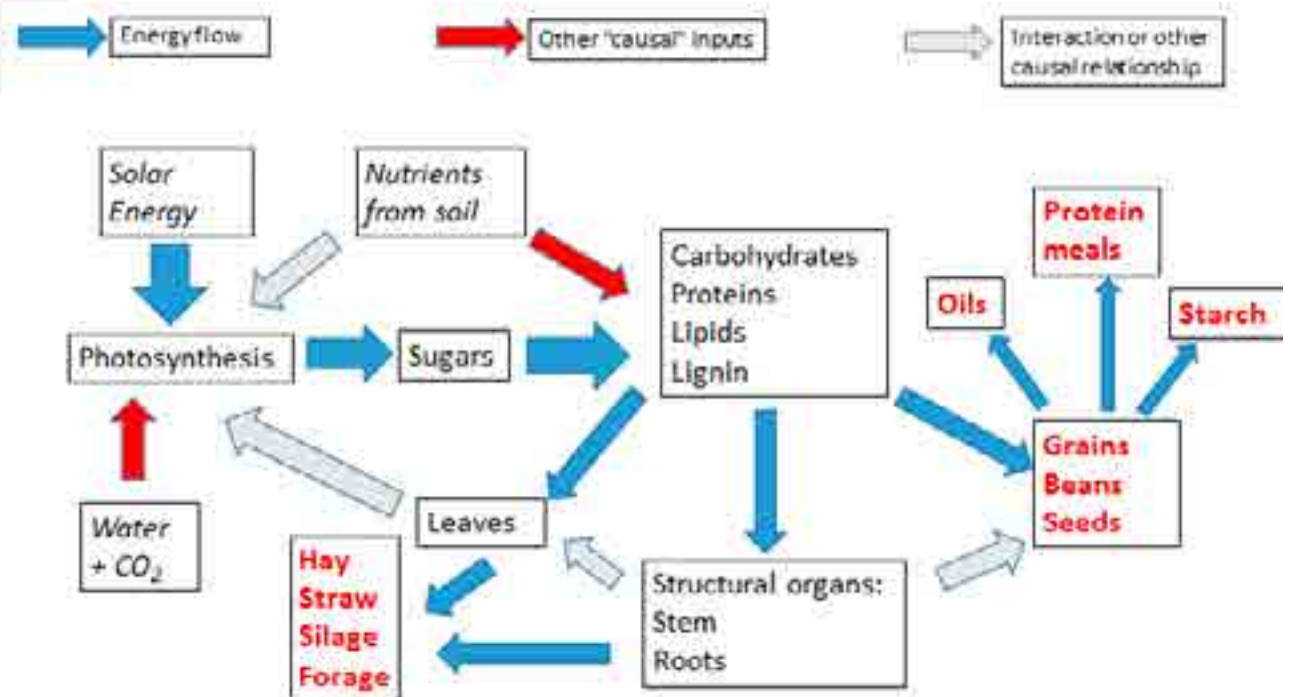


Figure 2 A simplified illustration of energy flow and other causal relationships in crop production. Inputs to the system are indicated in italics while potential co-products are in red bold font

P26. Improving regionalized life cycle inventories with mass-balance models and scale-consistent data

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ABSTRACT

Objective: Among all Life Cycle Assessment (LCA) stages, life cycle inventories (LCI) commonly demand the most time and effort. LCI datasets are rarely fully regionalized, particularly for agri-food materials. Agricultural products inputs and outputs crucially depend on location and technology. The level of regional representability of materials in standard databases is thus indeterminate. Here we propose a new mass-balance model to build inventories that are fully scale-consistent.

Method: We present the model framework that can be used to obtain the inputs from nature and the technosphere and the outputs to air, water and soil. We implemented the framework for carbon (C) and nitrogen (N) and present results for the soil N sub-model. The N cycle is determined for the main agricultural exports in Portugal (e.g. maize, vine). We used activity data from Portuguese agricultural fact sheets and quantified the parameters in the balance using process-based models for each variable (e.g. regional productivity, soil organic matter accumulation and loss, nitrate leaching) that use spatial environmental data. The model was applied at regional scales, and to assure scale consistency, in the sense introduced by Morais *et al.* (2016). We inserted as a restriction that the aggregated sum of all inventory flows is equal to country-level totals.

Results: We provide regional results that include all soil N-related inputs and outputs in the country. The model provides direct inventory flows that can be used in highly spatialized LCI. We found that the most important advantage is the spatial dimension of results, which due to the scale-consistency of data can be aggregate at any level of detail from the local to the country scale. Nitrate leaching is the main source of uncertainty.

Conclusion: LCI is lagging behind the geospatial detail of most recent impact assessment models. The approach presented here enables the construction of more powerful inventories relating field data with regional or country aggregates.

Keywords: Life Cycle Inventory; Portugal; Regionalization; Mass balance; Scale consistency

1. Introduction

Life Cycle Inventory (LCI) is, among all Life Cycle Assessment (LCA) stages, the most demanding in terms of time and effort. It is during this stage that data is compiled to depict the complete life cycle. An inventory usually involves secondary databases for background data (lower-tier processes in the supply chain) plus primary data collected specifically for foreground processes (processes in the first tier of production). While primary data is usually site-specific or product-specific, background inventory data is rarely fully regionalized beyond the country scale. Some sectors are more demanding in need for highly regionalized data, such as the agri-food sector (Haas *et al.* 2000). The variance of LCA results for agri-food products is high (Teixeira 2015); country-scale differentiation in background databases may be insufficient in these cases (Reap *et al.* 2008). The construction of national-level LCIs is a newer trend (Yang, 2015), but there are some examples in the literature. Project Agribalyse (ADEME 2013) was mandated by the French government with the objective of creating database directives for French agricultural products. The World Food LCA Database (WFLDB) (Nemecek *et al.* 2014) aims to produce directives that support the establishment of regional databases for global agri-food products. These examples demonstrate a worry of ensuring that the approaches are coherent and consistent with other methods and models, resorting to international frameworks so that the way inventories are regionalized is the same for all regions, and thus that studies remain comparable. When regionalizing inventories, two (often) opposing forces must be balanced: (1) the need to have high local detail, and (2) the need to ensure comparability with other studies from other regions.

Two limitations that are often disregarded in typical approaches are explored in this work. We believe these two principles can help balance the two forces mentioned. First, there are usually no explicit requirements for mass balance check of flows for each inventory process. The sum of the mass of all input flows must equal the sum of the mass of output flows if the process is consistent. However, when building inventories each flow is typically calculated independently using different data sources. Second, there is also no explicit or implicit guarantee in typical databases that the sum

of input or output flows for all processes in a country, if applied to all activities taking place in the country, would be equal to the total for that country, as accounted for in national-level official statistics. We name this requirement “scale-consistency” (Morais et al., 2016). As an example, assuming that one record is representative of a given region within a country, it means that the amount of fertilizers used to produce a certain amount of one crop in that region must be such that if the same calculation was done for all products and all regions of the country, the sum of all fertilizers would be equal to the amount of fertilizers used in the country according to official statistics. The fact that these principles are not respected, at least explicitly, by most agricultural databases in the quantification of field-level input and output flows means that a new approach is essential.

Together, these two aspects enable high regional resolution, as mass balance models are scale-insensitive and can be applied to cells as well as to regions and countries. Scale-consistency then ensures a strong restriction for study comparison, as all input/output flows must be a scaled fraction of official statistics. We illustrate these principles next by introducing a mass balance model for two particular cases: carbon (C) and nitrogen (N). We then picked a sub-model of high complexity, the N balance in soils, and used available scale-consistent data sources to illustrate how the model may be calibrated for particular crops and regions in Portugal.

2. Methods

2.1 Mass balance models

We follow the mass balance model originally proposed by Teixeira (2010), whose schematic depiction in the most complete case, which is for pastures, is shown in Figure 1.

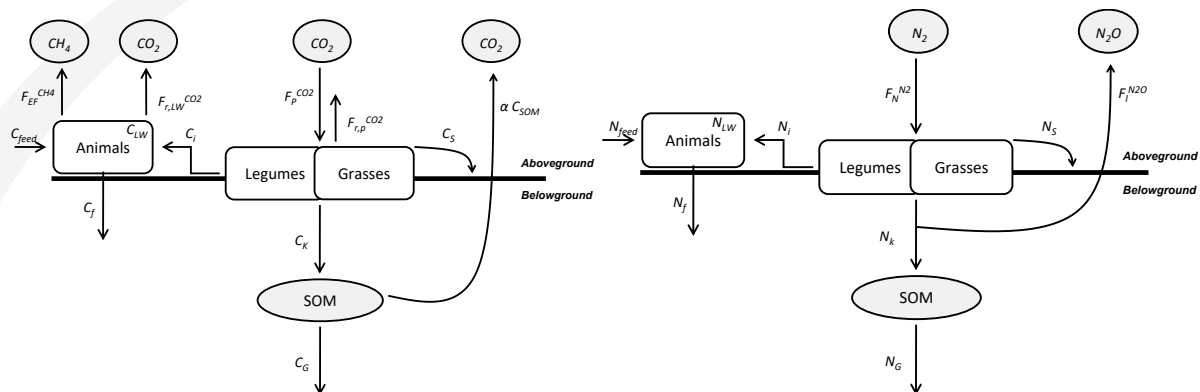


Figure 1: Carbon (left) and nitrogen (right) balances in grassland systems according to Teixeira (2010). C – Carbon; CH₄ – methane; CO₂ – Carbon dioxide; N – Nitrogen; N₂O – nitrous oxide; N₂ – Nitrogen gas; F – flux; SOM – Soil Organic Matter; P – Photosynthesis; r – respiration; S – Senescence; LW – Live weight; G – Groundwater; α – mineralization rate; f – faeces or manure; i – ingestion; K – organic matter input to soil from roots; l – legumes

Starting with the simplified scheme in Figure 1, which provides a generic framework for a mass-balance LCI, we introduced more detail and new variables into all sub-models (animals, plants and soil). Each compartment or sub-model can accumulate C and N. For animals and plants, this is equivalent to growth; for soils, this represents immobilization in the form of, for example, SOM. Accumulation can be calculated as balance between outputs and inputs, each of which should be calculated using separate models. It is therefore possible to perform this balance of input and output flows for the entire system or for each compartment. Due to limitations in space, in this paper we present only the most complex sub-model, which is soil N.

2.2 The soil nitrogen mass balance sub-model

The soil N mass balance (kg N/ha) follows the method laid out by the Organisation for Economic Co-operation and Development (OECD) in ‘Nutrient Budgets – Methodology and Handbook (Eurostat, 2013) which is also used in Portugal by Gabinete de Planeamento e Política Agro-Alimentar (GPP, 2015) of the Ministry of Agriculture. This balance calculates the accumulation of N in soils (N_{soil}) as a function of time (t), considering as inputs inorganic (N_{fert}) and organic fertilization (N_{man}), N fixation (N_{fix}), atmospheric deposition (N_{dep}) and seeds (N_{seed}), and as outputs the uptake of N by plants ($N_{pl,fert}$) and emissions to air of NO (N_{NO}), NH₃ (N_{NH3}) and N₂O (N_{N2O}).

The OECD calculates the balance for Portugal as a whole and it does not discriminate according to land use/cover class – which is required for use in LCA. We calculated a crop-specific balance at the agrarian region level in Portugal. We included wheat, maize (grain and silage), oat (grain and silage), barley, triticale, sunflower, grapes (table and wine), olive, pear, almond, tomato, rice, natural pastures and (improved) sown biodiverse pastures. When applicable, crops were divided between rainfed and irrigated. The time step is one year.

Additionally, some important N flows are not included in the OECD calculation. We included the N balance in eroded/deposited soils (ΔN_{ero}), the N deposited from feces and urine (only for pastures) during grazing (N_{fc}) and nitrate leaching (N_{leach}). The final balance equation is thus

$$\frac{\Delta N_{soil}}{\Delta t} = N_{man} + N_{fert} + N_{pl,fert} + N_{seed} + \Delta N_{ero} + N_{fix} + N_{fc} + N_{dep} - N_{leach} - N_{N2O} - N_{NH3} - N_{NO}. \quad (1)$$

2.3 Data used and scale-consistent adaptations

2.3.1 Inorganic (N_{fert}) and organic (N_{man}) fertilization

The quantity of inorganic N fertilizer applied for each crop was obtained from two sources. First, Morais et al. (2016) provides scale-consistent quantities of fertilizer applied per mass unit produced. The quantities were obtained using regional fact sheets from GPP, and corrected using total fertilizer application data in Portugal. The second source was the *Manual de fertilização das culturas* (LQARS, 2006) (“Manual of crop fertilization”, in Portuguese) which provides recommendations for best-practice fertilizer application to farmers, depending on the yield intended. Yields for both approaches were also obtained in Morais et al. (2016), considering the average for the period 2009-2014 for each crop and region as reported by INE (2015). The first strategy can be understood as the “real” amount of fertilizer actually applied by farmers, as the partial consumption scales up to the national total, the second case can be seen as a best case scenario in which farmers would not overuse fertilizers, applying only the minimum needed. As for organic fertilizers, only data from Morais et al. (2016) was used.

2.3.2 Nitrogen fixation (N_{fix})

Nitrogen fixation only takes place in sown biodiverse pastures due to the fact that, among the crop list we chose, they are the only land cover that includes legumes. The amount of N fixed was 0.026 kg N/kg dry matter according to the Portuguese *National Inventory Report* (NIR) of 2014 (APA 2014).

2.3.3 Atmospheric deposition (N_{dep})

Atmospheric deposition was included as the national average calculated by the *European Monitoring and Evaluation Programme* (EMEP, 2016) of 4.343 kg/ha.

2.3.4 Seeds (N_{seed})

Seeds possess N in their chemical composition, and as such are also an effective N input in soils. The amount of N in seeds was obtained from *Nutrient Budgets – Methodology and Handbook* (Eurostat, 2013) for annual cereals and sunflower. For permanent crops and pastures no N from seeds was included. We assume that the year of calculation was sufficiently far removed from

plantation/sowing for the effect to be negligible. Data for rice was obtained in Hara & Toriyama (1998) and for tomato in Gates (1954).

2.3.5 Nitrogen taken up by the plant ($N_{pl,fert}$)

N uptake by plants was obtained from several sources, depending on the crop, which for lack of space cannot be included here. The full list is available by request.

2.3.6 Feces and urine deposited during grazing (N_{fd})

Feces and urine deposition during grazing was calculated according to IPCC (2006) methods using national data from NIR (APA 2015). The amount of excrement deposited depends on the total amount of excrement produced by livestock. In this study we assumed pastures were grazed by adult cattle, using an excrement production factor of 80 kg N/animal. We also assumed that animals graze for a third of the day during the entire year (8 h/day). The stocking rate in natural pastures is 0.5 cattle units per hectare, and in sown pastures 1 cattle unit per hectare.

2.3.7 Balance of nitrogen lost and gained due to soil erosion (ΔN_{ero})

The balance of N in soil lost and gained is the difference between the amount of N that mechanically arrives in a plot by soil deposition (gain) and the amount that leaves the plot with eroded soil (loss). We used the universal soil loss equation (RUSLE) (Renard et al. 1991) and collected data for each parameter at the *European Soil Data Centre* (ESDAC) (<http://esdac.jrc.ec.europa.eu/>). We calculated one balance for each land use, using a specific C factor in the RUSLE according to Panagos et al. (2015). In Portugal, this factor for cereals is 0.3520, for vines is 0.3313, for olive trees is 0.2216 and for pastures is 0.1030.

For soil deposition, we followed the approach by Lugato et al. (2015). We assumed that erosion took place in all cells of the map of river basins in Portugal, but deposition only takes place in the 25% of the area of the basin with the lowest elevation. Among those 25% of cells we uniformly distributed 70% of the accumulated soil loss in the basin, and admitted that the other 30% are lost in water courses. Finally, we established the cell-by-cell deposition minus erosion balance, and averaged results per agrarian region for each land use.

2.3.8 Nitrate leaching (N_{leach})

We used several methods and models to calculate nitrate leaching as it is the term with highest uncertainty. First, whenever possible, we used actual field studies with measurements of nitrate loss. We assigned average values of those studies to particular regions and land uses. Limitations of space prevent us from quoting the studies we used here. Note that these are local studies so there is a large error attached in assuming they are valid for whole regions. The second strategy we used was modelling. We used three different models: IPCC (2006), SALCA-NO₃ (Richner et al. 2014) and Paz et al. (2009). All models have limitations. The IPCC model is generic and very simplified. For SALCA we were unable to calibrate the model for Portuguese conditions. Regarding Paz et al. (2009), the model is applicable to the Valenciana region in Spain, so due to the bioclimatic proximity to most regions in Portugal we assumed that it would be representative of the study area. Given these limitations, we did not choose one model over another and present results from the four approaches. Note that for sown biodiverse pastures we assumed in all cases that nitrate leaching was zero (Teixeira et al., 2015).

2.3.9 Emission of NO (N_{NO}), NH₃ (N_{NH3}) and N₂O (N_{N2O}) to the atmosphere

Air emissions of N substances were calculated according to EMEP/EEA (2013) and IPCC (2006), as suggested in *Nutrient Budgets – Methodology and Handbook* (Eurostat 2013). NO emissions were obtained by multiplying the amount of fertilizer applied in each case by an emission factor (0.026 kg NO/kg fertilizer). NH₃ emissions have two distinct origins. First, emissions from fertilizer application were calculated by multiplying amounts of fertilizers by an emission factor (0.081 kg NH₃/kg fertilizer). Second, emissions from grazing were calculated using the amount of excrement already determined previously and multiplying it by an emission factor of 0.06 kg N-NH₃/kg excreted N. Finally, the sources for N₂O emissions are due not only to fertilizer applications and excrements (in

which case the method for calculation is the same as presented for NH₃, but also plant residue and N mineralization in soils. For emissions from plant residue, we used GPG (2001), as recommended by NIR (2014). N mineralization was calculated according to IPCC (2006) or collected from field studies. The calculation involves an estimation of C:N ratios in soils depending on land use. Due to lack of space, we are unable to present all sources involved in this procedure here. Note that in sown pastures there is an additional source of N₂O due to legumes. Following IPCC (2006), we assumed that 1% of N fixed by legumes is re-emitted as N₂O.

3. Results

Results showed that the terms of the balance that weigh the most on the final balance are fertilization, N uptake by the plant and nitrate leaching. We used two methods to determine fertilizer use. Results using data by Morais et al. (2016) (“real” fertilization) is shown in Table 1. Results using best practices cannot be shown here, but in general using fertilization recommendations makes, in general, the balances are lower in absolute value. Leaching is the most uncertain parameter, with results from the methods and models used varying by two orders of magnitude (e.g. comparing the balance obtained for maize using field data and the balance according to Paz et al. (2009), the difference is approximately 110 kg N/ha).

Table 1: Nitrogen balance for all land uses, using data from Morais et al. (2016) as the source for fertilization, and including 3 modelling approach (and their average) and “in situ” data for leaching. Modelling results are shown in grey when “in situ” data is available

Land use	Agrarian region in Portugal	Irrigation	$\frac{\Delta N_{soil}}{\Delta t}$ (kg N/ha)					“In situ” data
			No leaching	With leaching				
				IPCC	SALCA-NO ₃	Paz et al. (2009)	Average	
Common wheat	Alentejo	Rainfed	5.18	5.09	-153.84	-104.16	-84.3	
	Alentejo	Irrigated	5.41	5.31	-155.93	-107.35	-85.99	
	Ribatejo e Oeste	Rainfed	21.55	21.41	-146.61	-98.11	-74.44	
	Ribatejo e Oeste	Irrigated	21.41	21.28	-144.57	-95.54	-72.94	
	Trás-os-Montes	Rainfed	-5.95	-6.04	-166.17	-117.9	-96.7	
Durum wheat	Ribatejo e Oeste	Rainfed	22.57	22.36	-124.68	-113.25	-71.86	
	Alentejo	Rainfed	7.2	6.99	-140.17	-131.93	-88.37	
Grain maize	Alentejo	Irrigated	2.49	1.87	-153.49	-249.1	-133.58	
	Beira Interior	Irrigated	-31.76	-31.84	-116.95	-147.62	-98.8	
	Beira Litoral	Irrigated	9.48	9.04	-118.91	-198.16	-102.68	-13.52
	Entre Douro e Minho	Irrigated	-19.39	-19.55	-115.04	-153.26	-95.95	
	Ribatejo e Oeste	Irrigated	28.77	28.04	-124.16	-255.52	-117.21	
	Trás-os-Montes	Irrigated	-16.81	-16.9	-104.3	-132	-84.4	
	Beira Litoral	Irrigated	80.23	79.18	-137.43	-100.33	-52.86	57.23
Silage maize	Entre Douro e Minho	Irrigated	168.26	166.55	-141.9	-100.16	-25.17	
	Beira Litoral	Rainfed	-4.83	-4.92	-165.97	-120.75	-97.22	-27.83
Oat grain	Alentejo	Rainfed	8.24	8.1	-152.82	-120.45	-88.39	
	Trás-os-Montes	Rainfed	-5.78	-5.84	-159.99	-116.58	-94.14	
	Algarve	Rainfed	-0.69	-0.81	-162.5	-125.15	-96.16	
	Ribatejo e Oeste	Rainfed	19.26	19.24	-131.68	-75.9	-62.78	
	Beira Interior	Rainfed	-21.25	-21.27	-169.92	-119.9	-103.7	
Forage oat	Entre Douro e Minho	Rainfed	64.21	63.15	-293.62	-131.12	-120.53	
	Trás-os-Montes	Rainfed	14.7	14.11	-246.49	-120.16	-117.52	
	Beira Litoral	Rainfed	10.96	10.46	-230.93	-116.38	-112.28	-12.04
	Beira Interior	Rainfed	-6.05	-6.46	-230.33	-127.99	-121.59	
	Alentejo	Rainfed	30.49	29.78	-253.75	-111.95	-111.97	

Land use	Agrarian region in Portugal	Irrigation	$\frac{\Delta N_{soil}}{\Delta t}$ (kg N/ha)					Average	"In situ" data
			No leaching	With leaching					
				IPCC	SALCA-NO3	Paz et al. (2009)			
	Ribatejo e Oeste	Rainfed	45.11	44.43	-234.5	-96.2	-95.42		
Barley	Alentejo	Rainfed	21.41	21.3	-136.84	-93.23	-69.59		
Triticale	Ribatejo e Oeste	Rainfed	19.71	19.64	-137.67	-92.71	-70.25	17.67	
Sunflower	-	-	16.37	16.19	-92.58	-81.67	-52.69		
Table grapes	Alentejo	Irrigated	34.75	34.47	-163.22	-92.85	-73.87		
	Algarve	Irrigated	19.43	19.19	-167.92	-103.63	-84.12		
	Alentejo	Rainfed	0.92	0.82	-147.61	-106.88	-84.56		
Wine grapes	Beira Interior	Rainfed	-19.19	-19.21	-145.56	-116.59	-93.79		
	Beira Litoral	Rainfed	-15.31	-15.35	-147.08	-111.97	-91.47		
	Ribatejo e Oeste	Rainfed	10.91	10.81	-135.31	-92.26	-72.25		
	Trás-os-Montes	Rainfed	-9.56	-9.6	-141.61	-109.6	-86.94		
	Entre Douro e Minho	Rainfed	-13.44	-13.48	-145.45	-109.58	-89.5		
	Alentejo	Rainfed	89.59	89.19	-141.19	-102.01	-51.34		
	Alentejo	Irrigated	85.16	84.77	-140.18	-101.77	-52.4		
	Beira Interior	Rainfed	14.26	14.11	-147.27	-117.79	-83.65		
	Beira Interior	Irrigated	19.11	18.94	-148.36	-118.04	-82.49		
	Beira Litoral	Rainfed	59.51	59.2	-143.97	-37.74	-40.84		
	Beira Litoral	Irrigated	69.37	69.03	-146.2	-109.04	-62.07		
Olives	Entre Douro e Minho	Irrigated	115.55	115.03	-148.79	-104.79	-46.18		
	Ribatejo e Oeste	Rainfed	109.78	109.34	-130.99	-90.27	-37.31		
	Ribatejo e Oeste	Irrigated	70.55	70.29	-122.14	-88.08	-46.65		
	Trás-os-Montes	Rainfed	53.04	52.79	-135.73	-102.43	-61.79		
	Trás-os-Montes	Irrigated	50.8	50.56	-135.23	-102.32	-62.33		
	Beira Interior	Irrigated	-7.09	-7.17	-149.92	-103.97	-87.02		
	Beira Litoral	Irrigated	19.54	19.41	-138.56	-86.78	-68.64		
	Ribatejo e Oeste	Rainfed	83.45	83.16	-119.67	-46.47	-27.66		
Pears	Ribatejo e Oeste	Irrigated	85.56	85.28	-118.41	-46.8	-26.64		
	Trás-os-Montes	Irrigated	12.8	12.7	-137.28	-93.27	-72.62		
	Trás-os-Montes	Rainfed	-9.01	-9.02	-133.41	-109.11	-83.85		
Almond	Algarve	Rainfed	-7.18	-7.18	-130.01	-105.89	-81.03		
	Ribatejo e Oeste	Irrigated	-9.63	-10.05	-123.33	-570.73	-234.71	-41	
Tomato	Alentejo	Irrigated	-24.37	-24.79	-138.23	-593.5	-252.18	-55.74	
	Algarve	Irrigated	-33.46	-33.88	-147.16	-598.44	-259.82		
	Beira Litoral	Irrigated	11.4	11.21	-133.6	-172.32	-98.24	5.39	
Rice	Alentejo	Irrigated	40.24	39.85	-104.76	-194.62	-86.51		
	Ribatejo e Oeste	Irrigated	48.24	47.93	-96.76	-161.26	-70.03	36.7	
	Alentejo	Rainfed	127.03	127.03	127.03	127.03	127.03	127.03	
Sown biodiverse pastures	Beira Interior	Rainfed	119.69	119.69	119.69	119.69	119.69	119.69	
	Ribatejo e Oeste	Rainfed	133.71	133.71	133.71	133.71	133.71	133.71	
Natural pastures	Alentejo	Rainfed	61.42	61.18	61.42	22.38	48.33		
	Ribatejo e Oeste	Rainfed	68.11	67.86	68.11	29.11	55.03		
	Trás-os-Montes	Rainfed	53.23	52.99	53.23	9.85	38.69		

Sown biodiverse pastures are special case, since there is no nitrate leaching. This means that, since the main source of uncertainty in calculations is absent, it is possible to use the data and compare to other sources to obtain some degree of validation of these results. The results for SBPPRL are approximately 0.12 t N/ha. As a simplification, we can assume that all N is organic, and that the C:N ratio is 13. If so, this accumulation of N is equivalent to 1.56 t organic C/ha. Assuming (carbon balance, not shown here) that all the C was obtained from the atmosphere through photosynthesis, and using the ratio between the molar mass of CO₂ and atomic mass of C, this is equivalent to 5.72 t CO₂/ha that are sequestered from the atmosphere by sown biodiverse pastures. This number is similar

to the average C sequestration of this pasture system as obtained by Teixeira et al. (2011) and reported by Portugal in the Kyoto Protocol (APA 2015).

4. Discussion

The method laid out here proposes (1) a general framework for mass balance models in future LCI development (figure 1); (2) a specific mass-balance model that is more thorough than the well established OECD model; (3) an operationalization of said model using specific methods of calculation of each parameter; and (4) an application to several agricultural products/land uses in Portugal and indirect validation. We calculated the soil N balance by quantifying each input and output, which in LCI could be introduced as separate entry/exit flows.

We observed that the most uncertain parameter is nitrate leaching, which is in fact left out of the assessment in the OECD approach. More research is needed before this crucial parameter is quantified sufficiently to be part of LCI as an outflow. Although all methods used in this work for leaching had limitations, it is clear that different models generate very different results and at the moment it is indeterminate which one is more appropriate. If we disregard leaching, the overall balance for N in soils, as validated using sown pastures, seems to be working as intended and presenting expected results. The use of scale-consistent data for fertilizer consumption, a crucial variable, helped keeping the balance in check.

Results also show that crop-level and regional-level modelling is essential. As seen in Table 1, the difference between results for e.g. common wheat in Alentejo and durum wheat in Ribatejo e Oeste is larger than between pears and rice in Beira Litoral. This model can very easily be generalized for calculations that take place not at the agrarian region level, but at the cell level, thus enabling even more geographical discrimination. Regionalization is crucial for accurate decision support particularly in the agri-food sector where location of activities is crucial.

Here we presented only the soil N balance. We also did not perform uncertainty analyses. One way to address variability in results of the combination of many different models is to use Monte Carlo or other iterative statistical methods. However, there is reason to believe that the procedure proposed here is self-consistent and can reduce uncertainty in the establishment of inventory flows once this approach is generalized and C and N sub-balances are determined for all compartments (soil, animal and plant). C and N balances share linkages (e.g. C:N ratios of soil organic pools) that can be established independently of the models used to calculate each flow, thus providing cross-validation or restricting the option space for parameters in each model.

5. Conclusions

Despite an abundance of methods for LCI development, we thus consider that the soil N balance model clearly demonstrates the need for regionalized inventories that are consistent both in terms of mass and scale. Our results showed that both product and scale discrimination is essential in results, as there are no significant trends that would enable an aggregation of results into larger geographical scales or product types. The results are limited, however, by the quality of models available to estimate nitrate leaching, whose uncertainty can change final balances dramatically.

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P27. Sustainability Assessment of Apple Value Chains: Mixed method approach towards Urban Transition

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ABSTRACT

The food sector will face numerous challenges in the next decades, arising from changing global production and consumption patterns, which currently go along with high resource use and further ecological and socio-economic impacts. Niche developments that focus on establishing sustainable behaviour patterns might be able to contribute to transforming society and economy towards a raised level of sustainability. Therefore special emphasis is put on the application for the specific food transition initiative "mundraub", which aims at avoiding the identified hot spots in the food sector by providing an online interactive map of e.g. unused apple trees in urban areas. The analysis is based on a mixed method approach comprising three different methodologies addressing sustainability aspects of apple value chains. The work is based on a sustainability hot spot analysis assessment with relevance for the German apple consumption focusing on conventional, organic apples, and apples of commons like orchards and urban areas. The ecological LCA comprised calculating the Material and Carbon Footprint of conventional German and New Zealand apples. Additionally results of an online survey among members of the mundraub initiative were included. The results show ecological and social impacts mainly attributed to conventional and organic apple value chains due to storage and transport effort. The Material Footprint calendar summarized the effects throughout the year. The mundraub initiative can indirectly contribute to avoid hot spots in conventional and organic apple value chains and directly contribute to the sustainability of orchards.

Keywords: Hot Spot Analysis, food chain, LCA, value chain, environmental and social impact

1. Introduction

Nutrition is one of the key topics when it comes to future societal challenges (Hahlbrock 2007, EEA 2013, UBA 2015a, Lukas et al. 2016). To succeed in transforming society and economy towards a raised level of sustainability, it is crucial to look at niche developments that focus on establishing sustainable behaviour patterns in complex social systems (Shove 2012). Learning processes that are based on these novel developments represent the foundation of successfully implementing innovative practices in society (UBA 2015b, WBGU 2011, 2016).

The German web-based initiative Mundraub represents such a niche development, which aims at showing how peri-urban food production and the use of urban grown food can contribute to sustainable development by pushing the utilization of currently unused sources of fruit (and others). Thus, Mundraub established an online map that lists fruit trees and bushes, and locations of nuts and herbs in the public sphere. To assess its possible contribution towards sustainable development we assessed ecological and social impacts of apple value chains. The apple serves as a fruit example due to high relevance in Germany (highest per capita consumption among fruits) and the high relevance for the mundraub initiative (harvesting and tree care camps) (WI & mundraub 2015a, 2015b).

The paper firstly presents the mixed method approach that has been used to identify key ecological and social impacts of three apple value chains (conventional, organic, orchards). Secondly the results are illustrated to finally draw conclusions, which of the identified hot spots could be avoided by the mundraub initiative.

2. Methods, goal and scope

The sustainability life cycle assessment has been conducted as mixed method approach combining the methodology of the Sustainability Hot Spot Analysis (SHSA) and ecological life cycle analysis calculating the Material Footprint and Carbon Footprint of selected apple value chains. Results have

been used to conclude on pros and cons of the mundraub initiative also based on an online survey among mundraub users. Figure 1 depicts the mixed method approach.

The **Hot Spot Analysis** methodology has been developed at the Wuppertal Institut in 2002 (for further information on method development see Liedtke et al. 2010 and Bienge et al. 2010). The method has been applied to several value chains and in the context of supply chain management in companies (Liedtke et al. 2010, Rohn et al. 2014, Geibler et al. 2016).

The main objective of the Sustainability HSA is to identify key environmental and social impacts along the whole value chain. The HSA itself combines the methods literature review, category-based systematic clustering of fact-based information, semi-quantitative relevance assessment, and stakeholder evaluation. Thus, the assessment is based on desk research (scientific literature, further fact-based information e.g. reports, media) and expert knowledge (e.g., company, sector, NGO, trade unions, federations, consumer associations, experts).

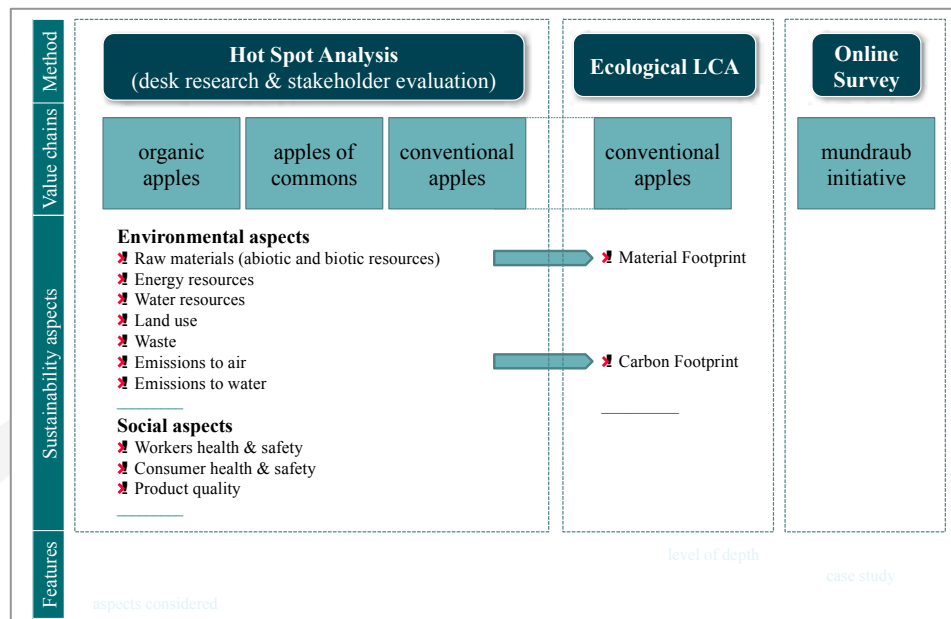


Figure 1: Mixed method approach for apple value chain assessment

The SHSA has a five-step approach described in Bienge et al. (2010). Starting with defining the life cycle phases (e.g. raw material extraction / cultivation, processing, trade, transport, packaging, use and waste disposal) and the categories. Table 1 shows the main environmental and social categories that need to be considered along the product or service life cycle. The aspects and descriptions are derived from international standards GRI and UNEP SETAC Life Cycle Initiative (Bienge et al. 2010 based on UNEP/SETAC 2009, GRI 2011). This paper focuses on all environmental categories ([1]-[8]) and a selection of three social aspects ([4], [7], [8]).

Table 1: Environmental and social aspects

Environmental aspect	Social aspects
[1] Abiotic resources: materials taken directly from nature, not renewable in hundreds of years, e.g. ores in a mine; products e.g. pesticides, fertilizers	[1] General working conditions: e.g. working hours, legal contracts, illegal workers, general working conditions
[2] Biotic resources: all organic materials taken directly from nature, before processing, (e.g. grass, trees, fish, fruits, cotton); organic products e.g. fertilizers	[2] Social security: e.g. contracts and obligatory social security provisions
[3] Energy resources: Energy used in terms of electricity and fuel	[3] Training & Education: education on their rights as employers and also training on working with hazardous materials
[4] Water resources: amount of water used in cultivation, cleaning during production	[4] Workers health & safety: e.g. occupational health and safety (hygienic working conditions)
[5] Land use: amount of land used, biodiversity, soil degradation	[5] Human rights: e.g. child labour, equal pay/benefits/opportunities between workers, forced labour (harsh and inhumane treatment), freedom of association, sexual harassment

[6] Waste: all excess solid wastes	[6] Living wages: Minimum wage / Living wage
[7] Emissions to air: chemicals released to air, GHG emissions through electricity usage and transport, other sources such as livestock	[7] Consumer health & safety: e.g. health standards of product, product safety, information and transparency regarding health issues (allergens, nutritional value), warnings, declaration of control mechanisms for health and safety
[8] Emissions to water: emissions to water such as nutrients and chemicals used for cultivation, detergents during the use/consumption phase	[8] Product quality: e.g. longevity, practicability (quantities / portion of product offered, safe packaging), transparency and information (reliable, adequate, voluntary), added value for society

Steps 2 and 3 are key elements of the assessment. Based on the above-mentioned desk research and identified impacts each aspect's relevance is assessed by a scale from 1 to 3 (1 means a low relevance and 3 means a high relevance). Same scale is used for weighting the relevance of the life cycle phases. The hot spots are determined by multiplying the value of the aspect and the value of the phase (step 4). If no information is available we applied "0". Hot spots are defined as relevance of 6 or 9. Step 5 comprises the critical review of the results in terms of scope, quality and completeness of identified information, and relevance assessment with the help of relevant stakeholders.

The **ecological LCA** is based on the calculation of the Material Footprint (based on the MIPS concept) and the Carbon Footprint. The ecological assessment serves as a more detailed analysis of the aspect abiotic and biotic raw materials and emissions to air, which are also addressed by the HSA.

Therefore we perform a material intensity analysis (MAIA) (Bringezu et al. 2003, Liedtke et al. 2014, Wiesen et al. 2014) calculating the Material Footprint of conventional apple value chains in Germany and New Zealand. The Material Footprint is an indicator for material use that additionally considers economically unused resource extraction (e.g. tailings from mining, excavated soil during the construction of infrastructure or loss of land through erosion). The MAIA is based on the MIPS concept (Material Input per Service Unit) that follows the logic that each resource extraction reveals impacts connected to the mass of extracted material, such as lowering the groundwater table, translocation of fertile soil or landscape changes (Liedtke et al. 2014). In order to consider the additional resource flows, we enhance the selected datasets in Ecoinvent, and apply MF as a further impact indicator within OpenLCA (Saurat and Ritthoff 2013, Wiesen et al. 2014). Finally, we compare the value chains, using the global warming potential (GWP) as an indicator of the GHG emissions.

Besides the HSA and Footprint results, selected results of an **online survey** amongst mundraub users have been used to discuss (avoided) social and ecological impacts of the mundraub initiative. The online survey has been conducted in September 2015. Out of 356 requests 200 datasets have been completed. Taking into account the range of assumed 20,000 addressed users (questionnaire dissemination via newsletter and social media) the response rate is about 1%. The questionnaire consists of 21 questions. Relevant for the mixed method approach are questions on waste, distance to fruit collecting, means of transport, number of tours and size of the group, context of collection.

3. Results

Hot Spot Analysis

We defined a mix of countries that cultivate apples relevant for the German apple consumption. They have been included into the desk research representing the most relevant countries. Due to annual fluctuation of production amounts the country relevance has been checked comparing several years (depending on data availability).

The following countries and regions are relevant for the conventional apple cultivation: Germany with an annual average of 0.94 million tonnes 2002-2014 (own calculation based on Statistisches Bundesamt 2015) and imported apples from EU-27 (Italy, Netherlands, Belgium, France, Austria, Czech Republic) and non-EU imports from New Zealand (together 0.67 million tonnes in 2011, fluctuations in 2002-2011 have been analysed based on FAOSTAT database). Thus, the scope includes 95% of imported apples.

The following countries and regions are relevant for the organic apple cultivation: Germany, Italy, Austria, New Zealand, and Argentina. There is only limited market data available (Schaack et al.

2011). However, we assume these four countries accounting for 90-92% of the imported organic apples (based on 2009 and 2010, BÖLW 2012).

The orchard cultivation and apple tree cultivation in urban areas is focused on Germany.

This step also includes the selection of ecological and social aspects (see Table 1).

Table 2 shows the identified hot spot based on significance assessment of aspects and life cycle phases. The hot spots appear above all in the cultivation in the conventional apple value chain. Most relevant are the use of abiotic resources (high amounts of pesticides and fertilizers, high degree of mechanization) and the land use aspect (low agrobiodiversity, monocultures, effects of chemical application). Energy use and emissions to air and water are also relevant (transport and storage dependent on cultivation region, month of harvest, duration of storage; energy use for chemical production, leaching of chemicals into waterbody). Although in the organic cultivation there is no hot spot we identified ecological impacts. That is why organic cultivation is similar to conventional cultivation in terms of high degree of mechanization, monocultures, storage and transport. The use of plant protection is clearly distinctive from that of conventional cultivation (and thus the lower impact). However the use of copper in the organic cultivation is a crucial problem. For apple cultivation in orchards and urban spaces no hot spots have been identified.

The aspects product quality and workers health & safety in conventional cultivation have been identified as hot spots (application of agrochemicals, lack of knowledge on active substance mixes in correlation with current approval mechanisms). In the trade & use phase we identified no hot spots, but regarding product quality and consumer health & safety evident research gaps have been identified (residue control only searches for limited list of harmful substances, unknown long-term effects on health). The orchard and urban value chain has only few effects. One is the declining number and quality of orchards (wrong management, little economic benefit) and the other is the unknown management and missing quality assessment of private grown apples (not traded).

Table 2: Hot Spots of apple value chains (hot spots in bold)

Life cycle phase	Cultivation			Trade & use		
	Conventional	Organic	Orchards & urban	Conventional	Organic	Orchards & urban
Value chain						
Relevance of phases	3	2	2	2	1	1
Hot Spots						
Abiotic resources	9	4	2	2	1	1
Biotic resources	3	2	2	0	0	0
Energy resources	6	4	0	4	2	2
Water resources	3	0	0	0	0	0
Land use	9	4	4	0	0	0
Waste	3	4	0	0	0	0
Emissions to air	6	2	0	4	2	2
Emissions to water	6	0	0	0	0	0
Workers health & safety	6	2	0	0	0	0
Consumer health & safety	0	0	0	4	1	1
Product quality	6	2	4	4	1	1

The stakeholder workshop (June 2015) basically confirmed the results of the desk research. The experts provided valuable hints for further relevant studies and pointed out that the ecological impact would need detailed analysis (month, region). Especially the topics impact of and knowledge about agrochemicals have been discussed. Crucial to the HSA method is a transparent documentation of the data quality and research gaps. The results of the workshop have been used to revise the HSA (steps 2 to 4). Table 2 shows the final results including the stakeholder evaluation.

Ecological LCA

The ecological life cycle analysis aimed at assessing the Material and Carbon Footprint of conventional apples cultivated in Germany (Meckenheim in North Rhine Westphalia) and New Zealand (Nelson), traded and consumed in Germany. The functional unit is 1kg fresh apples. Results are differentiated by month of harvest and consumption (over the year). The analysis includes cultivation, storage incl. post-harvest losses, transport and cooling. Basic assumptions are summarized in Table 3.

The yield per hectare varies considerably between years and regions. For the German cultivation in NRW we assume 33,11 t/ha. The national average yield is 29.8 t/ha considering the years 2003-2013 (FAOSTAT). They are not valid for the region NRW showing higher yields (2011: 18%, 2012: 4.2%, Statistisches Bundesamt 2012). Thus, we assumed an 11% higher yield than the national average. In New Zealand we assumed the yield in Nelson equals the national average of pomes in 2008-2012 of 53,65 t/ha (based on Ministry for Primary Industries 2012: data is given in Tray carton equivalent defined as 18.6 kg package weight). Hence, the yield in New Zealand is about 60% higher than in Germany.

Table 3: Basic assumptions

	Germany (NRW)	Source	New Zealand (Nelson)	Source
Months of harvest	August - November		March - May (available in Germany one month later after transport)	
Yield	33.11 t/ha	[1]	53.65 t/ha	[2]
Fertilizer	Nitrogen 0.0018 kg/kg apple Phosphor 0.0008 kg/kg apple Potassium 0.0023 kg/kg apple	[3] [4]	Nitrogen 0.0009 kg/kg apple Phosphor 0.0002 kg/kg apple Potassium 0.0013 kg/kg apple	[5]
Pesticides	Fungicides 0.00049 kg/kg apple Herbicides 0.00005 kg/kg apple Insecticides 0.00008 kg/kg apple	[6]	Fungicides 0.00034 kg/kg apple Herbicides 0.00015 kg/kg apple Insecticides 0.00012 kg/kg apple	[7]
Cultivation	Tractor (Diesel: 0.0436 l/m ²)	[12]	Tractor (Diesel: 0.0436 l/m ²)	[12]
Post-harvest losses	11% in CA-storage	[8]	11% in CA-storage	[8]
Initial Cooling	0.086 MJ/kg apple (electricity, low voltage, Germany)	[10]	0.086 MJ/kg apple (electricity, mix New Zealand)	[10]
CA-storage	0.0054 MJ/d/kg apple (electricity, low voltage, Germany)	[10]	0.0054 MJ/d/kg apple (electricity, low voltage, Germany)	[10]
Transport - shipping	none		23.000 km cargo ship incl. use of the port and heavy fuel (0.0025 kg/t*km + 0.0003 kg/t*km for cooling), one way, CA-storage on board increases heavy fuel use by 14%	[10]
Transport - lorry	Transport to sorting: 10 km one way; 20 km return (lorry 16-32 t) Transport to wholesale: 20 km one way; 40 km return (lorry >32 t) Transport to supermarket: 150 km one way (lorry >32 t)	[10] [11]	Transport to port: 20 km one way; 40 km return (lorry 16-32 t) Shipping 23,000 km, one way Transport to regional storage: 200 km one way (lorry >32 t) Transport to supermarket: 150 km one way (lorry >32 t)	[10] [11]
Transport - car	6 passenger km (3 pkm one way, 6 pkm return), 20 kg apples	[10]	6 passenger km (3 pkm one way, 6 pkm return), 20 kg apples	[10]

Sources: [1] Own calculation based on auf FAOSTAT 2015 and Statistisches Bundesamt 2012, [2] Own calculation based on Ministry for Primary Industries 2012, [3] Ryser et al. 2003, [4] BMLFUW 2008, [5] Palmer 2012 [6] EC 2007, [7] Holland & Rahman 1999, [8] TI / MRI / JKI 2013 - assumed for both regions, [10] Blanke & Burdick 2005, [11] Schröder 2007, [12] Frischknecht et al. 2005

The post-harvest losses are assumed with 11% during CA-storage (based on TI / MRI / JKI 2013). Apples that are not directly sold are usually stored under CA. In Germany this can take up to 9 months after harvest. New Zealand apples are CA-stored on the ship and can similarly to German apples be CA-stored in Germany.

Transport assumptions are described in Table 2. Transports by lorry are partly refrigerated. However, the increased fuel use (diesel) for the cooling is very low (own calculation based on Blancke & Burdick 2005: 0.028 MJ/kg for 95 km results in about 0.00084 l/kg apple) and has not been included.

After harvest and grading the apples are cooled (initial cooling). In Germany we assume a following CA-storage (controlled atmosphere with low oxygen and high carbon dioxide and low temperature of 1°C to slow down the ripening process). In New Zealand the apples are transported to the port in Nelson. The shipping takes 23,000 km and 28 days to Antwerp (incl. CA-storage). They are then transported to the wholesale in Germany. Also the German apples are transported to the wholesale. From here two further transports are assumed (to supermarket and at home).

The electricity for cooling processes is calculated with country specific electricity mix. However, there is no specific New Zealand mix available in ecoinvent. Therefore the German electricity mix has been adopted based on Ministry for Business, Innovation & Employment (2015). The main energy sources in New Zealand are hydro power (55%), natural gas (16%), geothermal (14%), wood energy (6%) and wind power (5%). Hard coal and lignite power plants have a little share (4%).

Table 4 and 5 show the assumptions for fertilizer and pesticide use in Germany and New Zealand.

Table 4: Fertilizer use

Region	Yield	N	P ₂ O ₅	K ₂ O	Comment
Germany	<40 t/ha	70 kg/ha	35 kg/ha	90 kg/ha	BMLFUW 2008
	30 t/ha	50 kg/ha	15 kg/ha	60 kg/ha	Ryser et al. 2003
		60 kg/ha	25 kg/ha	75 kg/ha	Average of Ryser et al. 2003, BMLFUW 2008
	33.11 t/ha	0.0018 kg/kg	0.0008 kg/kg	0.0023 kg/kg	Scaling to yield in NRW
New Zealand		50 kg/ha	13 kg/ha	70 kg/ha	Palmer 2012
	53.65 t/ha	0.0009 kg/kg	0.0002 kg/kg	0.0013 kg/kg	Scaling to yield in Nelson

Table 5: Pesticide use

Region	Pesticide	Fungicides	Herbicides	Insecticides	Comment
Germany	Sulphur	9.5 kg/ha	-	-	- based on EC 2007: source provides
	Mancozeb	4.3 kg/ha	-	-	- no data for Tolyfluanid and
	Tolyfluanid	1.7 kg/ha	-	-	- Glyphosat (confidential), amounts
	Metiram	0.6 kg/ha	-	-	- calculated (Total minus given data
	Glyphosat	-	1.7 kg/ha	-	- equals 3.3 kg/ha: divided to both
	Other	-	-	-	- pesticides); Others assumed as
	Total			20.5 kg/ha	2.7 kg/ha
		0.00049 kg/kg	0.00005 kg/kg	0.00008 kg/kg	Scaling to yield in NRW
New Zealand	Canterbury	15,2 kg/ha	7,3 kg/ha	4,3 kg/ha	
	Hawkes Bay	18,5 kg/ha	8 kg/ha	9 kg/ha	Average based on Holland &
	Waikato	21,2 kg/ha	8,4 kg/ha	6 kg/ha	Rahman 1999
	Average	18,3 kg/ha	7,9 kg/ha	6,43 kg/ha	
		0.00034 kg/kg	0.00015 kg/kg	0.00012 kg/kg	Scaling to yield in Nelson

Figure 2 and Figure 3 show selected results for apples that are not stored in Germany and ones that are stored for 3 months. Both, the Material and Carbon Footprint of not stored and stored New Zealand apples are higher than German apples. This is remarkable due to the higher yield per hectare and the lower Material Footprint of electricity in New Zealand. This shows the importance of the shipping. Further the duration of storage is important for both regions leading to increasing Footprints generally.

The initial cooling (25%), transports to wholesale and supermarket (29%), and transport by car (29%) dominate the abiotic Material Footprint of not stored German apples. The cultivation has a share of 13%. The CA-storage has a significant influence on the results depicted for a 3 month stored apple. One can see a share of 58%. Thus the shares of initial cooling (11%), transports to wholesale and supermarket (11%), transport by car (12%), and cultivation (6%) are respectively lower.

There are similar results for New Zealand apples. The abiotic Material Footprint of not stored apples shows a share of 42% for shipping, 29% for transports to wholesale and supermarket, 16% for transport by car, and 7% for cultivation. The CA-storage has a significant influence on the results depicted for a 3 month stored apple. One can see a share of 42%. Thus the shares of shipping (25%),

initial cooling (2%), transports to wholesale and supermarket (16%), transport by car (9%), and cultivation (4%) is respectively lower.

The biotic Material Footprint increases due to the post-harvest losses.

The Material Footprint (biotic and abiotic) of not stored apples is 1.47 kg/kg German apples and 1.86 kg/kg New Zealand apples. The MF of 3 months stored apples is 2.27 kg/kg German apples and 2.69 kg/kg New Zealand apples.

The Carbon Footprint of not stored apples is 0.15 kg CO_{2eq}/kg German apples and 0.43 kg CO_{2eq}/kg New Zealand apples. The MF of 3 months stored apples is 0.25 kg CO_{2eq}/kg German apples and 0.56 kg CO_{2eq}/kg New Zealand apples.

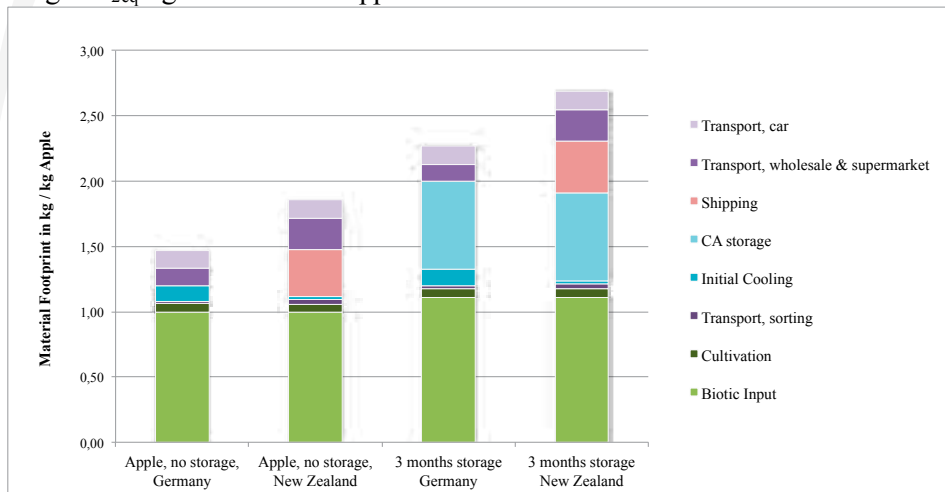


Figure 2: Material Footprint of German and New Zealand conventional apple value chain (no storage in Germany, 3 month storage)

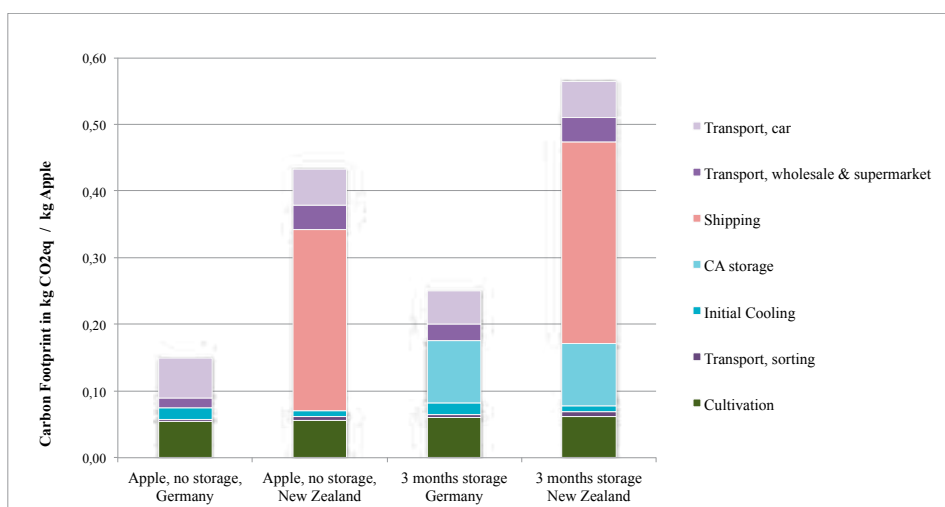


Figure 3: Carbon Footprint (GWP 100) of German and New Zealand conventional apple value chain (no storage in Germany, 3 month storage)

The Online Survey among mundraub participants shows that apple is one of the top 3 products collected (together with berries and nuts). 23.5% of the respondents harvested apples. Most of the overall harvest has been eaten directly (70.5%) or processed to e.g. marmalade (55.5%) or juice (23%) (multiple answers). Two third of the respondents do not throw away (parts of) the harvest. Some do because of the quality, a few said that they harvested too much. Most of the respondents drive less than 10 km to harvest. Only 2% drive more than 30 km. Most of the respondents use the bicycle (61%) or walk (54.5%). 37% drive with their own car and 20% use public transport (multiple answers). If they drive by car they usually do not drive alone. The motivation for participation is basically to harvest fresh and regional fruits (79%), enjoy nature (60%), environmental protection

(49%), conservation of local orchards (45.5%), and saving money (40.5%). Additionally the mundraub initiative has mostly access to orchards and (peri)urban apple trees.

4. Discussion

The mixed method approach has been valuable to first get a broad understanding of key ecological and social impacts of several apple value chains (HSA). Especially the stakeholder workshop was important to evaluate the HSA results. The advices to bring up more detailed analysis of ecological impacts (LCA) and point out problems with agrochemicals and data quality have been addressed. The online survey showed the matching of results to the mundraub initiative.

Thus the above-presented ecological LCA results for specific durations of storage have been translated into impacts throughout the year. Figure 4 shows the impact development for German and New Zealand apples for the Material Footprint. Orange data points depict German apples starting with the harvest season in September to November. Each of the three harvest dates is then added up with additional storage up to 9 months (e.g. data point in December for a German apple harvested in September represents and 3 months stored apple). The same pattern has been applied to New Zealand apples (blue data points) with a harvest season from March to May (availability on German market one month later).

The Figure also shows which of the apple has a lower impact compared to each other on a monthly basis (darker points). This calendar shows best results for seasonal German and New Zealand apples. From September to March German apples and from April to August New Zealand apples have a lower impact. The Carbon Footprint shows the same results.

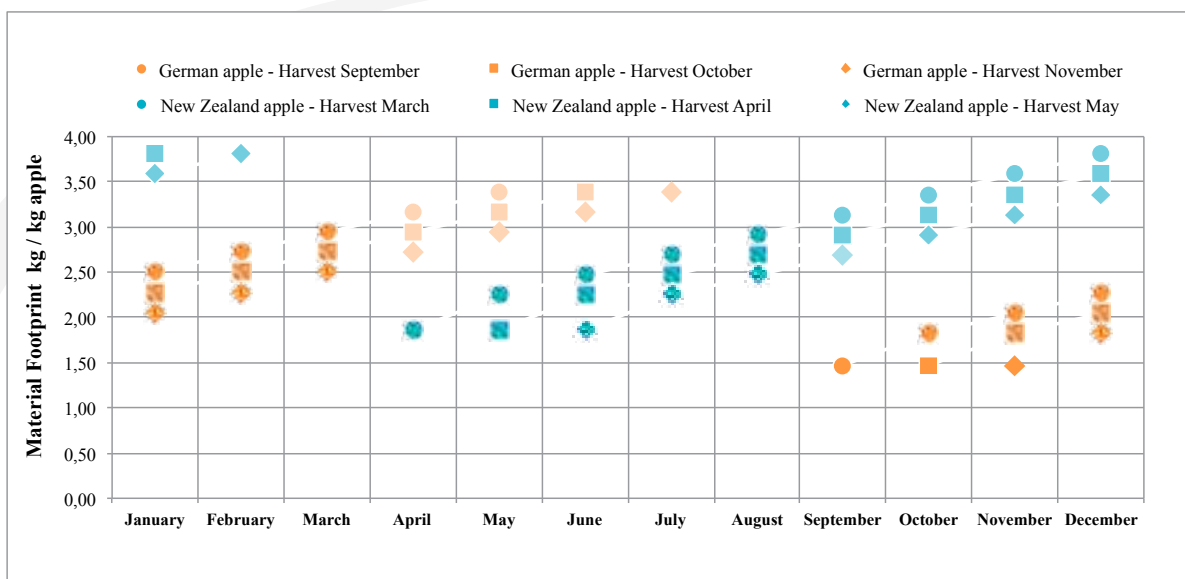


Figure 4: Material Footprint Calendar of German and New Zealand conventional apple value chain

The ecological LCA did not specifically show results for organic apple value chain and apples from orchards and (peri)urban areas. We assume lower impacts for the organic apple value chain due to lower abiotic resource use and energy use (only agrochemicals). Even a lower impact of orchards is assumable due to low grade of mechanisation, usually no storage, and few transport. The impact of urban apples also might be lower although there is lack of information on the use of agrochemicals.

5. Conclusions

The results show that the mundraub initiative can indirectly contribute to avoid hot spots in conventional and organic apple value chains. Due to the fact that this niche development access orchards and (peri)urban areas there is little direct improvement in other value chains. The online survey shows that people actively attending the initiative are motivated by harvest fresh and regional fruits, enjoy nature, protect the environmental or help to conserve local orchards. Therefore they directly avoid some of the ecological and social effects in the orchard value chain (conservation of orchards) unless they improve the management of orchards (e.g. expert support needed for good tree

care). The general environmental motivation might allow the conclusion that participants have a preference for seasonal and regional fruits in general.

The applied mixed method approach itself revealed the need for further research. Especially the HSA showed little country specific data and data gaps and insecurities for the impacts of agrochemicals. Also the selection of three social aspects might hide social hot spots in the other five aspects. The ecological LCA provides a good assumption for selected apple value chains, but could be enhanced with analyses on further countries, cultivation types, storage data and means of transport. Further research is needed on the effects of niche developments and changing individual social practices in food consumption in general.

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P28. The Missing Link: Environmental Impact of Food Distribution

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Post-processing paths of food include distribution-retail center networks and extend to the end consumer. Current life cycle assessment (LCA) research often does not account for the environmental impact of food distribution including distribution and retail centers. In the United States, the distribution-retail networks are complex systems that depend on location including climate and regional fuel mixes for electricity production. In addition, moving and storing foods is a dynamic economic process, which affects allocation of environmental impact to particular food items. This research is a missing link to environmental impact of food distribution, which will bridge a data gap between food processors and consumers.

We used the National Renewable Energy Laboratory (NREL) EnergyPlus models for warehouses and supermarkets for different locations and year of construction to calculate energy requirements of the buildings. Athena was used to calculate environmental impact of building construction, material, and demolition. Both EnergyPlus and Athena results were used to populate life cycle inventory in the Simapro LCA models. The impact results were calculated using the TRACI 2.1. characterization factors. Results from Athena showed that material manufacturing for building construction has the highest environmental impact in a building construction to demolition LCA, but the full LCA showed the building operation has overall highest environmental impact in most impact categories, except for carcinogenic impact, which is attributed to building material with 98% and 70% for ambient and refrigerated buildings, respectively. Climate change impact is driven by heating (50%) and cooling (90%) for ambient and refrigerated areas, respectively.

P29. A Data-Driven Approach to Address Life-Cycle Inventory Data Gaps for the Quantification of Farm-Specific Greenhouse Gas Emissions on a Global Scale: A Case Study on Open-Field Tomato Production

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Greenhouse gas footprinting has been increasingly used to support strategies for sustainable sourcing and reduction of environmental impacts within agricultural supply chains. However, it is often challenging and costly to collect farm or location-specific inventory data. This renders the use of proxy or extrapolated data sets to bridge data gaps, often without quantifying the associated uncertainty introduced. The goal of this study was to develop a data-driven regression model to quantify greenhouse gas emissions of farms with limited data availability on a global scale, using open-field tomato production as a case study. Our framework hypothesizes that farm-specific greenhouse gas emissions are related to so-called predictors such as location of farm (eco-region), farm management practice, gross domestic product of country of farm residence, soil conditions. We quantified the farm-specific greenhouse gas footprints of over 1000 farms spanning 16 countries from 2013 to 2015. Regression models, along with their associated uncertainties, were calibrated and optimized with increasing number of predictors. We started with predictors representing data that are commonly available and moved on to data that require larger collection efforts. The novelty of the research lies in the use of a large amount of data covering spatial, temporal and technological variability for model-building. The models provide an alternative to full life-cycle assessment using proxies or non-site-specific extrapolated data and enable LCA practitioners to minimize data collection efforts according to the tolerated uncertainty for the application of the results.

P30. The Good, the Bad and the Ugly in Collecting Farm Survey Data for Agricultural Life Cycle Assessments

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Abstracts

Collecting farm survey data for agricultural life cycle assessments (LCAs) is critical to develop a reliable and representative life cycle inventory data. However, it is a challenge to collect a large number of farm samples that will satisfy data requirement of reliability and representativeness because of high variations in data resulted from a complexity of agricultural production systems and other variable factors including climate condition, soil type, farming system and management practices. In addition, data collection is one of most time-consuming process in LCA.

Alberta Agriculture and Forestry (AF) commissioned Quantis Canada to conduct LCA of chicken, egg, canola and potato production systems in Alberta using farm survey data. The main purpose of conducting LCA is to inform producers of potential environmental impacts of their production, as well as identify potential improvement areas and mitigation strategies. The AF project team and Quantis Canada collaborated with four commodity association groups in collecting farm survey data for 2012 production year to quantify potential environmental impacts of production in Alberta.

Survey results confirmed that the number of sample size varied depending on commodity groups, ranging from about 1% to 28% of total production in Alberta. The level of data representativeness was likely to be higher in a commodity association group which had business strategies to integrate sustainability into their operational plans. The results also identified a variety of technology, cropping system and management practices in Alberta agricultural production system. How to increase the number of sample size in farm survey data collection remains a main challenge. Survey pretesting and piloting is identified as the best practice for farm data collection process because it can help to remove irrelevant questions to participants and to make questions clear enough to attract more participants in the survey.

P31. How to manage eco-quali-conception concept based on LCA analysis in the wine sector ?

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The French wine sector is subject to increasing regulatory and social pressures concerning environmental issues. Furthermore, there is an imperative about quality and typicality of wine through PDOs (Protected Designation of Origin), PGIs (Protected Geographical Indication) and consumers' expectations. Winegrowers are questioning their technical choices and thinking about possible changes in their practices in order to improve their environmental performances. But viticultural practices are the only way they can manage grape quality to achieve the predefined wine quality goal. The objective of this PhD work is to create and develop a new approach (named "eco-quali-conception[©]") which integrates quality and environmental evaluations in the improvement process.

LCA results of some case studies will be presented to different actors (winegrowers and technicians) of the wine sector in the Loire Valley. Meetings will then be organized in order to design new technical management routes (TMR) based on co-design approach. New TMRs will be created by identifying practices with the highest environmental impacts, propose solutions to lower impacts of these practices and questioning the impacts of these solutions on grape quality and on other practices of the TMR.

This approach will achieve two objectives: improving environmental performances while achieving the desired grape quality. A second expected result is the identification of the current best viticultural practices regarding environmental and grape quality performances.

First results show that there are relatively high differences between TMRs concerning their environmental impacts. These differences depend on choices of equipment, viticultural inputs (pesticides, fertilizer,...) and climatic conditions.

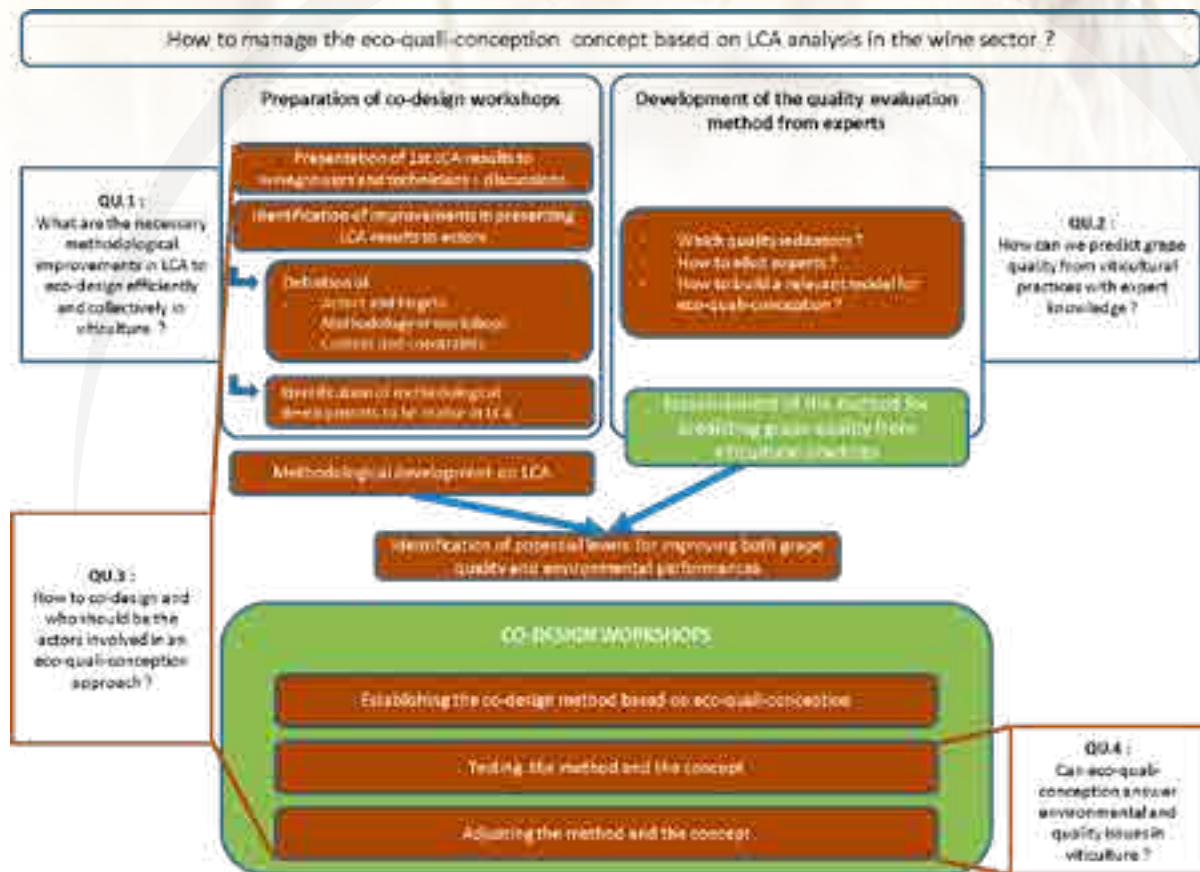


Figure 1: Diagram of the eco-quali-conception concept and its challenges to be answered

POSTER PAPERS

SOIL, CARBON AND PESTICIDES

5. Soil, Carbon and pesticides

P32. Soil quality in LCA: erosion impact assessment for French grapevine production. Application to Muscadet and Carbernet-Franc vine plots.

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Purpose: Soil quality assessment is still an ongoing difficult issue in LCA. Soil quality indicators have been developed for agricultural LCA: erosion, organic matter change or compaction (Garrigues et al 2012). Vineyards soils may be very sensitive to erosion due to topography, soil type, climate and soil management practices. In the purpose of completing viticulture LCA methodological framework, the objective of the study was to develop an erosion indicator for grapevine production in France in LCA context, sensitive to the different erosion factors cited above. It was tested by an application to ten plots of Muscadet and Carbernet-Franc vineyard in the Loire Valley.

Method: The method was established following several steps: (1) determining the dominant soils according to the different wine production region in France (2) description of six viticulture specific agricultural practices mainly based on weed management and (3) erosion simulation with RUSLE2 model. The method was tested on 10 vine plots with different soil conditions and soil managements. The functional unit is hectare.

Results: The method combines 14 French viticulture region X 6 agricultural practices X 6 planting densities X 2 planting conditions. The data are implemented in the Rusle2 model. Assessment routine has been tested for the 10 different situations cited above. The result of soil loss range is between 0.1 to 2 t/ha/year with the average slope steepness between 0.8 to 10%.

Conclusions: LCA practitioners can assess erosion impact of vineyard production with a method as simple and easy to use as possible, keeping the simulation of the complex processes involved in erosion phenomenon, including the soil management practices and soil type and conformation.

Reference:

Garrigues E, Corson MS, Walter C, Angers DA, Van der Werf H (2012) Soil-quality indicators in LCA: method presentation with a case study. In: Corson MS, van der Werf HMG (ed) Proceedings of the 8th International Conference on Life Cycle Assessment in the Agri-Food Sector, Saint Malo, pp 347-352.

P33. Soil organic matter change and crop type affect the sustainability of dryland crop rotations: A case study using emergy and energy analyses

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We analyzed the emergy and energy flows for the semiarid region of Canada based on an 8-yr field experiment involving six 3-yr cropping rotations having durum wheat (*Triticum turgidum* var *durum* L.) in combination with bare fallow, pulse (legume), and oilseed crops. The change in soil organic matter (SOM) was measured.

Crop diversification was positive as durum in rotation with either a pulse or oilseed crop had emergy (analysis with inputs expressed in solar energy equivalents) and energy (analysis of external energy inputs and the energy of produced outputs) indices that indicated better sustainability than monoculture durum.

All the rotations that included fallow lost SOM and so are decidedly not locally sustainable because of loss of soil productive capital. SOM is typically neglected in analyses although its loss releases nitrogen (N) for crops that is an emergy and energy input. Including SOM loss and gain affected rankings greatly and provided more valid assessments of relative sustainability based on emergy and energy. Rotations with pulse crops and without fallow had best relative sustainability based on emergy and energy performance. N inputs was important portion of inputs, explaining why SOM change and pulse crops were important.

Correct emergy and energy performance require including SOM change.

P34. Toxicity: no systematic differences in trace metals concentrations in manure of organic farming compared to manure from conventional farming

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Comparison between production systems like conventional versus organic farming requires adapted reference values. At the moment, data are missing considering trace metals concentrations in manures of organic farming systems (Van Stappen et al., 2015) that contribute to toxicity impacts of such production system ([Prasuhn, 2006](#)).

Twenty-two samples of organic manures from dairy and beef cattle were analysed for cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), nickel (Ni), zinc (Zn) and compared (t-test after log transformation) to forty-eight samples from conventional farming systems according to their type (slurry and farmyard manure) and their origin (dairy or beef systems; table 1). Human toxicity (total of cancer and non cancer, agricultural soils) and Ecotoxicity (soils, unspecified) using to Usetox® (V1.04) characterisation factors were used to estimate toxicity potentials.

No systematic differences between organic and conventional manures were observed even if differences were detected for some trace metals (table 1), *i.e.* at least for Zn concentration in dairy slurry that were higher in conventional systems. Zn and Cu were the main contributors to toxicity potentials (figure 1) but not well characterized due to lack of characterisation factors for human toxicity-cancer that were set to 0 for those two trace metals.

Additional information on mercury and characterisation factors of Cu and Zn are required for a more accurate comparison but some differences may exist between conventional and organic manure for dairy slurry.

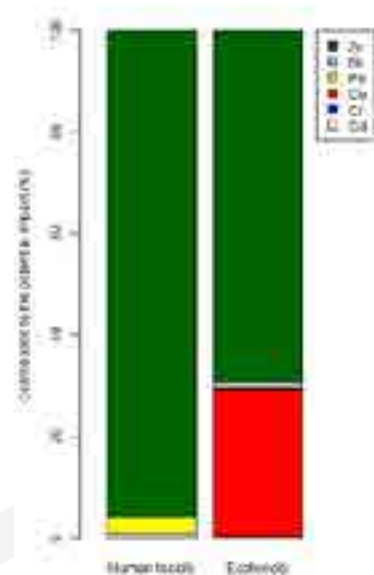
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Prasuhn, V., 2006. Erfassung der PO4-Austräge für die Ökobilanzierung - SALCA-Phosphor. AgroscopeReckenholz-Tänikon (ART), 20p.

Table 1: Concentration (quantiles 2.5%, 50% (median) and 9.75%) of trace metals in the manures (underlined= significant differences ($p < 0.05$, t-test after log transformation) between organic and conventional).

		Slurry-Dairy		Farmyard manure-Dairy		Farmyard manure-Beef		
		Conv.	Org.	Conv.	Org.	Conv.	Org.	
		(n=14)	(n=7)	(n=12)	(n=2)	(n=22)	(n=13)	
Cadmium (mg/kg DM)	2,5%	0.29	0.19	0.28	0.21	0.26	0.08	
	50%	0.60	0.51	<u>0.47</u>	<u>0.22</u>	<u>0.46</u>	<u>0.10</u>	
	97,5%	1.47	0.92	0.66	0.23	0.87	0.48	
Chromium (mg/kg DM)	2,5%	4.9	4.1	2.7	13.1	4.0	5.1	
	50%	7.4	5.5	<u>4.6</u>	<u>16.5</u>	6.8	11.7	
	97,5%	14.7	11.9	10.3	19.8	20.0	25.1	
Copper (mg/kg DM)	2,5%	39	37	16	32	13	15	
	50%	<u>77</u>	<u>56</u>	26	36	30	34	
	97,5%	116	66	106	40	170	101	
Lead (mg/kg DM)	2,5%	2.9	5.7	2.2	7.2	1.7	4.8	
	50%	<u>4.9</u>	<u>9.1</u>	<u>3.1</u>	<u>7.8</u>	5.9	8.0	
	97,5%	14.5	11.8	6.7	8.4	29.2	12.4	
Nikel (mg/kg DM)	2,5%	5.8	6.5	3.6	7.7	3.0	2.6	
	50%	8.6	7.2	<u>5.0</u>	<u>11.8</u>	6.5	8.2	
	97,5%	11.1	10.8	8.5	16.0	14.6	15.6	
Zinc (mg/kg DM)	2,5%	177	102	65	116	52	58	
	50%	<u>268</u>	<u>161</u>	95	131	131	84	
	97,5%	481	197	149	147	372	188	
Human toxicity (CTUh/kg DM)	2,5%	7.9E-06	4.7E-06	3.0E-06	5.3E-06	2.4E-06	2.7E-06	
	50%	<u>1.2E-05</u>	<u>7.3E-06</u>	4.1E-06	6.0E-06	6.6E-06	3.9E-06	
	97,5%	2.1E-05	8.9E-06	6.7E-06	6.6E-06	1.7E-05	8.4E-06	
Ecotoxicity (CTUh/kg DM)	2,5%	5.2	3.7	1.9	3.5	1.5	1.8	
	50%	<u>8.3</u>	<u>4.6</u>	2.9	3.9	3.7	2.9	
	97,5%	13.2	6.1	6.3	4.4	11.7	7.0	

Figure 1: Contribution (mean of all samples) of the trace metals in the human toxicity (total of cancer and non cancer, agricultural soils) and Ecotoxicity (Soil, unspecified) potentials, using Usetox® (V1.04) characterisation factors.



P35. Development of Characterization Factors for Pesticides in Feed Production

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Pesticides are widely applied to agricultural fields to protect and enhance crop yield. In life cycle impact assessment of cropping systems, the ecotoxicity effects are often ignored or not discussed in depth; this is mainly due to methodological challenges, such as the uncertainty of the toxicity impacts compared to other impact categories or the lack of data or characterization factors (CF's) for potential key pollutants. Here we provide new CF's for freshwater ecotoxicity potential of pesticide active ingredients (a.i) used in the production of the livestock feed: barley, maize, grass, soybean, and wheat.

The substance-specific CF's were obtained using USEtox 2.0 as characterization model, recommended by the Product Environmental Footprint (PEF) guideline to assess toxicity impacts. Physicochemical data was collected from different databases (PPDB, ECOTOX, and EFSA among others) or derived from the Estimation Program Interface Suite™ - EPISuite. The new CF's, are expressed as comparative toxic units (CTUe).

The resulting CF's may be used for ranking and identification of key a.i in the evaluation of pesticide substitution. Impacts for Danish feed production were calculated, linking pesticide emissions to impacts through environmental fate, exposure and effects. The highest impact score for freshwater toxicity potential derived from the pesticide use per hectare was for wheat and barley (5969 and 5950 CTUe/ha) and significantly lower for maize and grass.

Much remains to be done before ecotoxicity due to pesticide use is routinely included in agricultural LCA's. However, development of CF's contributes improving the analysis of toxic impacts from pesticides in cropping systems.

Keywords: Ecotoxicity, Life cycle impact assessment LCIA, impact categories.

Figure 1. Impact score for grass production in Denmark. Fraction of applied pesticide active ingredient emitted in the environmental compartments: water, air and agricultural soil (left axis) and the resulting freshwater ecotoxicity impacts (right axis).

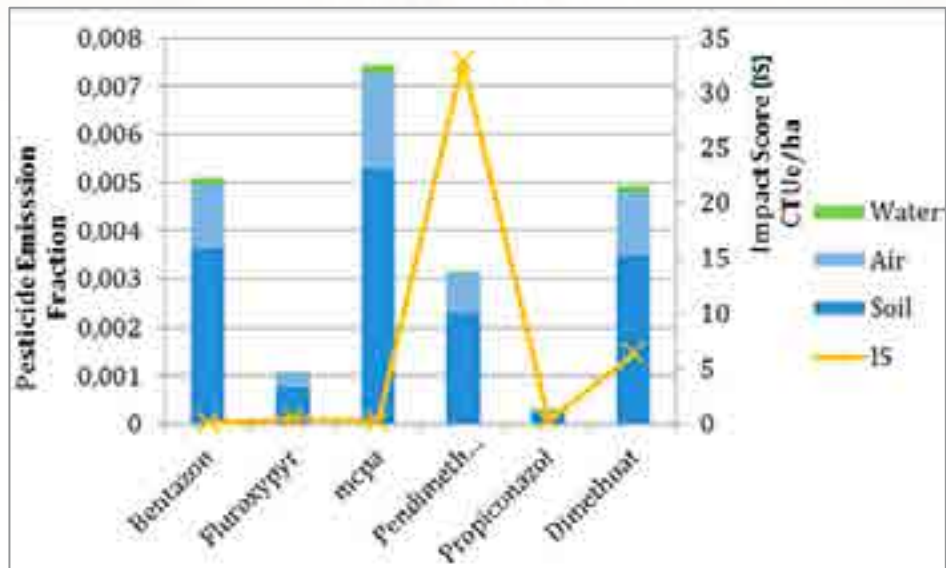
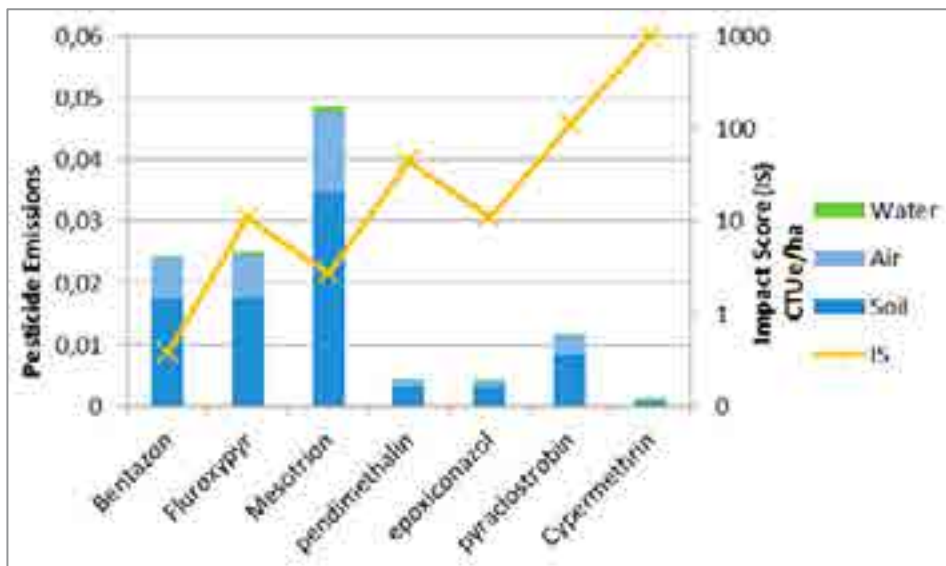


Figure 2. Impact score for maize production in Denmark. Fraction of applied pesticide active ingredient emitted in the environmental compartments: water, air and agricultural soil (left axis) and the resulting freshwater ecotoxicity impacts (right axis, logarithmic scale).



P36. Net GHG Benefits of Using the Nitrification Inhibitor Dicyandiamide (DCD)

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The nitrification inhibitor dicyandiamide (DCD) is known to be a useful mitigation option in agriculture, by reducing the conversion of ammoniacal N to nitrate N, and is itself an N fertiliser. The work aimed to quantify the net effectiveness of DCD in reducing N₂O emissions using LCA.

UK experiments from three papers were simulated with a systems-LCA model (Williams et al., 2010) using LCI data for DCD from the *Umberto*© database (5.8 kg CO₂e/kg) (Table 1). DCD significantly reduced the greenhouse gas emissions (GHGE) from direct N₂O emissions by 38% to 63% (Table 1). Including the burdens of DCD production and application superficially resulted in, at best, a decrease of 36% (compared with GHGE from N₂O only) and, at worst, an increase of 75%. Including all other processes in the systems-LCA arable model considerably reduced changes in GHGE from -8% to +4% (Table 1). The largest net reduction in GHGE emissions at field level was with the highest application rate of 360 kg N/ha on grass while using 120 kg N (in another trial) caused a net GHGE increase. Uncertainties were high when DCD was included owing to its high standard deviation. Hence, conclusions about the net benefits of using DCD as a mitigation method are tentative. It is clear that ignoring the burdens of DCD manufacture and use could cause misjudged policies to be made from a narrow GHG inventory (“end of pipe”) perspective only. DCD may be only well suited to use with high N application rates, types of N and/or particular soil textures or temperatures, needing clearer understanding of these. Better quality data on DCD manufacturing are hence needed, given the fine balances found and the high uncertainty of the only current data source.

Understanding the benefit of using DCD alone warrants investigating. The GHGE from manufacturing and application of DCD per unit N appear to be about 1.3 to 3 times higher than common N fertilisers, but the net effect is the critical outcome and, with its self-fertilisation capacity, DCD may have a valuable role to play.

Reference

Williams, et al. (2010) DOI 10.1007/s11367-010-0212-3

Table 1 Effects of using DCD on GHGE in field experiments and simulations

	Urea N rate, kg/ha	DCD rate, kg/ha	GHGE from direct N ₂ O, kg CO ₂ e/ha	Change in GHGE from direct N ₂ O	Change in GHGE from direct N ₂ O and DCD	Change in GHGE from arable LCA model including DCD
Winter wheat ⁽¹⁾	200	30	709	-63%	-36%	-8.3%
Winter wheat ⁽¹⁾	160	30	155	-48%	75%	3.9%
Spring barley ⁽²⁾	120	20	864	-40%	-24%	-12%
Spring barley ⁽³⁾	120	12.5	610	-39%	-26%	-8.1%
Winter barley ⁽¹⁾	160	30	632	-38%	-8%	-2.0%
Grass ⁽¹⁾	120	30	297	-50%	14%	
Grass ⁽²⁾	360	25	4075	-56%	-52%	
1	Misselbrook (et al., 2014) doi:10.1088/1748-9326/9/11/115006					
2	Hinton et al. (2015) doi:10.1016/j.geodrs.2014.12.002					
3	McTaggart et al. (1997) doi: 10.1007/s003740050312					

P37. Soil organic carbon depletion as a land use indicator: new modelling strategies

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ABSTRACT

Objective: Soil organic carbon (SOC) depletion is the recommended indicator for land use occupation and transformation impact assessment at midpoint – which is of critical importance in agri-food Life Cycle Assessment (LCA). Practical implementation of the well-established SOC depletion model for the calculation of characterization factors (CFs) is limited by lack of data on SOC stocks and also the need for assumptions regarding SOC regeneration times. Here, we compare modelling assumptions and propose new modelling options to address these issues.

Methods: Three main ways are used to devise SOC depletion-based CFs. Milà i Canals et al (2007) used constant land cover-specific regeneration times. Brandão & Milà i Canals (2013) and Morais et al (2016) assume constant regeneration rates. Alternatively, constant regeneration times could also be used as a simplification (e.g. equal to 20 years, as recommended by the IPCC). Here we use a global database of SOC stocks with a 1 km² resolution and apply the three modelling strategies above.

Results: We calculated CFs using all modelling strategies and determined that assumptions regarding regeneration are crucial for the absolute results of the models. However, in relative terms, all modelling choices provide similar results. We propose, as a way to move forward, the possibility of determining the regeneration time using statistical or process-based models. These novel approaches are limited by the lack of data covering all possible land cover transformations, but provide regional parameters that can increase the local applicability of models.

Conclusions: The lack of accurately determined SOC regeneration time is a crucial limitation to land use midpoint indicators. The strategies proposed in this work comply with Koellner et al's (2013) framework while providing improved CF estimations.

Keywords: Life Cycle Impact Assessment; soil carbon; land use; regeneration time

1. Introduction

Impacts from human occupation and transformation of land are increasingly recognized as essential to the accurate depiction of the environmental performance of products and services. Life Cycle Assessment (LCA) developers have introduced in the past decade several strategies to deal with land use in Life Cycle Impact Assessment (LCIA). Several models and indicators have been proposed thus far (Vidal-Legaz et al., 2016). One of these indicators is Soil Organic Carbon (SOC) depletion. The Joint Research Centre of the European Commission (EC) recommends SOC as the indicator for midpoint land use impacts (EC-JRC, 2011). The dynamics of SOC accumulation and loss are a function of soil type, climate region, land-use type and land management, which makes it a key parameter in soil management (Bot and Benites 2005) and a good proxy for land use damages to soil and biotic production potential. Among other effects, SOC improves soil nutrient availability, plant productivity and water holding capacity; it reduces surface runoff of water, sediment loss and soil erosion (Van-Camp et al. 2004). SOC increases in soils are also connected with atmospheric carbon sequestration (Teixeira et al. 2011), and is therefore essential in LCA studies interested in calculating carbon balances that involve biogenic carbon. Using SOC at midpoint level thus captures effects that lead to multiple endpoint categories in the land use impact pathway proposed by Koellner et al. (2013).

The EC suggests the use of characterization factors (CFs) for SOC depletion at midpoint that were obtained using the model developed by Milà i Canals et al. (2007). More recently, Brandão and Milà i Canals (2013) presented an updated version of the same underlying model that also uses SOC stock depletion as a midpoint indicator. They calculate regionalized CFs that consist of aggregated averages typically representative of different climate regions according to the (IPCC, 2006). Brandão and Milà i Canals (2013) concluded that more refined data was needed before accurate, geographically explicit CFs could be calculated. That was precisely the intention behind Morais et al. (2016), who operationalized the same model proposed by Brandão and Milà i Canals (2013) using a large number (more than 19,000) of field measurements for the European Union, namely the LUCASOIL database (Tóth et al., 2013). The success of this operationalization was curtailed only by restriction in the geographic scope.

In this study we aim to carry along the same path and extend the work of Brandão and Milà i Canals (2013) and Morais et al. (2016) by introducing a new global SOC database that can be used to calculate CFs. We test several methodological choices that may influence results, depending on the choices made by previous models. As explained next, those methodological choices deal mostly with the regeneration time and the locally applicable potential SOC stock.

2. Methods

2.1 Model specifications

The underlying model in the three approaches was first proposed by Milà i Canals et al. (2007) and then revised by Brandão and Milà i Canals (2013) and applied by Morais et al. (2016), and is shown in Figure 2. The model divides impacts between occupation impacts (foregone carbon due to land use) and transformation impacts (depleted carbon due to land use/cover change). All three cases agree that occupation impacts should be determined as the difference between the SOC stocks under the current (human) land regime (from hereon after “LU₂”) and the baseline SOC stocks for the same site under Potential Natural Vegetation (PNV) if land management was discontinued. As such, we calculated but do not show results here for occupation CFs.

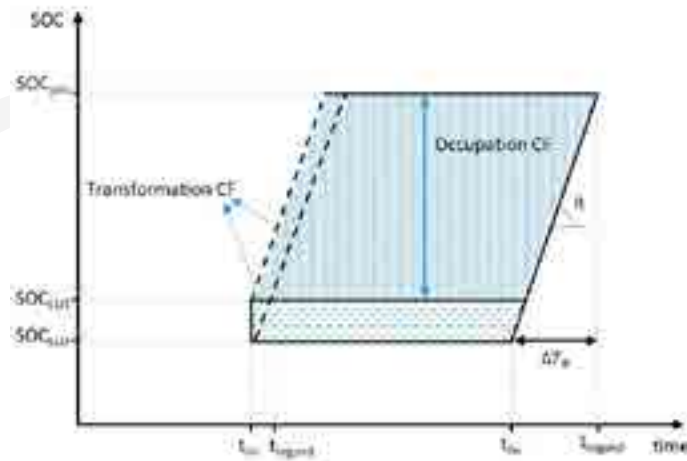


Figure 1: Schematic depiction of the model for CF calculations using SOC content (adapted from Morais et al., 2016)

The transformation CF from a certain land use 1 to another land use 2 is the change in C stocks that would take place due to the change from LU_{PNV} to LU₂ minus the change in C that already took place when LU_{PNV} was converted to LU₁. This difference is graphically equal to the area shown in Figure 2, which can be calculated using equation (1), where SOC stock units are kg C.ha⁻¹,

$$\begin{aligned} \Delta C [kg C. year. m^{-2}] &= (SOC_{pot} - SOC_{LU1}) \times (t_{regen1} - t_{ini}) + \frac{1}{2} (SOC_{LU1} - SOC_{LU2}) \times (t_{regen1} - t_{ini}) \end{aligned} \quad (1)$$

where SOC_{pot} is potential SOC stock under natural vegetation (PNV), SOC_{LU1} is SOC content before transformation, SOC_{LU2} is SOC stock in the new land use, t_{ini} is the instant when the transformation and subsequent occupation happens, t_{regen1} is the instant when SOC has reverted to the previous land use SOC stock, calculated using equation (2),

$$t_{regen1} - t_{ini} = \frac{SOC_{LU1} - SOC_{LU2}}{R} = (SOC_{LU1} - SOC_{LU2}) \times \frac{\Delta t_R}{SOC_{pot} - SOC_{LU2}} \quad (2)$$

where R is the regeneration rate and Δt_R is the regeneration time, assuming that the regeneration time (period necessary for SOC to recover to its potential maximum), is the time difference between t_{regen2} and t_{fin} . CFs are expressed in $\text{kg C}\cdot\text{ha}^{-1}\cdot\text{year}^{-1}$. The implicit indicator is depletion of SOC, which means that a positive CF implies a *loss* of SOC.

It is for transformation CFs that the three models proposed up until now vary the most, due to modelling differences regarding the regeneration time. Absence of local field data requires assumptions to enable the calculations. Milà i Canals et al. (2007) does not consider LU_1 , and for t_{regen2} uses fixed estimates valid for the entire world. The regeneration time thus changes according to the transformation process but is not regionalized. An additional problem with this approach is that the estimates were not obtained from a meta-analysis of studies but are rather expert-based. Brandão and Milà i Canals (2013) and Morais et al. (2016) assume t_{regen2} to calculate R (which is assumed to be constant for each land use in each region) and then use it to calculate t_{regen1} using equation (2). Morais et al. (2016), assumed a constant t_{regen2} for all land uses and regions of 20 years. Given the uncertainty surrounding both approaches, a simpler third option would be to consider both t_{regen1} and t_{regen2} constant and equal to 20 years, which is an approach inspired by the IPCC (2006). This approach would not require the assumption by Brandão and Milà i Canals (2013) that the regeneration time for LU_1 depends on regeneration time of LU_2 .

In this work we used all three modelling choices. These can be systematized as follows (where the expression “ $f(\text{LU}_x)$ ” should be read as “is a function of LU_x ”):

1. Following Milà i Canals et al. (2007) and Koellner et al. (2013):
 - $t_{\text{reg}}(\text{LU}_1)=f(\text{LU}_1)$
 - $t_{\text{reg}}(\text{LU}_2)=f(\text{LU}_2)$
 - $R(\text{LU}_1)=f(\text{LU}_1)$
 - $R(\text{LU}_2)=f(\text{LU}_2)$
2. Following Brandão & Milà I Canals (2012) and Morais et al. (2016):
 - $t_{\text{reg}}(\text{LU}_2)=20$
 - $t_{\text{reg}}(\text{LU}_1)=f(\text{LU}_2)$
 - $R(\text{LU}_1)=R(\text{LU}_2)$
3. In accordance with IPCC guidelines:
 - $t_{\text{reg}}(\text{LU}_1)=t_{\text{reg}}(\text{LU}_2)=20$
 - $R(\text{LU}_1)=f(\text{LU}_1)$
 - $R(\text{LU}_2)=f(\text{LU}_2)$

In the first case, the regeneration times are not functions of each other, and can vary freely as shown in figure 2.

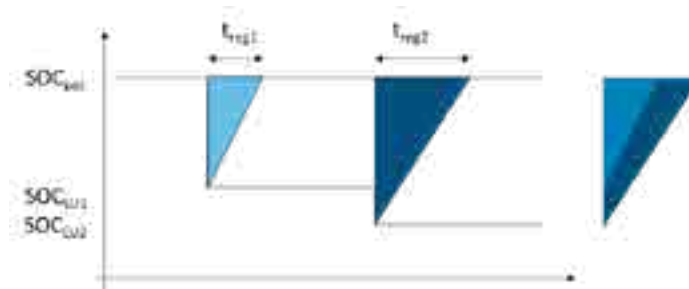


Figure 2: Schematic representation of the first modelling approach, where regeneration times and rates vary freely. The area of the triangle on the left is equal to the transformation characterization factor (CF) for the first or current land use; the one in the middle represents the CF for the second land use only; the final triangle present the difference between the two first triangles (CF as calculated in Eq. 3).

In this case the CF must be calculated graphically as the difference of areas between triangles in figure 2, which means that

$$CF = \frac{1}{2}(SOC_{pot} - SOC_{LU2}) \times t_{reg2} - \frac{1}{2}(SOC_{pot} - SOC_{LU1}) \times t_{reg1}. \quad (3)$$

Re-arranging the expression, the CFs are calculated as

$$CF = \frac{1}{2}SOC_{pot}(t_{reg2} - t_{reg1}) - \frac{1}{2}(SOC_{LU2}t_{reg2} - SOC_{LU1}t_{reg1}). \quad (4)$$

The second strategy is presented graphically in Figure 3. In this case the second regeneration time is fixed (20 years), and the first is adjusted because the regeneration time must be constant.

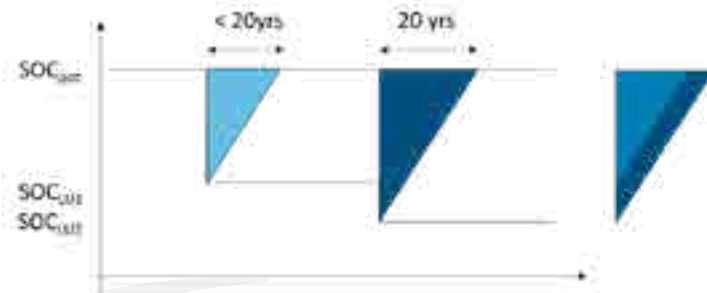


Figure 3: Schematic representation of the second modelling approach, where regeneration rates are constant. The area of the triangle on the left is equal to the transformation characterization factor (CF) for the first or current land use; the one in the middle represents the CF for the second land use only; the final triangle present the difference between the two first triangles (CF as calculated in Eq. 5).

The CFs are (assuming $t_{ini}=0$) calculated using

$$CF = (SOC_{pot} - SOC_{LU1}) \times t_{reg1} + \frac{1}{2}(SOC_{LU1} - SOC_{LU2}) \times t_{reg1}. \quad (5)$$

Finally, in the third case, the regeneration times are the same, as shown in Figure 4.

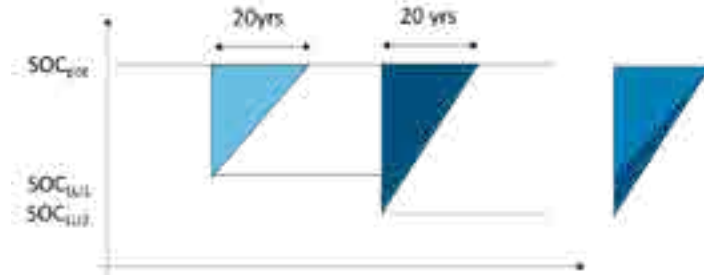


Figure 4: Schematic representation of the third modelling approach, where regeneration times are constant (20 years). The area of the triangle on the left is equal to the transformation characterization factor (CF) for the first or current land use; the one in the middle represents the CF for the second land use only; the final triangle present the difference between the two first triangles (CF as calculated in Eq. 6).

The CF is thus calculated as the difference between the areas of the triangles as in equation (3). In this case, the equation simplifies to

$$CF = \frac{1}{2}t_{reg}(SOC_{LU1} - SOC_{LU2}). \quad (6)$$

Equation (6) means that, when regeneration times are constant and equal, SOC_{PNV} is irrelevant. This means that we only needed to assume that, after 20 years of recovery from LU_1 or LU_2 , SOC

stocks will be equal, regardless of whether they are at PNV levels or not. This also means, as seen in Figure 4, when SOC stocks for the first land use are higher than for the second, that LU_1 is slower to regenerate than LU_2 , regardless of what LU_1 and LU_2 are.

2.2 Data used

We used the topsoil (0-30 cm) SOC stock map obtained from the European Soil Data Center (ESDAC) (available from: <http://esdac.jrc.ec.europa.eu/content/global-soil-organic-carbon-estimates>), with a resolution of 30 arc second (which corresponds to a grid size of approximately 1km x 1km at the Equator). We aggregated SOC stocks per geographical unit, using climate regions as described below, and soil types. This process was similar to the previous approaches and assumes that SOC stocks characteristic of each land use class are homogeneous at each combination of climate region and soil type.

Our relevant spatial scale, in line with the prior models, were the climate regions, a classification system also used by the IPCC (2006). The map of climate regions was also collected from ESDAC (<http://esdac.jrc.ec.europa.eu/projects/renewable-energy-directive>), with a similar resolution to the SOC map. Soil type classifications and maps were obtained from the World Reference Base (WRB), as depicted in Fischer et al. (2008), with the same resolution of previous maps. Finally, the land cover map was obtained from the International Steering Committee for Global Mapping (ISCGM), with the same resolution of previous maps. We reclassified this map to obtain CFs for four land use/cover classes (forest, pasture, agriculture and urban).

An additional innovation we introduced, compared to the three models mentioned previously, was a new calculation procedure for SOC_{PNV} . To avoid uncertainty in determining PNV, and also error due to the use of a different SOC database specifically for this parameter, we defined SOC_{PNV} as the maximum SOC for any land class in each geographical unit (defined as the intersection between climate region and soil type). By construction, this strategy assumes that SOC_{PNV} is always the highest SOC attainable in each geographical unit.

3. Results

Due to lack of space, we can only present here as results three maps in Figure 5, corresponding to each of the three approaches we followed, displaying CFs for the inventory flow “transformation, to agriculture”. The CFs are available at request.

Figure 5 shows that the highest CFs in absolute value are found using the first approach. This is because regeneration times are obtained from outside sources and, for agriculture, they are typically above 20 years (which is assumed to be the regeneration time for agriculture in both other cases, since agriculture is LU_2 in this case).

However, global hotspots of SOC depletion are clearly marked in all three cases. Transformation to agriculture leads to higher SOC losses in Northern Europe and Canada, along with regions in the Equator. CFs calculated using the three approaches are very correlated (data not show here), which means that there is no geographical distortion from using one approach over another.

This means that, despite the striking differences in absolute values, it is possible that when applied to case studies these CFs do not yield significantly different results in the midpoint land use impact category. The difference in CFs is likely to be very similar if flows are taken from any two different regions of the globe, as it can be observed that case a) in figure 5 always has much higher CFs everywhere when compared to b) or to c).

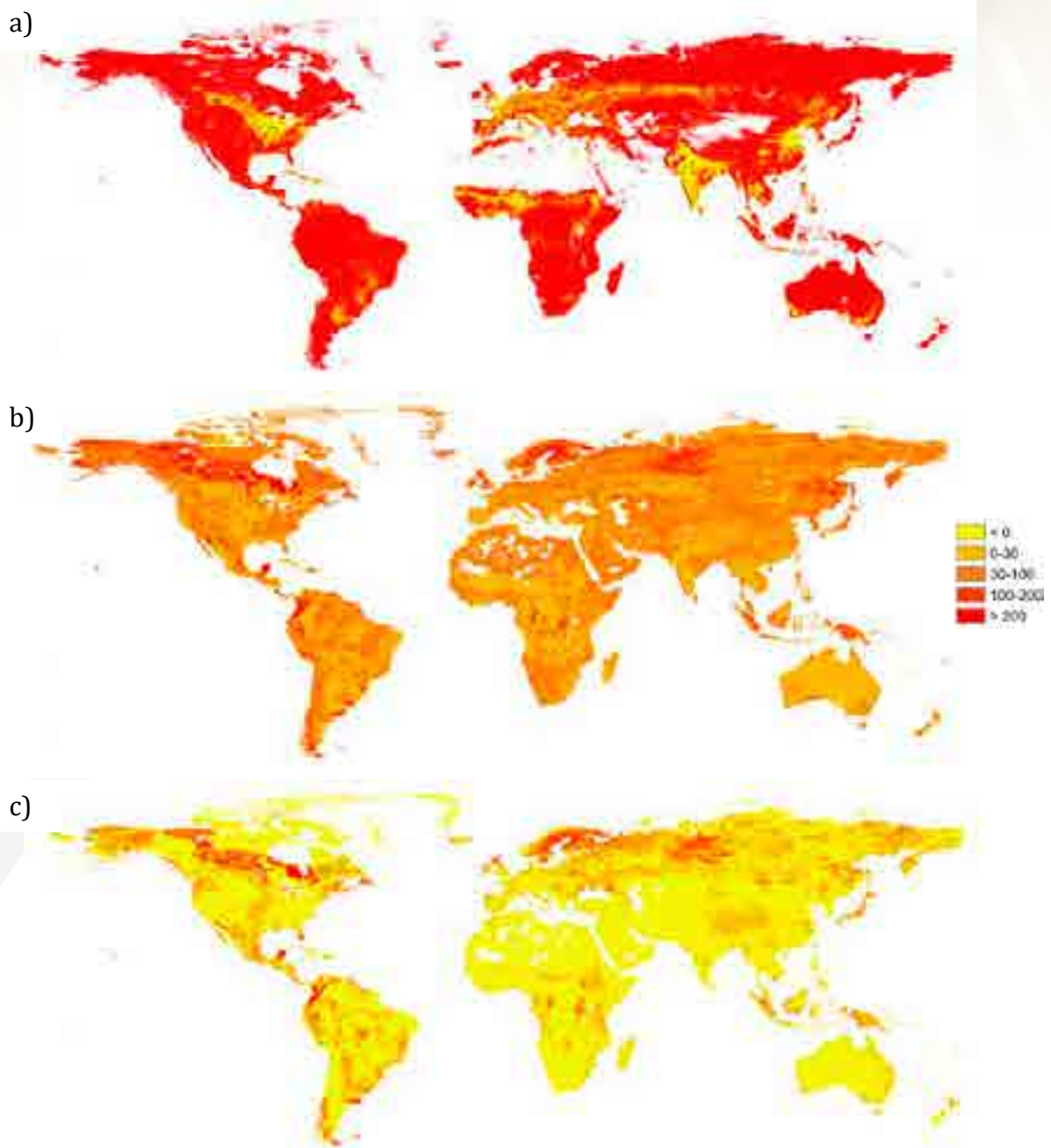


Figure 5: Graphical representation of characterization factors ($\text{kg C deficit.ha}^{-1}$) for transformation from agriculture according to the three modelling approaches described in the text: a) approach 1; b) approach 2; c) approach 3.

4. Discussion

In this study we use three different approaches to calculate CFs for land use LCIA. We used a global dataset, which is an advantage over the work of Morais et al. (2016) which was for the European Union only, and also improves the geographical detail of the global CFs in Brandão & Milà i Canals (2013). We additionally tested different assumptions regarding regeneration of SOC, which is a necessity to calculate CFs using these models as there are no quality datasets for SOC regeneration times. We showed that all modelling approaches lead to similar results relative to each other, but very different CFs in absolute value.

Further work should now try to apply these CFs to case studies to understand whether or not the use of one modelling strategy over another leads to different LCIA results for particular products. If, as we expect, differences are small, then the simplest model can be used (which in this case would be the third approach). If, however, differences are large, more research will also be needed to improve our understanding of SOC response to regeneration. One process that could take us closer to finding regionalized estimates for regeneration times and rates would be statistical modelling. There are two possible pathways (1) statistical analysis of global databases; or (2) meta-analysis of regeneration studies. The first approach would require, for each geographical region, pinpointing regions where there has been land use change and compare the SOC stocks in those areas with others where no land use change has taken place (in a certain time between land use observations). It is possible, however, that the diversity of land transitions in all regions is insufficient to obtain good estimates. The second approach is to look up studies where passive and active regeneration projects took place and where monitored using SOC sampling and measurement. Geostatistical methods can then be used to extrapolate from local test sites to larger geographical regions.

In LCIA there is one final approach that should be considered to get around the issues with these modelling approaches. It would be possible to change the paradigm and start working with process-based models. These models, integrating many complex biogeochemical processes formulated on mathematical-ecological theory and accounting for climatic variations, agricultural management practices and soil conditions (Cuddington et al., 2013), would enable the construction of scenarios with and without human management for virtually every region in the world. They would also introduce a temporal component, addressing different time and spatial scales, which would enrich the current modelling framework in figure 1. They are more computationally heavy, requiring many inputs and more data than the proxy-based model (such as Milà i Canals et al., 2007; Koellner et al., 2013) but carry higher level of detail and lower uncertainty allowing a more precise assessment (Othoniel et al., 2016).

There are many examples of models that can be used to achieve this end. RothC is a model of carbon turnover in non-waterlogged soils. It is developed to model the carbon turnover in arable soils, grassland and forest. It takes into account the effects of temperature, moisture content and soil type. The model is divided in five compartment systems: inert organic matter, easily decomposable plant material, resistant plant material, microbial biomass and humified organic matter (Smith et al., 1997).

DNDC models the dynamics of carbon and nitrogen biogeochemistry in agricultural ecosystems. It is composed by 6 sub-models: soil climate, crop growth, decomposition, nitrification, denitrification and fermentation. SOC is divided in 4 different pools: plant residue, microbial biomass, active humus and passive humus.

CENTURY is a model that is used to simulate carbon and nutrient dynamics for different types of ecosystems (grassland, agricultural crop, forest and savanna). The model runs using a monthly time step and can simulate the dynamics of Soil Organic Matter (SOM) for one year, centuries or even for thousands of years. The model is divided in 6 sub-models: SOM, nitrogen, phosphorus, sulphur, plant production and water budget, leaching and soil temperature sub-model. The SOM sub-model includes three soil organic matter pools (active, slow and passive). CENTURY also enables for the user to schedule management events and crop growth controls at specific times, and can be defined in blocks that are repeating sequences of series of events. Options such fertilizer addition, different types of harvest, effects of fire and grazing, senescence for crops, addition of organic matter, irrigation and erosion are available in the event commands of the model.

5. Conclusions

We conclude that even when the same underlying model is applied to the calculation of CFs for land use, results may be very different (in absolute terms) depending on particular modelling choices nested within the framework. However, the relative differences between regions seem to cancel out the differences which may mean that when applied to particular case studies the results may be less affected by the modelling choices than expected. More work is necessary to apply the CFs introduced here (and others), and new modelling options are available to move this line of research further.

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P38. Soil organic carbon contribution to GHG balance at cropping system scale

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Mitigating the greenhouse gas (GHG) balance at the scale of an agricultural territory implies to identify the most contributing combinations of cropping systems (i.e. crop rotations, management practices) and soil types. Now, GHG balances usually do not account for soil organic carbon (SOC) evolution in time.

Following the IPCC framework, the avoided or additional CO₂ emissions consecutive to SOC changes were integrated to a GHG balance at the cropping system scale. SOC evolution was assessed by C-AMG model. In the case of SOC losses from mineralization, direct and indirect N₂O emissions due to annually mineralised N were added. N₂O direct and indirect emissions from other N sources were also included in the calculation, as well as upstream and direct GHG emissions from fertilisers and agricultural machinery. To manage data availability constraints at the territory and at the cropping system scales, we inferred crop practices influencing SOC and nitrogen inputs in the cropping system from existing databases. The Tardenois (Picardy, Northern France) farming systems are mainly field crop, potatoes, and mixed crop-livestock.

In Tardenois, most of the 2154 assessed situations showed carbon storage. Additional N₂O emissions from SOC mineralization showed limited contribution to total GHG emissions of cropping systems. At the cropping system scale, our approach shows that, in the long term, SOC evolution greatly influences the GHG balance. SOC fluxes can increase the GHG balance of 1 ha up to 70% and decrease it up to 76%, under the combined influences of the cropping system and the soil type.

POSTER PAPERS

MEAT PRODUCTION 1

6. Meat production (1)

P39. The Water Footprint Of Beef Production In The Brazilian Biomes Pantanal And Pampa

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The increasing worldwide demand for animal protein has raised consumers' concern about the sustainability of the production process used in livestock chains. As Brazil is one of the main producer and exporter of animal products, it is relevant to investigate the sustainability of the production process used in livestock chains in this country. Besides, given the climate variation, diversities in the environment, lack of resources and loss of biodiversity, the production of sustainable animal products has become a challenge. Moreover, even in countries like Brazil, which has abundant availability of water sources, the water crisis has become an issue. Indeed, it has been observed water restrictions in some of the Brazilian regions. The production of all kinds of food requires water. Therefore, it is necessary to evaluate how much water is used in all food production process. Few studies have analysed the water footprint of beef production in different regions and under different production systems, which demonstrated the need of more research to quantify the water use in the beef production chain. Therefore, an indicator of water footprint was used to determine the water use for one kilogram of beef ready for human consumption. Using life cycle assessment, this study aims to determine the water footprint of beef production along the whole beef production chain. The study regions encompass the Brazilian Biomes Pantanal and Pampa. Most of the beef production in Brazil occurs in these two biomes, and therefore, most of the animal protein consumed in Brazil is from Pantanal and Pampa. It is expected that stakeholders in the beef production chain can use the results of this research to improve the production process and commercialization of animal protein.

P40. Life Cycle Assessment of Intensive Beef Production System using Agricultural Residues in Thailand

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We previously assessed the environmental impacts of extensive and intensive beef production systems in northeastern Thailand, using life cycle assessment (LCA). The results showed the intensive system had lower greenhouse gas (GHG) emissions than the extensive system; however, the GHG emissions derived from purchased concentrate feed was high in the intensive system. Here we assessed an improved intensive beef production (IINT) system utilizing agricultural (processing) residues as feed sources and compared the results with the previous results.

An LCA model was developed based on data collected by farm survey as well as literature and LCA databases. The IINT system produces grass and corn on farm and purchases some concentrate feeds. In addition, the IINT system utilizes agricultural residues as feed such as pineapple peel, flour from pineapple stalk, and soybean peel. Bulls are slaughtered at 25 months of age, whereas cows are fattened after one calving and then slaughtered at 36 months of age. The enteric methane emission was calculated using an equation based on a number of studies that have measured enteric methane emissions in Southeast Asia.

As a result of LCA, the IINT system had the lowest GHG emissions (10.6 kg-CO₂e/LW) and smaller energy consumption than the intensive system. The IINT system reduces enteric methane emissions by increasing productivity of cattle and reduces GHG emission associated with purchased concentrate feeds by utilizing agricultural residues as feed sources.

The results suggest that use of agricultural residues helps to balance the increasing productivity with the environmental sustainability of beef production.

P41. Eco-efficiency of the meat and leather production in a transitional stockbreeding system located in a Mediterranean island

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I. Objective

Stockbreeding in islands is essential to preserve native species, as well as to provide the population with local products and approach self-sufficiency. In this sense, the environmental and economic performance of these systems needs to be addressed to define the most suitable production systems. The goal of this contribution is to analyze the eco-efficiency of the meat and leather produced in the Biosphere Reserve of the Island of Minorca (Spain) through Life Cycle Assessment (LCA) and Life Cycle Costing (LCC).

II. Method

An LCA and a simplified LCC were combined to provide data on the eco-efficiency of the meat and leather production based on ISO 14045:2012. The system under analysis (Figure 1) consisted of three main stages, namely breeding, fattening and processing, each of them taking place in different farms and facilities. This process was the evolution towards a decentralized production system, but seeks to improve its performance. The functional units were the production of 1 kg of meat and 1 kg of leather; mass allocation was applied to the total impact of the system considering the live weight of fattened beef. Data on the inputs and outputs to and from each stage were supplied by the facility managers.

III. Main result

The estimated carbon footprint of meat and leather was 13.8 and 2.69 kg of CO₂eq, respectively (Table 1). In this system, 84% of the environmental impacts were related to the fattening process due to the extensive use of fodder. In contrast, the breeding stage was the main contributor to the economic costs (62%), which were associated with the diesel used for working the land. This flow was inexistent in the fattening process. The breeding stage resulted in fewer emissions per invested euro (Figure 2) and was the most eco-efficient stage of the system.

IV. Implications

This approach shows that the sustainability of meat products that are originally bred in natural areas might worsen due to failures in the supply chain and final treatment. To ensure the quality of native and original products, an integrated assessment of all of the stages is needed. At the end, this should lead to an ecological stockbreeding system that increases the eco-efficiency standards of meat products in the island.

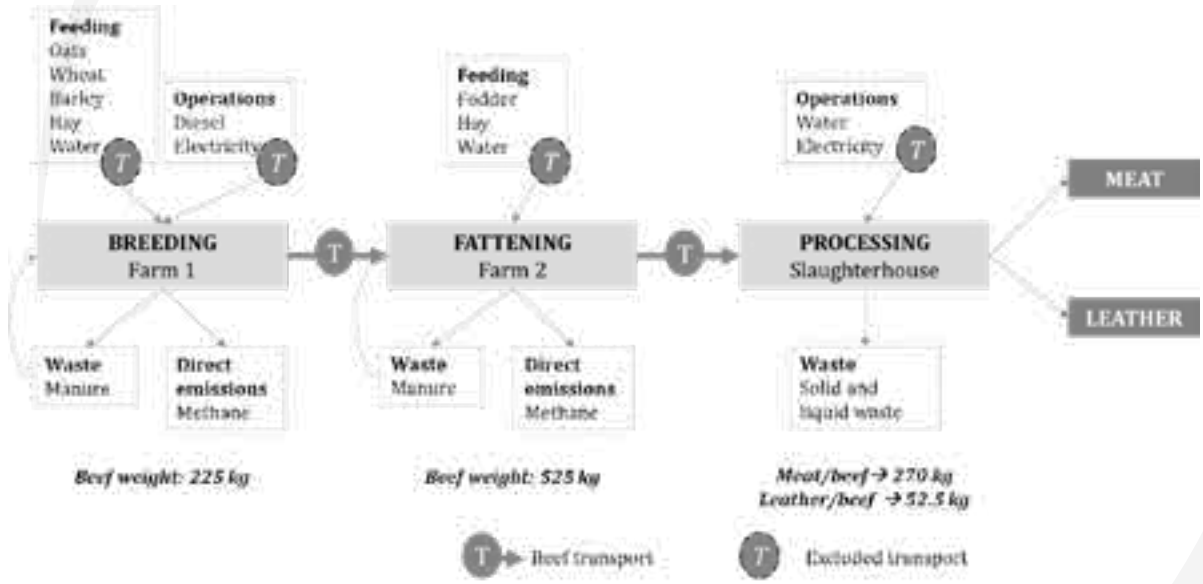


Figure 1 System boundaries of the production system under analysis

Table 1 Environmental and economic profile of processed meat and leather in Minorca

Products	Climate Change Potential (kg CO ₂ eq.)	Economic Cost (€)
Meat (1 kg)	13.8	1.93
Leather (1 kg)	2.69	0.37

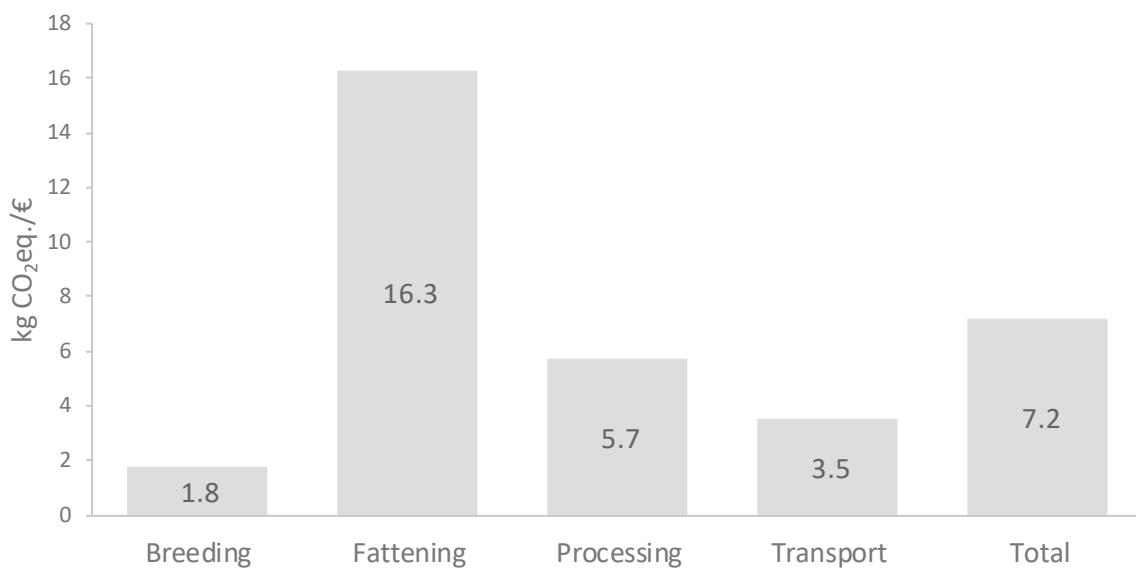


Figure 2 Eco-efficiency ratio for each of the stages of the production system



P42. Quantifying Animal Welfare of Beef, Pork and Poultry

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Meat consumption is already high on most continents with the exception of Africa and some Asian countries (Fig. 1). The consumption is expected to further increase with a growing population and its affluence. The development raises concerns about the environmental sustainability of meat consumption and also the associated animal welfare. Few studies have considered animal welfare in life cycle assessment (LCA) so far. This study seeks to develop an animal welfare indicator which allows for simple integration into the LCA framework.

The newly developed indicator takes into account the (1) life quality (mainly space allowance), the (2) life duration (relative to the animal's life expectancy), and the (3) number of animals affected by meat consumption. Following the assessment of human health impact in LCA, the indicator expresses animal welfare loss as animal-equivalent life years (ALY). The indicator was applied to the three most commonly eaten types of meat, which are beef, pork and poultry. The lives of the animals are either equally valued (scenario 1) or values differ among species (scenario 2). The required input data was obtained from existing literature (12 cases for beef, 7 for pork, 8 for poultry).

Poultry causes by far the highest impact, followed by pork and lastly beef (Fig. 2). The large difference is mainly driven by the number of animals affected, as this parameter varied most. Since chickens are much smaller than pigs or cattle, they also provide much less meat per animal. The indicator is not sensitive to the life quality, because all three types of animals are slaughtered at a very young age.

The difference in animal welfare loss per kg of meat indicates that, besides the amount of meat, the type of meat consumed clearly matters. The preference for the type of meat differs among countries (Fig. 1). Countries like the United States, which are characterized by a large meat consumption and a preference for poultry, can improve animal welfare substantially by replacing some of their poultry by pork or beef. However, this is in contrast to environmental impacts related to meat production.

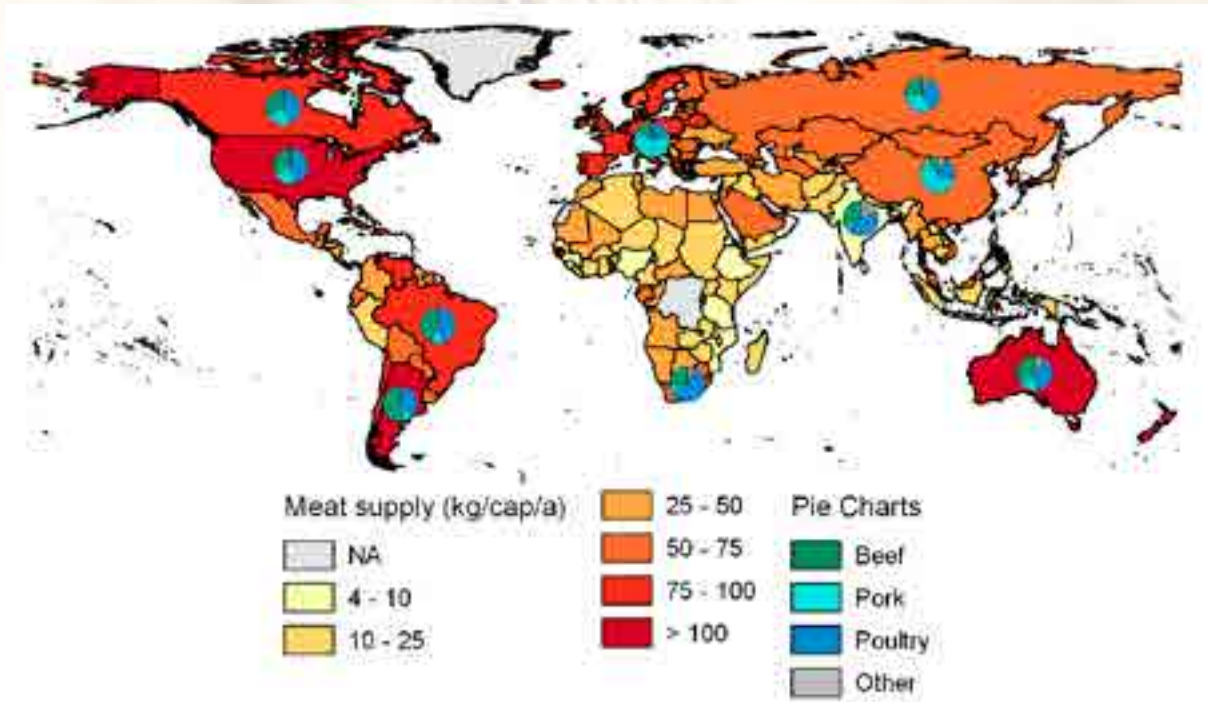


Fig. 1: Meat supply in 2011 based on FAOSTAT data.

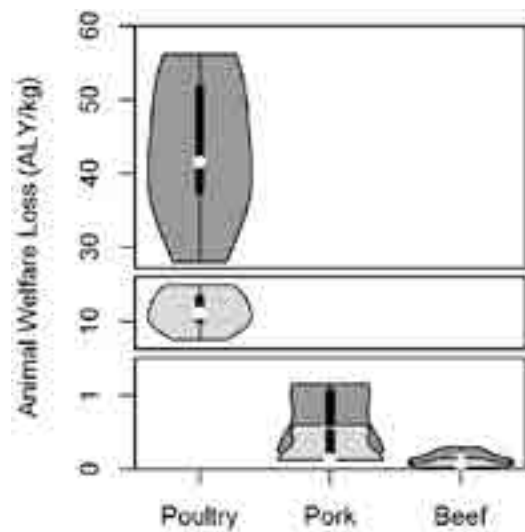


Fig. 2: Animal welfare loss of meat. The violin plots combine box plots with density plots. The white dot marks the median. The y axis is broken and scales differ above and below the first break. In dark grey violin plots, all life is valued equally (scenario 1). In light grey violin plots, animal life values differ among species (scenario 2).

P43. Virtual Water Consumption Of Beef Production And Industrialization In Brazil's South Region

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The growing global demand for animal protein has raised some concerns about the sustainability of its production processes. Considering that Brazil is one of the largest beef producers and exporters in the world, it is essential to study the environmental sustainability of Brazilian beef supply chains. Brazilian beef production has grown since the year 2000, and will maintain its growth to accomplish protein demand. Thus, evaluating beef's production processes and its water consumption is important. To this study, we used data from beef cattle of the Hereford breed produced in areas with natural pastures. Cattle was reared in the State of Rio Grande do Sul, Brazil's southern region, where Pampa biome predominates. The state produced ~27.000.000 heads of cattle in 2014.

To evaluate the virtual water consumption, this study used the method presented by Chapagain A. K. and Hoekstra A. Y. (2003) (See Table 1). The whole animal's life cycle was considered. For calculations, the variables are presented as follows: VWCdrink represented the water consumption by the animal from its birth until its arrival to the slaughterhouse; VWCfeed represented the water in the pasture consumed by the animal throughout its life cycle (data from CROPWAT 8.0, and FAOSTAT); VWCabattoir represented the water used for slaughter and processing at the slaughterhouse (Pacheco & Yamanaka, 2006).

The specific water demand (SWD) represented the volume used to produce 600 hectares of pasture. Data related to the animal's water consumption in its different phases was taken from Palhares (2005). The weight for slaughter was 430 kg, and the average time for each phase was 280 days, considering Rio Grande do Sul's soil and climatic conditions. Each phases' time, water consumption and pasture consumption are presented in Table 2.

The resulting water consumption for beef production was 5.894 liters/kg. Most water needed represents green water (5.891 liters/kg), and gray water (1 liter/kg). Blue water

came mostly from rain. Therefore, this study demystified other studies in which water consumption for beef production was found to be 13.133 liters/kg or 16.000 liters/kg.

Table 1 – Equations for calculating water footprint

Equation	Formula
1	$VWC_a = VWC_{feed} + VWC_{drink} + VWC_{abattoir}$
2	$VWC_{feed} = (\int \{ SWD \times C \} dt) / W_a$
3	$SWD = CWR / CY$
4	$VWC_{drink} = \int q_d dt / W_a$

Table 2– Data for the measurement of water consumption per kilogram of meat in the Pampa biome


	Slaughter weight (kg/animal)	430
Time (days)	Pregnancy	281
	Calf (0 to 6 months)	180
	Steer (6 to 12 months)	330
	Steer (12 to 36 months)	330
	Total	1121
	Average	280
Water consumption (l/day/animal)	Pregnancy	55
	Calf (0 to 6 months)	9
	Steer (6 to 12 months)	18
	Steer (12 to 36 months)	39
	Average	30,25
Pasture consumption (kg/year)	Pregnancy	3.531,38
	Calf (0 to 6 months)	655,36
	Steer (6 to 12 months)	1.379,70
	Steer (12 to 36 months)	1.972,83
	Feed Average Volume (kg / year)	1.884,81
	Specific water demand (l / kg)	435
	Water needed for pasture (l / year)	819.894,25
	Total water supply	2.518.086,19
	Slaughterhouse water consumption (l/animal)	1.500

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P44. Carbon Footprint Of Different Beef Production Systems In The Pampa Biome

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Pampa biome is an area covered with natural grassland located in the southern region of Brazil. In this biome, soil and climatic conditions are propitious to livestock rearing. Continuous and extensive grazing is the main type of feeding used for cattle production. Moreover, the search for production processes that can decrease environmental impacts of agricultural systems is necessary.

The State of Rio Grande do Sul has ~ 7% of Brazilian's cattle herd. Even though the use of Native Pasture for livestock production is significant in the region, advances in technology are responsible for the implementation of other types of pasture, increasing the stocking rate. Therefore, measuring the environmental impacts of different systems is important. In this sense, the life cycle analysis (LCA) methodology was applied with the participation of a multidisciplinary team. The objective of this study was to analyze the carbon footprint from beef cattle produced in the Pampa biome using LCA on the three most widely used systems: Native Pasture (NP), Improved Native Pasture (INP) and Fertilized Native Pasture (FNP). This study analyzed livestock production in its full cycle consisting of the stages of pregnancy, calf rearing, and fattening.

The animals were Hereford breed. The pregnancy and growth phases used data from Ruviaro et al. (2015), while the rearing and fattening phase used data from Genro et al. (2015). Table 1 presents the most common production systems, involving NP, INP, and FNP, making a combination of these systems in 20%, 40%, 60% and 80% ratio. The weight gained varied according to the diet, and each scenario determined a different final weight, being 460 kg, 450 kg and 440 kg of live weight for NP, FNP, and INP, respectively (Table 2). Depending on the degree of intensification, it was possible to note a reduction in greenhouse gas emissions. On system I, CH₄ accounted for 95% of the emissions from animals in native pasture while in system III, Fertilized Native Pasture, CH₄ accounted for 89% of the emissions (Figure 1).

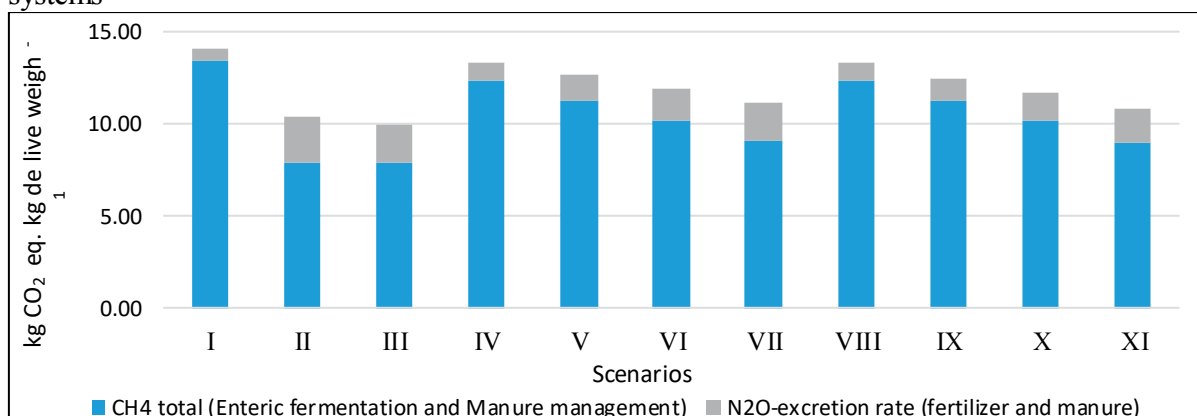
Table 1 - Scenarios used on GHG's calculations

Scenario	Productive Sistem
I	Native Pasture
II	Fertilized Native Pasture
III	Improved Native Pasture
IV	Native Pasture 80% - Fertilized Native Pasture 20%
V	Native Pasture 80% - Improved Native Pasture 20%
VI	Native Pasture 60% - Fertilized Native Pasture 40%
VII	Native Pasture 60% - Improved Native Pasture 40%
VIII	Native Pasture 40% - Fertilized Native Pasture 60%
IX	Native Pasture 40% - Improved Native Pasture 60%
X	Native Pasture 20% - Fertilized Native Pasture 80%
XI	Native Pasture 20% - Improved Native Pasture 80%

Table 1 – Systems description, days of grazing, live weight gain and live weight supported

Scenario	I	II	III	IV	V	VI	VII	VIII	IX	X	XI
days of grazing	1266	759	667	1165	1063	962	860	1146	1026	907	787
Calf	165	180	190	168	171	174	177	170	175	180	185
Rearing	255	334	360	271	287	302	318	276	297	318	339
Fattening	460	450	440	458	456	454	452	456	452	448	444
Calf	133	148	158	136	139	142	145	138	143	148	153
Rearing	90	154	170	103	116	128	141	106	122	138	154
Fattening	206	116	81	188	170	152	134	181	156	131	106
Total	428	418	408	426	424	422	420	424	420	416	412
Calf	0.970	1.575	1.357	1.091	1.212	1.333	1.454	1.047	1.125	1.202	1.279
Rearing	0.940	1.296	1.396	1.011	1.082	1.153	1.224	1.031	1.122	1.213	1.304
Fattening	0.940	1.296	1.396	1.011	1.082	1.153	1.224	1.031	1.122	1.213	1.304
Média	0.950	1.389	1.383	1.038	1.125	1.213	1.301	1.037	1.123	1.210	1.296

Figure 3 - Methane emissions, nitrous oxide and CO2 equivalent in the different systems





Reference

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POSTER PAPERS

FEED AND FERTILIZER

7. Feed and Fertilizer

P45. Comparative analysis of regionalized inventories: Life Cycle Assessment of Portuguese maize, wheat, barley and oat

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ABSTRACT

Objective: Regionalization is one of the current priorities in Life Cycle Assessment (LCA). To enable accurate regional-level studies, it is essential to improve Life Cycle Inventories (LCI). Increasingly new methods and initiatives for regionalized LCI are being proposed, as for example the Agribalyse project in France or the guidelines by the World Food LCA Database. Consequently, food databases are progressively moving towards including more regional and country-specific records. So far, however, there is an absence of studies that test regionalized inventories to check for inclusiveness and representability. In this paper we used Portugal as an example and compared the inventory interventions included in the Agri-Footprint database v2 for maize, wheat, barley and oat with scale-consistent regional statistics from Portugal.

Methods: We considered only life cycle stages occurring at the farm, i.e. a cradle-to-gate approach. The functional unit used was 1 kg of product. The Portuguese LCI interventions adapted were obtained in Morais et al (2016a), who adapted some inventory interventions for agri-food products in Portugal consistently with Agribalyse rules. Main adaptations are in fertilizers and pesticides applied, crop yield, land use and transformation. For impact assessment (LCIA), we used the impact category Global Warming Potential (GWP), measured in kg CO₂ eq. We used SimaPro v8.1 to perform calculations and analyses.

Results: Results show that fertilization was the key contributor to differences between results obtained using national statistics and Agri-Footprint. These differences in fertilizer use explain most of the differences between the records. Land use and respective farming activities in the LCI are also particularly relevant to explain the differences among the interventions.

Conclusions: The comparison of results from using a secondary database (Agri-footprint) with the same inventory adapted using highly specific, regionalized data for Portuguese products, yielded significant differences that illustrate the need for a better understanding of when secondary data can be used to represent the impacts of each product.

Keywords: Life Cycle Inventory; Regionalization; Agri-food; Inventories comparison

1. Introduction

Life cycle assessment (LCA) is a method used to measure the performance of a product or service in every stage of its life cycle (Hellweg and Milà i Canals, 2014). Life cycle inventory (LCI) is the stage where most time and effort is dispended, as it is the phase where data is compiled to characterize the system. The LCI is often separated in three components: background data (processes deep within the supply chain), foreground processes (first tier of processes required in production) and activity data (process data measured in situ for the target of the study).

Regionalization in agri-food LCA studies is a relevant issue, due to the fact that the variance of results for agri-food products is relatively high (Haas et al., 2000; Teixeira, 2015). Nevertheless, standard databases may be insufficient to grasp accurate regional data (Reap et al., 2008), as location is a critical aspect in agriculture stages (Roy et al., 2009). Thus, local data in inventories is required to capture the regional features of the agricultural operations and the processes it depicts. Nevertheless, LCI regionalization should be led using a coherent and consistent approach, rather than study-specific updates, resorting to international frameworks to ensure comparability with international LCA studies (Yang, 2016). At country scale, Agribalyse (ADEME, 2013) was a pioneer project conducted to produce a regionalized agricultural inventory, mandated by the French government. The World Food LCA Database (WFLDB) (Nemecek et al., 2014) is a project that aims to produce directives that support the establishment of regional databases for global agri-food products, developed by Quantis and the Swiss Institute for Research in Agriculture, Nutrition and the Environment, Agroscope. Recently, Morais et al. (2016b) assessed progress in Portugal towards regionalization in agri-food sector and Morais et al. (2016a) started to produce a national, consistent inventory for Portugal. The Blonk Agri Footprint BV also produce a new version (v2) of Agri-Footprint database (Blonk Agri-

footprint BV, 2015), covering Portugal. These attempts highlight the importance of regionalization as the next step in the evolution of accurate inventories.

This study aims to compare results using an LCI adaptation drawn from Morais et al. (2016a) and the Agri-Footprint database v2, for the all four Portuguese products included in Agri-Footprint (Blonk Agri-footprint BV, 2015). We assessed how the two approaches affect results for the agricultural systems mentioned.

2. Methods

The objective of this study was to compare the inventory flows obtained using the method laid out by Morais et al. (2016a) and the flows in the Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015). We performed the comparison for all Portuguese products present in Agri-Footprint (Blonk Agri-footprint BV, 2015), which are: maize, wheat, barley and oat. We opted for the complete list of products in order to enable a thorough comparison between these two inventories for one country (Portugal).

Regarding the approach of Morais et al. (2016a), the LCI interventions adapted are fertilizers and pesticides applied, crop yield, land use and transformation, soil loss and greenhouse gases (GHG) emissions (including carbon dioxide after urea or lime applications), i.e. it is not a complete inventory. These interventions are completed with background data from ecoinvent v3 (Weidema et al., 2013). The main intervention adapted by Morais et al. (2016a) was the type and amount of fertilizers applied by crop. Fertilization, in agri-food products, is particularly important as the main source of direct and/or indirect GHG emissions. Fertilizers application was adapted to Portugal using regional data, at Agrarian region level, using official data from Gabinete de Planeamento, Políticas e Administração Geral (GPP), Portuguese Agriculture Ministry. Next, these regional fact sheets (GPP, 2001) for each crop were corrected using the total national consumption obtained from the Portuguese statistical office, INE (INE, 2015). This process guarantees scale consistency, in the sense that the sum of all fertilizer consumption from all crops produced in all regions is equal to the amount of fertilizers used in the country.

Table 1 presents the correspondence between product processes from Morais et al. (2016a) and Agri-Footprint database v2. For Agri-Footprint database v2 products we opted to use mass allocation (rather than economic or energy allocation). Oat is missing from ecoinvent v3 (which is necessary as background data for processes in Morais et al., 2016a), and thus the comparison between inventories was realized only for direct emissions associated with fertilization and crop residues.

Table 1: SimaPro processes used in this study from ecoinvent v3 adapted with Morais et al. (2016a) and Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015)

Morais et al. (2016a)	Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015)
Maize grain {PT} production Alloc Def, U	Maize, at farm/PT Mass
Barley grain {PT} barley production Alloc Def, U	Barley grain, at farm/PT Mass
Wheat grain {PT} wheat production Alloc Def, U	Wheat grain, at farm/PT Mass
Oat (does not have ecoinvent process)	Oat grain, at farm/PT Mass

Regarding system boundaries, we used a cradle-to-gate approach, adapted to the maximum level of detailed allowed by the Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015), e.g. indirect machinery emissions related to product transport inside the farm. Figure 1 presents the stages considered (only stages occurring in the farm). To simplify the comparison between inventories we used a mass basis functional unit (FU), i.e. 1 kg of product.

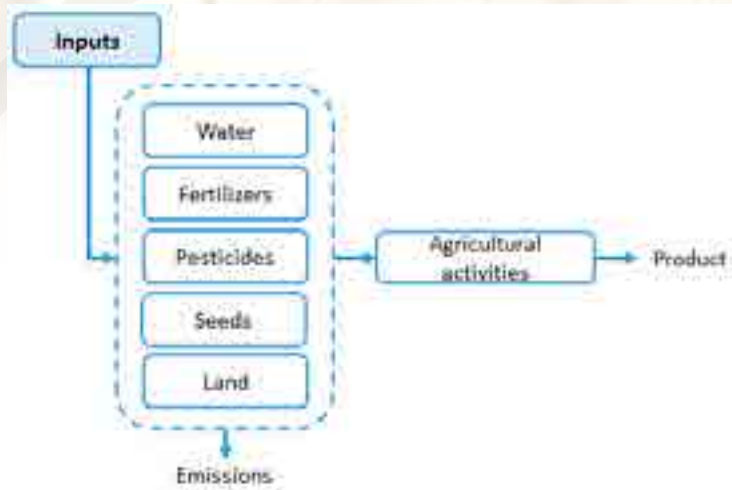


Figure 1: System boundaries considered in the study

Regarding Life cycle impact assessment (LCIA), we used only Global Warming Potential as an impact category due the fact that interventions covered by Morais et al. (2016a) have an implication mostly on GHG emissions. We used GWP potentials from Intergovernmental Panel on Climate Change (IPCC), with a time horizon of 100 years, and recommended also by the European Union (EC - JRC, 2011). We used SimaPro v8.1 to perform calculations and the analysis.

3. Results

Figure 2 shows results for the GWP impact category for the four products assessed. The main contribution to GWP is fertilization. GHG emissions from machinery used in agricultural activities are also relevant to explain the differences in results, due to the fact that these activities depend on land use occupation (expressed in SimaPro as “Occupation”) and respective activities areas. This means that higher land use area typically leads to higher GHG emissions per functional unit.

Results also show that wheat is the only product where the method put forth by Morais et al. (2016a) led to higher GHG emissions. This fact is justified by the assumed fertilizer quantity applied, but also due to high land use. In fact, wheat is the only product where fertilizer quantity applied and land use (i.e. “Occupation” process) according to Morais et al. (2016a) is higher than what is obtained when using the Agri-Footprint database v2 (Blonk Agri-footprint BV, 2015).

However, the differences between inventories are relatively low, approximately 0.31 kg CO₂eq. The highest difference is found in wheat, about 0.61 kg CO₂eq, and the lowest difference is in maize, about 0.03 kg CO₂eq.

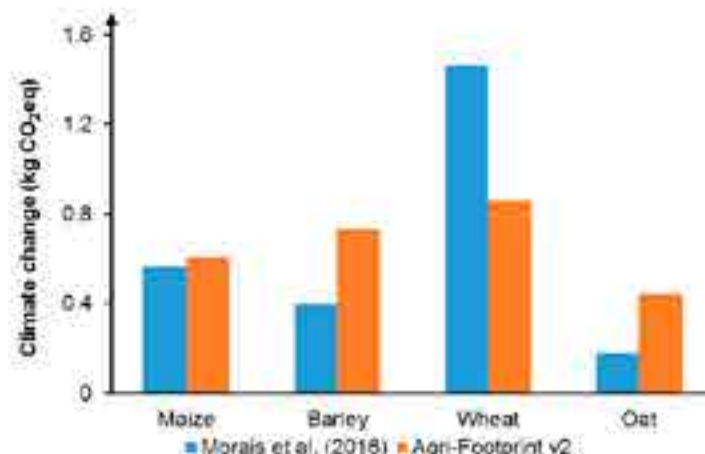


Figure 2: Difference between in the climate change impact category, according to the ILCD method.

4. Discussion

When conducting an LCA study, LCI is the stage where most effort is required. Secondary databases reduce this effort, but they should be accurately regionalized if they are intended to depict the processes accurately. In agri-food studies regionalization is particularly relevant due to the locally specific processes and activities involved in the agricultural stage of food products. However, this cannot come at the expense of comparability between studies, which is a risk if inventories and models are built independently for very specific situations. There is thus a pressing need for regionalization methods that are built consistently with international frameworks, as the WFLDB (Nemecek et al., 2014). ADEME (2013) and Morais et al. (2016a) are two cases of international frameworks application in a concrete cases, France and Portugal, respectively.

The comparisons between inventories performed in Morais et al. (2016a) between their interventions and ecoinvent v 3 (Weidema et al., 2013), and in this study between Morais et al. (2016a) and Agri-Footprint v2 (Blonk Agri-footprint BV, 2015) reveal significant differences in the outcomes of impact assessment models depending on the method used to draw regionalized inventories.

System boundaries considered in this study is the same that Agri-Footprint v2 (Blonk Agri-footprint BV, 2015). Therefore, we disregarded the emissions from transportation inside the farm and after the farm. These simplifications do not influence results, since the study area was constricted and is assumed to be the same for all products (all are representative of the same geographic area).

Besides different data sources and the scale adaptation carried out by one of the references of this study (Morais et al., 2016a), an additional important difference observed in the comparison presented in this study was the model used for GHG emissions. Both inventories use the same underlying method to calculate GHG emissions, i.e. IPCC (2006). However, Morais et al. (2016a) used it indirectly by resorting to the Portuguese National Inventory Report (NIR) (APA, 2014). The NIR, for some emission factors (e.g. N₂O emission factor due synthetic fertilizers application) suggests a different reference value compared to the IPCC (2006). Since Agri-Footprint v2 (Blonk Agri-footprint BV, 2015) used the reference value of IPCC (2006), this aspect can be relevance in GHG emissions and the GWP impact category.

5. Conclusions

LCI is the life cycle stage where more efforts are dispended in LCA studies. Regionalized inventories, which depict locally specific processes, facilitate the work of LCA practitioners, and are necessary to ensure inter-study comparability and accuracy. In this study we performed a comparison between a generic inventory (Agri-footprint v2) with other inventory adapted using highly specific and regionalized data for Portuguese products (Morais et al., 2016a), in agri-food sector. We concluded that, even in the GWP impact category, which is highly standardized, there are significant differences if the inventories are built according to different rules.

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POSTER PAPERS

PEF, LABELLING AND COMMUNICATION

8. PEF, labelling and communications

P46. Product Environmental Footprint Category Rules for Marine Fish products on the EU market – Experiences so far

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Introduction

This presents experiences from the development of a Product Environmental Footprint Category Rule (PEFCR) for marine fish products. A goal is that all products on the EU market shall have a documentation of their environmental footprint. To enable decision makers to make informed choices and to provide a trusted, transparent, holistic and scientifically based alternative to the eco labels and certification schemes that exist today.

Findings and results

The PEFCR development have provided insight into new and known challenges for more extensive use of PEF/LCA in the seafood sector:

- The lack of relevant LCI databases makes it difficult to set data quality and sampling requirements based on statistical parameters and to develop a PEFCR according to the requirements set by the EC.
- To secure the acceptance of a PEF as trustworthy and holistic, the methods and data to include biotic impacts from seafood production systems have to be improved.
- The default impact assessment method for a PEF (the ILCD method), have several errors and weaknesses in the handling of emissions to water. Also it is not harmonised with commonly used LCI databases.
- Despite the goal of enabling decision makers to make conscious choices, the PEFCRs for food commodities are not harmonised
- The screening analysis show that for seafood biotic carbon can be an important climate aspect. Also that packaging materials should be modelled with a higher precision on the material- and energy recovery rates than what is common today, the availability of relevant data makes this challenging.

The requirements for the PEFCR development are extensive and it can be questioned if some of them is an obstacle for more use of PEFs. At the same time the movement towards "a common green market" can be an important incentive to solve the challenges in this list.

P47. Communicating environmentally friendly food consumption to consumers

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Previous consumer studies have showed that Finnish consumers have knowledge gaps in understanding environmental impacts of foods. The Finnish food industry acknowledged that there is a need to move forward together to educate consumers. Thus, in Climate Communication III –project educational material is produced and consumer campaign conducted regarding environmentally friendly food consumption.

Firstly, a literature review was made of scientific articles on dietary choices and sustainable food consumption. Based on the review, first draft of the content of the communication was made. A workshop was organized for the industry and its stakeholders, where they expressed their views on the content and in addition, on the style of communication.

There was strong consensus, that communication works the best when it's positive and simple. Contradictory information should be avoided, for example positive and negative impacts of food choices on different environmental impact categories, and concrete and simple suggestions should be preferred.

Therefore, three key messages were formulated: 1) Add the share of vegetables, fruits, berries and grain products in your diet, 2) Avoid food waste, and 3) You can eat more environmentally friendly by doing good daily food choices. There are a good number of scientific articles on climate impacts of diets from different countries. Many of the recently published ones also include nutrition in their analysis and thus, provide a reliable base for communication to consumers of climate impacts. Fewer studies can be found on other environmental impacts, except a few, which focus on nitrogen footprint or eutrophication. From those few, though, similar conclusions can be drawn.

Despite uncertainties related to assessment of environmental impacts of food products and diets, the discussion in scientific literature show that it is important to start to communicate them to consumers as dietary changes would be the most efficient way to reduce environmental impacts of food.

P48. DECiDE, a tool for assessing greenhouse gas emissions and energy consumption

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Energy consumption and climate change are core issues, as is evident from the many initiatives at European, national and regional levels. The agricultural industry also accounts for 12% of anthropogenic greenhouse gas (GHG) emissions in our region. However, agricultural systems' energy consumption and GHG emissions vary considerably and depend very much on the types of farms and agricultural practices. The first step towards putting reduction measures in place therefore requires a sound knowledge of the relationship between practices, energy balances and GHG which, given the specific nature of the practices and the pedoclimatic contexts, requires regional benchmarks to be developed.

To meet that need CRA-W has developed a tool called DECiDE (**D**iagnostic **E**nergie-**C**limat **D**es **E**xploitations agricoles wallonnes), which is subsidized by Wallonia's Air and Climate Office (AWAC). The tool can be used to produce energy and GHG balances for Wallonia's agricultural systems. The object is to provide an open, transparent tool which is accessible to all via an Internet platform, aimed at farmers and institutions (research/administration/decision-makers), that can be used for (1) comparing one farm with other farms of the same type to show up differences and the practices that give rise to them; (2) advising on reducing energy consumption and GHG emissions; (3) supplying reliable benchmark values; and (4) quantifying the services rendered to society by the agricultural sector (carbon storage, renewable energy sources).

System boundaries take into account all the emissions from extraction of raw material to farm gate on a one year period (Table 1 and 2). Estimations of emissions are mainly based on the methodologies of IPCC and EMEP. DECiDE is composed of about 10 000 data. Most of them are adapted with specific Belgium soil and pedoclimatic conditions.

The results are expressed in MJ for energy and in kg eq. CO₂ for GHG.

It is hoped that using this tool will enable farmers to identify levers for improving their practices. The tool is currently designed for cattle rearing and arable farms, but is due to be extended to pig and poultry farming.

Table 1: Emissions items take into account into energy diagnostic

Energy	Items	Description
Direct energies	Fuel	Annual quantities of oil and gasoline bought, with deduction of fuel use from work for third party.
	Electricity	Annual quantity of electricity mix consumed
	Other fossil fuel	Annual quantity of other fossil fuel bought : gas and lubricant
	Renewable combustible	Annual quantity of renewable fuel bought
Indirect energies	Feeds	Feed bought
	Fertilizers	Fertilizers bought
	Pesticides (PPP)	Herbicides, fungicides, insecticides bought
	Seeds	Seeds bought

Table 2: Emissions items take into account for GHG diagnostic

Items	Gas take into account	Description
Electricity	CO ₂ eq.	Generation of electricity
	CO ₂ eq.	Production and transport of electricity
Fossil fuel	CO ₂ / CH ₄ / N ₂ O	Use of fossil fuel by the farmer
	CO ₂ eq.	Emissions from fossil fuel production (farm)
	CO ₂ / CH ₄ / N ₂ O	Use of fossil fuel for work for third party
Renewable fuel	CO ₂ eq.	Emissions from production of fossil fuel (third party)
	CH ₄ / N ₂ O	Use of biomass by the farmer
	CO ₂	Use of pure vegetable oil extracted on farm from cultivated grains
Renewable energy (except bioenergy)	CO ₂ eq.	Emissions from fabrication of RE (except biomass)
Equipment of refrigeration and air conditioning	HFCs et PFCs	Emissions from losses of refrigerant of the installations
Direct fields emissions	CO ₂ / CH ₄ / N ₂ O	Emissions linked to a change in farming practices
	N ₂ O + NH ₃	Emissions from spread of mineral fertilizers
	N ₂ O + NH ₃	Emissions from spread of organic fertilizers
	N ₂ O + NH ₃	Emissions from spread of dung and urine excreted on pastures
	N ₂ O	Emissions from residue of crops and symbiotic fixation

POSTER PAPERS

BIODIVERSITY

9. Biodiversity

P49. The role of pollination in the LCA studies of honey

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Life Cycle Assessment (LCA) has been increasingly used for the improvement of the environmental performance of products and services, amongst which the food systems. For this reason, a great number of food LCA case studies and reviews of case studies has been published in the scientific literature. Nevertheless, amongst the food products, honey appears to have been rarely analysed. Indeed, a preliminary literature review, carried out by the Authors, resulted in only two peer-reviewed scientific papers (one of which, was on carbon footprint). Honey is one of the natural products, which is considered to have a great range of benefits for the human health.

Besides that, the role of honey bees as pollinators can be regarded as one of the functions of an apiculture system and is undoubtedly of utmost importance both for natural ecosystems and agriculture. The main objective of this paper is to explore this role, by means of a literature review, as one of the apiculture system functions. Furthermore, the management of this multifunctional system in honey-related LCA case-studies, is addressed.

The literature review was performed using the EBSCOhost Discovery service, using a combination of keywords, such as “honey” AND “pollinat*”, “pollinat*” AND “economic value”, “LCA” AND “pollinat*”, etc. This resulted in 9 articles. The examined papers showed that pollination regards an issue that has not been tackled with so far in LCA case studies. Indeed, none of these papers tackled this issue. On the other hand, the issue of the economic evaluation of the pollination service was found to have been fairly examined. Indeed, several papers (9 articles) discussed its economic value. This paper proposes the inclusion of the pollination service as one of the functions of a multifunctional system in LCAs (other functions including, e.g., the provision of honey, beeswax, etc.) and provides some insight for the management of the multi-functionality of this kind of system in LCA studies. The discussed economic allocation, indeed, can be important as the provided economic value of the pollination can be comparable to -and in some cases even higher than (estimated to be even 8 to 10 times higher)- the one of the main product (honey) itself.

P50. Capturing the benefits of responsible forestry practices in LCA: focus on biodiversity

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The benefits of using responsibly sourced products are still difficult to quantitatively capture in the context of LCA in particular with regards to their benefits for biodiversity. Nestlé (client), UPM (supplier), and Quantis (LCA consultant), have developed an approach to quantify the relevant differences between conventional and responsible forestry practices for LCA indicators such as ecosystem quality (PDF.m2.y PDF being potentially disappeared fraction of species), land use (m2.y), and GHG emissions. The study is for one cubic meter of wood, at mill gate, in Finland, and encompasses forestry management, logging, logistics until the mill and the differences in energy inputs and outputs for heat recovered from wood residues in the mill. The steps after the mill are not considered. The method for biodiversity accounts for four indicators, native tree species composition, deadwood volume and quality, protected valuable habitats, and forest structure, that are grouped into one indicator between 0 and 1. The results show that responsible practices have consistently lower impacts than conventional practices. For example, the impacts on ecosystems quality for responsible forestry practices are about half of those for conventional practices.

This method can objectively capture the benefits of biodiversity protection in wood fibre production. Companies can use it in complex LCAs to consistently quantify impacts and benefits in supply chain. This method can be used to communicate externally about the benefits of biodiversity protection associated with responsible wood sourcing within an LCA context in a more robust way than what is done until now.

POSTER PAPERS

SEAFOOD AND AQUACULTURE

10. Seafood and Aquaculture

P51. Life Cycle Assessment of Ecuadorian processed tuna

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Objective

Ecuador is an important player in the global tuna fishing and processing industry: the Ecuadorian industrial tuna fleet represents 17% of the global tuna purse seiner fleet, and it is the second largest tuna processing country after Thailand. The fishing and processing operations of one of the largest vertically integrated tuna processing firms in Ecuador were evaluated regarding their environmental impacts, and assumed representative of the Ecuadorian tuna processing industry. Results were compared with those of other international fish processing and other sources of animal protein for human consumption. Directions are finally identified towards reducing environmental impacts of both the tuna fishery and processing industry.

Method

Detailed operational fishery and processing data was collected from a representative Ecuadorian tuna processing firm, and the life cycle assessment framework applied to it for identification of hotspots. Two functional units were used: 1 tonne of final product (for canned, pouched, vacuum bagged and 'average' products) and 1 tonne of "fish in product", which includes all process losses and normalises the final product:raw fish ratios among the different processing routes analysed. The ReCiPe impact assessment method was used, including all midpoint and endpoint impact categories. Impacts were allocated by mass between tuna products and residues (which are rendered into residual fishmeal). The system boundary included the construction, use, maintenance and end-of-life of the tuna fishery until the landing port, and those of the construction, use and maintenance of the processing plant, from fish landing until the storage of final products. Primary data were collected only for use and maintenance of both fishing vessels and processing plants.

Main result

In the period 2012-2013, the studied sub-fleet featured a fuel use intensity of 835 L per landed tonne (Fig. 1), which was 235% higher than reported values for all tuna landings in the Pacific Ocean in 2009. Reasons for such underperformance may include inter-annual variations in tuna catchability and the fact that fuels are generally subsidised in Ecuador, and thus skippers perhaps do not apply sufficient fuel-saving strategies. The main contributors to impacts associated with tuna processing were the provision of tinplate cans (58.0% of the ReCiPe single score) and fuel use by the fishery (22.6%). Ecuadorian tuna products feature environmental impacts generally higher than those of other fish processing industries worldwide, yet lower than those of many alternative sources of fish and land animal protein (Table 1).

Implications

Efforts to reduce environmental impacts of Ecuadorian tuna processing should focus on the fuel performance of the providing fleet, and on the container technology. Increased use of larger tinplate cans, aluminium cans, or other non-metal container technologies (e.g. pouches, retort cups) would decrease environmental impacts of tuna processing. The sources of relative inefficiency observed for the Ecuadorian tuna fleet should be thoroughly investigated. Possible solutions could involve applying fuel saving strategies.

Keywords: canning; Ecuador; fuel use intensity; tuna

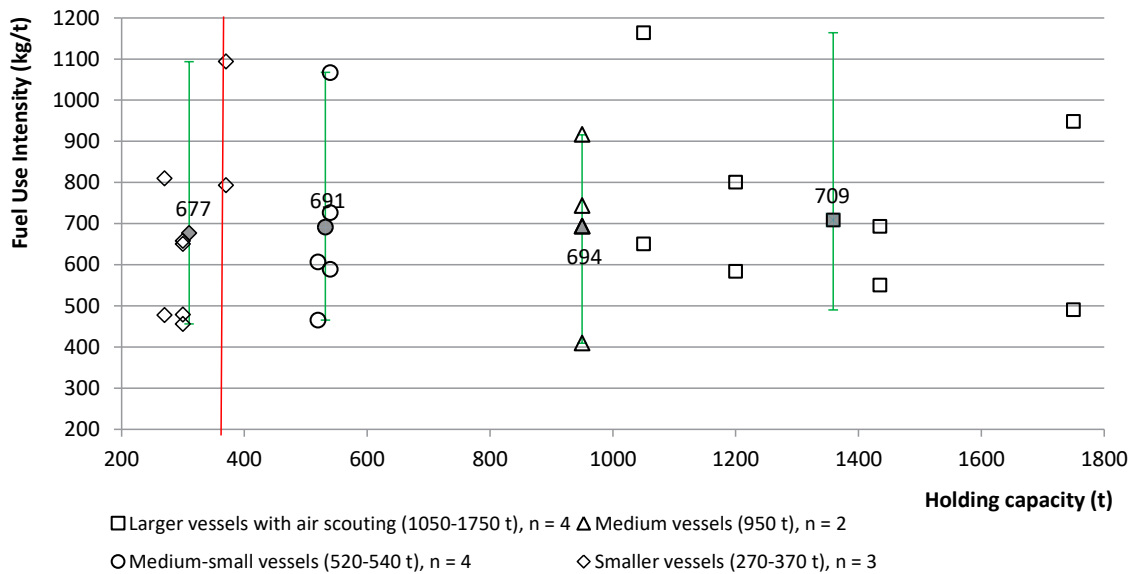


Fig. 1. Fuel-use intensity of four segments of the tuna fleet (sample n = 13 purse seiners with 25 FUI/year/vessel data points) for the period 2012-2013. Grey-fill symbols represent landings-weighted fuel use intensity per segment; the vertical line represents the lower limit of the upper class of the official Inter-American Tropical Tuna Commission industrial tuna purse-seiners classification, and error bars represent the range of FUIs within each segment. Density of marine diesel: 0.832 kg/L

Table 1. Comparison of climate change impacts (kg CO₂ eq) per kg product and kg protein of Ecuadorian tuna products and other animal products from global supply chains. Sources listed in Avadí et al. (2015)

Product	Protein content (%)	Impact per kg product	Impact per kg protein
Ecuador: Canned tuna in vegetable oil	26.5	3.7	14.0
Ecuador: Pouched loins	28.2-29.2	2.7	9.3-9.7
Ecuador: Bagged (frozen) loins	22.0-24.4	3.1	12.9-14.3
Peru: Canned anchoveta in vegetable oil	21.3	1.7	8.1
Peru: Fresh cultured tilapia	18.3	1.9-4.1	10.4-22.4
Peru: Fresh cultured trout	18.4	2.8-3.4	15.2-18.5
Portugal: Canned tuna in olive oil	26.5	7.7	29.1
Portugal: Frozen tuna	22.0-24.4	1.0	4.1-4.5
Spain: Canned tuna in tomato sauce	20.8	2.5	12.1
International: Various animal protein sources, without packaging (Nijdam et al., 2012)			
Beef (studies = 15, products = 26)	20	9-129	45-640
Pork (studies = 8, products = 11)	20	4-11	20-55
Poultry (studies = 4, products = 5)	20	2-6	10-30
Eggs (studies = 4, products = 5)	13	2-6	15-42
Milk (studies = 12, studies = 14)	3.5	1-2	28-43
Cheese (based on milk studies)	25	6-22	28-68
Seafood from fisheries (studies = 9, products = 18)	16-20	1-86	4-540
Seafood from aquaculture (studies = 7, products = 11)	17-20	3-15	4-75

Notes: Protein content values from the USDA National Nutrient Database for Standard Reference Release 27 <http://ndb.nal.usda.gov/ndb/foods> (except for Peru; values based on measurements).

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P52. Achieving a more sustainable production and consumption of the Cantabrian canned anchovy: environmental assessment under a life-cycle perspective

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The quality and prestige of the canned anchovies is of particular relevance in the Cantabria region. However, its manufacture generates a high amount of effluents, fish residues and packaging waste. Therefore, it is necessary to design and implement strategies for the sustainable production and consumption of this food product.

This work aims to identify the main hot spots of the production of one can of anchovies in extra virgin olive oil (EVOO) under a life cycle approach. The use of life cycle assessment (LCA) methodology will help to minimize the consumption of natural resources (NR) and thus, the environmental burdens (EB) of this food product that is worldwide consumed. LCA was conducted from cradle to grave. The cradle to gate (Cr-Ga) stage included the production and transportation of raw materials and packaging. The gate to gate (Ga-Ga) stage comprised the anchovy transformation, whereas the gate to grave (Ga-Gr) stages considered the distribution of the product in Cantabria, the use and end of life. The results showed the whole life cycle of one can of Cantabrian anchovies in EVOO generated 0.18 kg CO₂ eq. In particular, the Cr-Ga processes displayed the greatest contribution, this stage consumed 77 % of the total energy, 83 % of the total materials and 99 % of the total water (Table 1). This was mainly due to the production of the aluminum can and, to a lesser extent, the production of EVOO. Moreover, Figure 1 displays that the production of the aluminum can had the highest environmental impacts in the categories of AA (88 %), GW (83 %), HEE (99 %), POF (83 %), SOD (53 %), AOD (70 %), NMEco (38 %) and EU (76 %). The treatment of fish solid residues, the distribution, use and end of life of canned anchovies are processes that also need to be improved.

This work allows achieving a more sustainable production and consumption of the Cantabrian canned anchovies promoting circular economy by means of the valorization and minimization of waste along the supply chain of the product.

Table 2. Natural resources consumption in the life cycle of one can of Cantabrian anchovies in extra virgin olive oil.

	Energy (MJ)	Materials (kg)	Water (kg)
Cradle to gate	2.20	0.66	793
Gate to gate	0.37	0.13	8.96
Gate to grave	0.72	0.18	16.2
Total	3.38	0.96	819

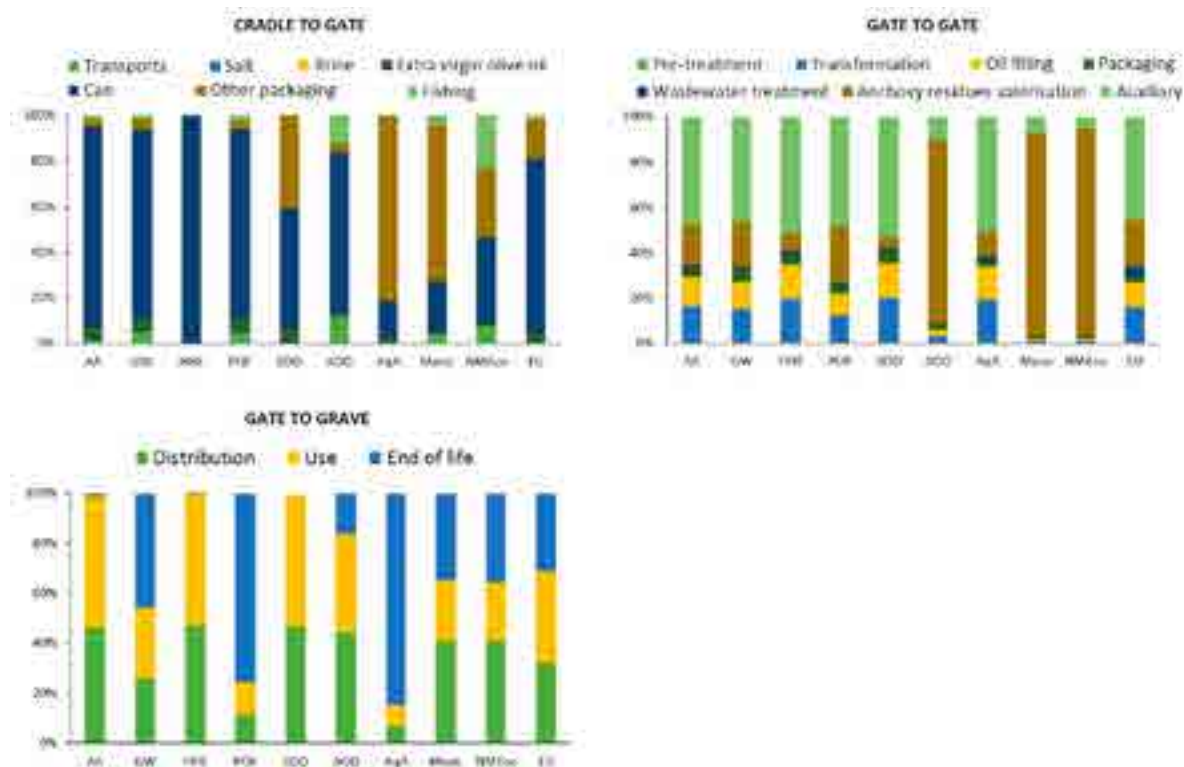


Figure 2. Contribution of each process to the environmental burdens in the Cr-Ga, Ga-Ga and Ga-Gr stages. AA: atmospheric acidification; GW: global warming; HHE: human health effects; POF: photochemical ozone formation; SOD: stratospheric ozone depletion; AOD: aquatic oxygen demand; AqA: aquatic acidification; MEco: ecotoxicity to aquatic life (organics); NMEco: ecotoxicity to aquatic life (metals); EU: eutrophication.

P53. Life cycle assessment of Tilapia (*Oreochromis niloticus*) and Brazilian Sorubim (*Pseudoplatystoma spp.*) using animal co-product based feed.

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Brazilian fish production has been growing fast. National production increased 20% in the 2013-2014 crop, reaching ~ 474,000 tonnes. Despite economic and social benefits from fish farming, the environmental impacts that the activity may cause are raising concerns among stakeholders. We used Life Cycle Assessment (LCA) to evaluate the environmental impacts of tilapia and sorubim from a 200 ha commercial fish farm in Brazil's Midwest Region.

We conducted a detailed inventory at the fish farm and feed supplier. The feed ingredients inventory (agricultural and processing phase) came from nationally published data. Otherwise, background data from Ecoinvent 3.0 and Agri-footprint® databases was adapted. We used SimaPro® 8.2 to run the data using CML-Baseline 3.02 world 2000 and Cumulative energy demand (CED) methods. The mass-balance methodology was according to Henriksson (2015). A cradle-to-farm-gate approach was adopted, the infrastructure and equipment have not been accounted for. Moreover, the functional unit used was one tonne of live weight fish and we used mass allocation. The impacts categories and the results are presented in Table 1. Sorubim performed worse than tilapia in all categories, this was mainly due to the higher feed consumption of 2.1 tonnes to Sorubim versus 1.6 tonnes to Tilapia and harvest size of 1.3 kg to Sorubim versus 0.76 kg for tilapia. It should be noted that Sorubim is a carnivorous fish and requires feed with high protein. Table 2 shows the ingredients and formulations used at the farm whilst Table 3 presents the related impacts to produce 1 tonne of each feed. Sorubim used a mix of the four feeds and Tilapia used mainly feed of 32 % crude protein. Brazil has a great supply of animal co-product meals and the use of animal meal in fish feed has benefits, such as avoiding using high-quality fishmeal. However, our study shows that this practice increases the environmental impacts of fish production (Figure 1), primarily due to the embodied emissions that animal meals already carry from their origin and secondarily because the animal co-products have high levels of phosphorus and relative low digestibility, increasing pond emissions and reducing fish growth performance. Animal co-product emissions contributed respectively for Tilapia and Sorubim production with 4,500 and 6,720 Kg CO₂ eq; 23 and 71 kg SO₂ eq; 29 and 49 kg PO₄ eq. We strongly advise further investigation into the impact of using animal co-product meals to feed fish.

Furthermore, economic issues should be taken into account, because of its great role when choosing feed ingredients.

Table 3- Life cycle impact associated with the production of one tonne of live Sorubim and Tilapia.

Impact category	Unit	Sorubim			Tilapia		
		Total	Pond	Feed	Total	Pond	Feed
Global warming (GWP100)	kg CO2 eq	9,776	1,176	8,600	6,562	756	5,833
Acidification	kg SO2 eq	90	4	86	37	3	34
Eutrophication	kg PO4 eq	178	123	55	112	65	36
CED	GJ	27	1	26	20	1	19

Table 4 - Ingredients used to produce one tonne of feed with 40%, 38%, 36% and 32% of crude protein

Ingredients	Feed 40	Feed 38	Feed 36	Feed 32
Soybean meal 45%	100	50	166	303
Whole maize meal	120	140	162	200
Wheat meal	120	170	150	200
Rice broken	50	50	50	
Meat/bone meal	193	173	168	220
Fish meal mix	130	130	120	
Blood meal	50	50	50	50
Offal meal	150	150	80	
Feather meal	50	50	22	
Soybean oil	20	20	15	10
Salt	5	5	5	5
Minerals/vitamins*	12	12	12	12

*Not accounted for the present study.

Table 5 - Life cycle impact associated with the production of one tonne of feed 40%, 38%, 36% and 32% of crude protein.

Impact category	Unit	feed 40	feed 38	feed 36	Feed 32
Global warming (GWP100)	kg CO2 eq	4,468	4,225	3,687	3,612
Acidification	kg SO2 eq	48	47	32	20
Eutrophication	kg PO4 eq	29	28	23	22
CED	GJ	13	12	12	12

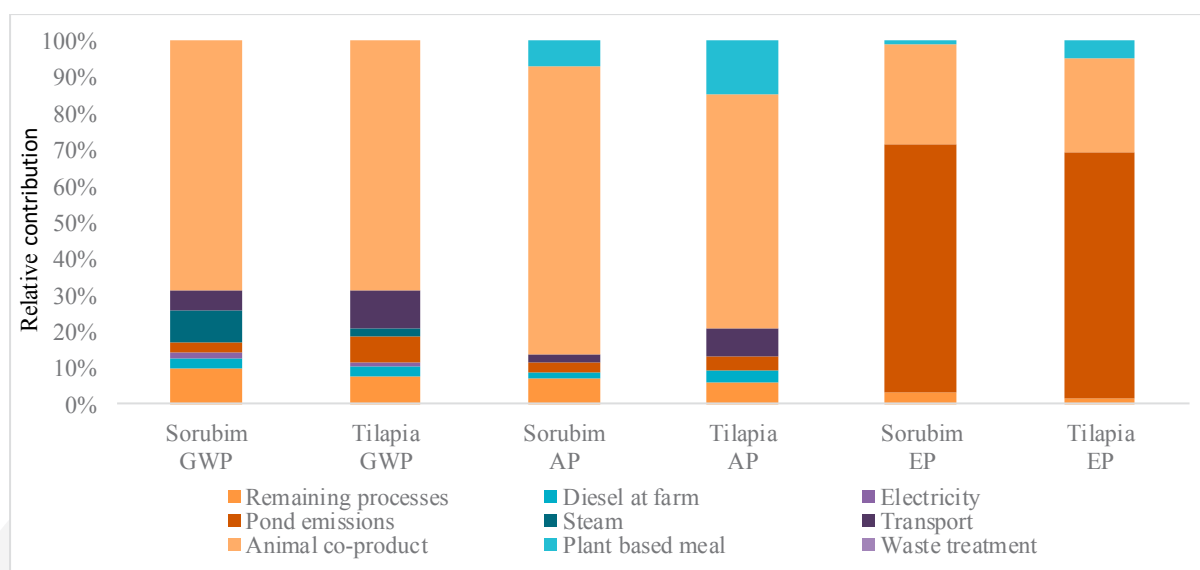


Figure 3 - Relative contribution of systems processes to total impacts associated with one tonne of Sorubim and Tilapia production, 1% cut-off to remaining processes was adopted.

Reference

HENRIKSSON *et al.* Comparison of asian aquaculture products by use of statistically supported life cycle assessment.: **Environmental science & technology**, 2015. v. 49, n. 24, p. 14176–14183.

P54. Environmental assessment under a life cycle approach of the Cantabrian anchovy (*Engraulis encrasicolus*) season in the Cantabria region (Northern of Spain)

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The anchovy canning industry has high importance in the Cantabria Region (Northern of Spain) from the economic, social and touristic point of view. The Cantabrian canned anchovy is world-renowned due to its handmade and traditional manufacture. Therefore, it is necessary to design and implement strategies for the sustainable production and consumption of this food product. In this context, the Life Cycle Thinking (LCT), which promotes the inclusion of environmental, social and economic impacts of a product over its entire life cycle, has already been applied to the manufacture of the Cantabrian canned anchovies and to the valorisation of anchovy residues. However, the fish extraction has not been studied yet. Furthermore, fishery studies have been based on relatively short periods of time owing to the difficulty of obtaining thought inventory data for prolonged period of time.

Therefore, the objective of this work was to evaluate the environmental impact related to fish extraction on a temporal basis in order to analyze the effect that stock abundance variations may have on reporting environmental burdens. Figure 1 shows the system under study. Inventory data for the fishing season were collected over a 5-year period and used to carry out a life cycle assessment (LCA). The selected fishery corresponds to the Cantabria coastal purse seining fleet (Figure 2).

The functional unit (FU) considered in this work was set as 1 t of landed anchovy during the fishing season for each of the selected years. The selected data for the life cycle inventory were gathered from personal communication from CONSESA (Canners Association of Santoña) and from a fish first sale register in the Cantabria Region. A series of fishery-specific impact categories and indicators were included in the evaluation together with conventional impact categories.

Conventional LCA impact categories showed that the environmental impact is dominated by the energy use in the fishery, despite of the low fuel effort identified with respect to other fisheries. Nevertheless, strong differences were identified between annual environmental impacts, attributed mainly to remarkable variations in anchovy stock abundance from one year to another. Fishery-specific categories, such as the discard rate or seafloor impact showed reduced impacts of this fishery. Finally, the fishery in balance (FiB) index identified the evolution of anchovy stock abundance for this particular fishery.

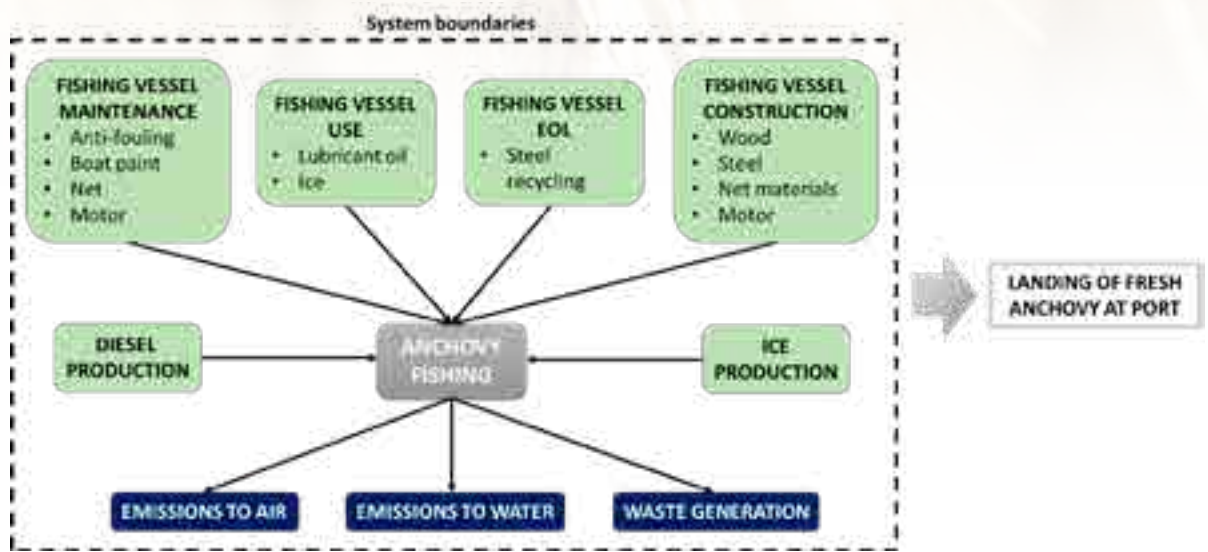


Figure 4. Inputs and outputs included in the Cantabrian anchovy fishing.



Figure 2. Cantabria coastal purse seining fleet.

P55. Life Cycle Inventory (LCI) of French fishery products

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For several years, the entire French fishing industry has been mobilizing to define an economically, socially and environmentally sustainable fishery. The 'Sea product LCI' project fits into the framework of the Agribalyse program, which aims to build a database for the environmental assessment of agricultural and food products (cereals, livestock etc...) through the Life Cycle Assessment (LCA). In our project, two major objectives are pursued. First, it aims at improving knowledge and methodologies on environmental impact assessment of fishery products via LCA. Secondly, it consists in producing LCI of some representative products of French fisheries in order to include them in the Agribalyse database.

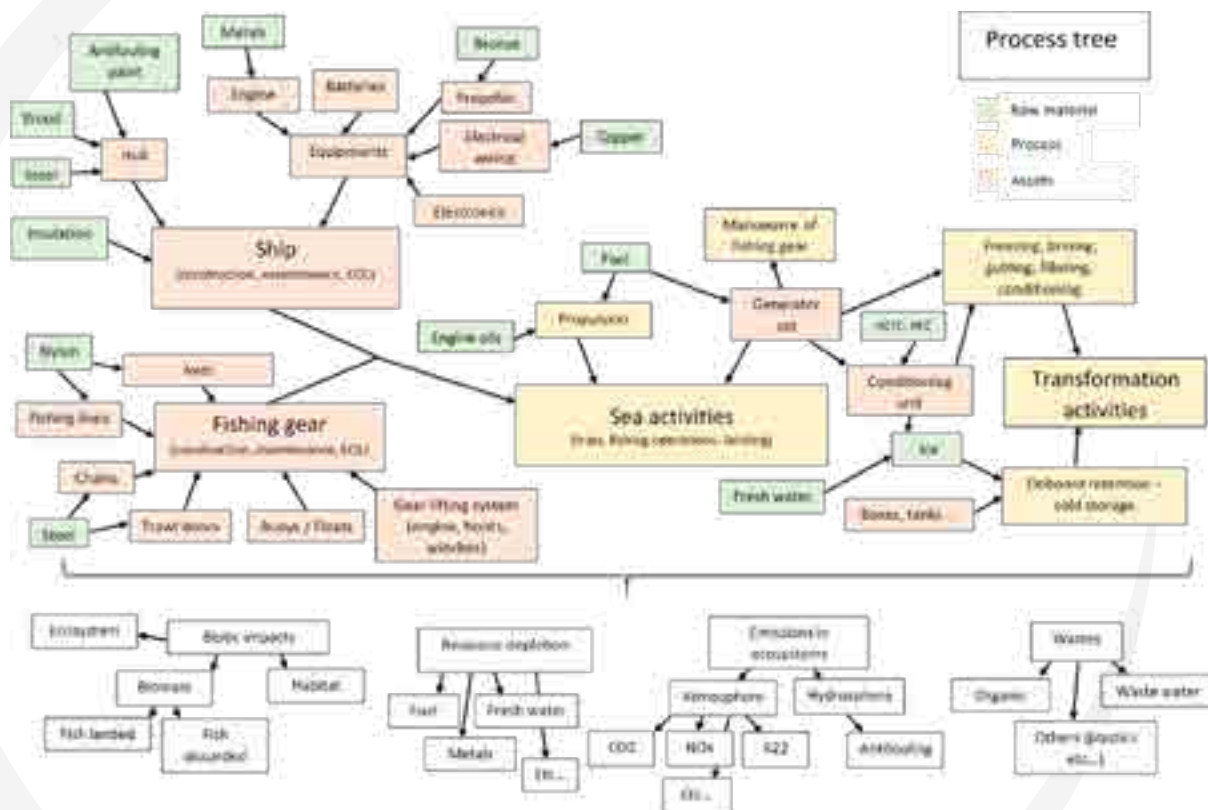
In this context, the project partners have sought to identify a sample of products representative of the diversity of French fishing activities and environmental constraints. Fifteen "triplets", resulting from the combination species/fishing area/fishing gear, were identified for study (Table 6). The analysis focuses on a cradle-to-gate approach, thus ranging until landing of the fish. The study stresses on impacts related to the construction, maintenance and end-of-life of ships and fishing gears, fishing operations and onboard transformation (Table 7). Besides the usual environmental impacts (energy use, climate change, acidification etc...), some impacts, such as impacts on benthic habitats or on fish stocks, are specific to fishing and raise specific methodological issues. Regarding LCA methodology, consistency with the PEF initiative of the European Commission will be sought. The expected results of this project are (1) the construction of environmental assessment tools that could be used by different actors of the French fishing industry to carry out environmental assessment, (2) allowing a reflection on the ways of improvement of the sector, (3) the communication on different iconic products of the fisheries sector in France as part of AgriBalyse database. The large cooperation between professionals, technicians and scientists will ensure the collection of quality data and the development of relevant impacts indicators for fisheries. This study is an important step forward for the fishery sector to

acquire benchmarks and more broadly for food LCA community, considering that currently hardly any fishery data are available in public LCI databases.

Table 6 : List of the fifteen « triplets » studied

Species	Fishing area	Fishing Gear
Scallop	The Channel	Dredge
Gadidae (cod, haddock, whiting)	Celtic Sea	Benthic trawl
Herring	North-East Atlantic	Pelagic trawl
Saithe	North Sea	Benthic trawl
Mackerel	North-East Atlantic	Pelagic trawl
European pilchard	Eastern Central Atlantic	Seine
European pilchard	Bay of Biscay	Seine
Sole	Bay of Biscay	Fishing net
Albacore tuna	North-East Atlantic	Pelagic trawl
Atlantic bluefin tuna	Mediterranean sea	Longlines
Atlantic bluefin tuna	Mediterranean sea	Seine
Yellowfin tuna	North-East Atlantic	Seine
Yellowfin tuna	Indian Ocean	Seine
Skipjack tuna	North-East Atlantic	Seine
Skipjack tuna	Indian Ocean	Seine

Table 7 : Process tree of the LCA



POSTER PAPERS

DIET AND NUTRITION

11. Diet and Nutrition

P56. A Combined Nutritional and Environmental Life Cycle Assessment of Average Canadian Diet and Additional Pulse Servings

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Abstracts

Sustainable diets are defined as “those diets with low environmental impacts which contribute to food and nutrition security and to healthy life for present and future generations” (FAO 2010). Pulses are highly regarded as an ideal candidate for sustainable diets because pulses contribute to lower environmental impacts due to less fertilizer requirement compared to other crops and pulses are a low fat source of protein, with a high fibre content and a low glycaemic index which help improve metabolic control and decrease risk factors for some non-communicable diseases including cardiovascular disease, obesity, diabetes and colon-rectal cancer. The 2016 International Year of Pulses (IYP 2016) is promoting awareness of nutritional, health and environmental benefits of pulses.

Environmental benefits of pulse crops in crop rotation are confirmed in many agricultural life cycle assessments. Similarly, consumption of pulses is widely acknowledged to be a well-balanced and healthy diet providing the benefits of maintaining health and preventing non-communicable diseases in clinical nutrition studies. However, a combined assessment of nutritional, health and environmental benefits of pulses in a diet has not been studied yet. Therefore, the nutritional, health and environmental effects associated with additional pulse servings to an average Canadian diet were assessed using a combined nutritional and environmental life cycle assessment (CONE-LCA) developed by Stylianou et al. (2015).

Data on loss-adjusted food availability from Statistics Canada were used to develop the average Canadian diet. Greenhouse gas emissions (GHGE) and particulate matter emissions of the average Canadian diet were estimated from existing food LCA studies. The majority of data on food and food groups were drawn from Canadian studies including wheat flour, apples, potatoes, tomatoes, meat and alternatives, milk and dairy products, fats and oils, sugar-sweetened

beverages and hot drinks. Since the United States of America accounted for 43% of fresh fruit import and 77% of fresh vegetables import (Statistics Canada 2012), the rest of fruits and vegetables were used from the U.S. food LCA study except for banana and pineapples. Data on banana and pineapples were used from Costa Rica LCA study.

Nutritional effects of additional pulse serving were estimated using data from nutritional and epidemiological studies and reports on Global Burden of Disease. Potential human health impact related to greenhouse gas emissions, particulate matter and nutrition effects of additional pulse serving to the average Canadian diet was expressed in Disability Adjusted Life Years (DALYs).

Global warming potential and respiratory organics of the average Canadian diet (2000 kcal/person/day) were 2.69 kg CO₂-eq/person/day and 1.8 g PM_{2.5}-eq/person/day. Potential human health impact of the average Canadian diet was 23.6 μDALYs. One serving of Canadian field peas (147 g) had a nutritional energy content of 173 kcal, accounting for 0.046 kg CO₂-eq and 0.024 g PM_{2.5}-eq. Potential health impact of one serving of field peas was 0.0394 μDALYs. One serving of Canadian field peas contributed to health benefits of 1.69 μDALYs, resulting from a benefit of 0.91 μDALYs for decreasing a risk of colorectal cancer and a benefit of 0.78 μDALYs for decreasing a risk of coronary heart disease. Results suggested that additional serving of Canadian field peas to the current average Canadian diet could result in net health benefits for Canadians.

P57. Protein2Food – a EU Horizon 2020 project on novel plant-based protein-rich food for fostering sustainable food consumption

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PROTEIN2FOOD is an *EU Horizon2020* funded five years project which started in April 2015. The poster presented at *LCA Food 2016* will provide an overview of the objectives and the structure of the project as well as – where possible - some first insights obtained during the first year.

PROTEIN2FOOD aims at developing innovative, cost-effective and resource-efficient food crops that are high in protein content, have a positive impact on human health, the environment and biodiversity. This shall be achieved by significantly enhancing the quality and quantity of proteins from selected seed crops (quinoa, amaranth and buckwheat) and grain legumes (lupine, faba bean, chickpea and lentil), by using a multi-disciplinary approach, involving genetics, agronomy, and food-processing engineering, as well as sensory, socio-economic and environmental assessments.

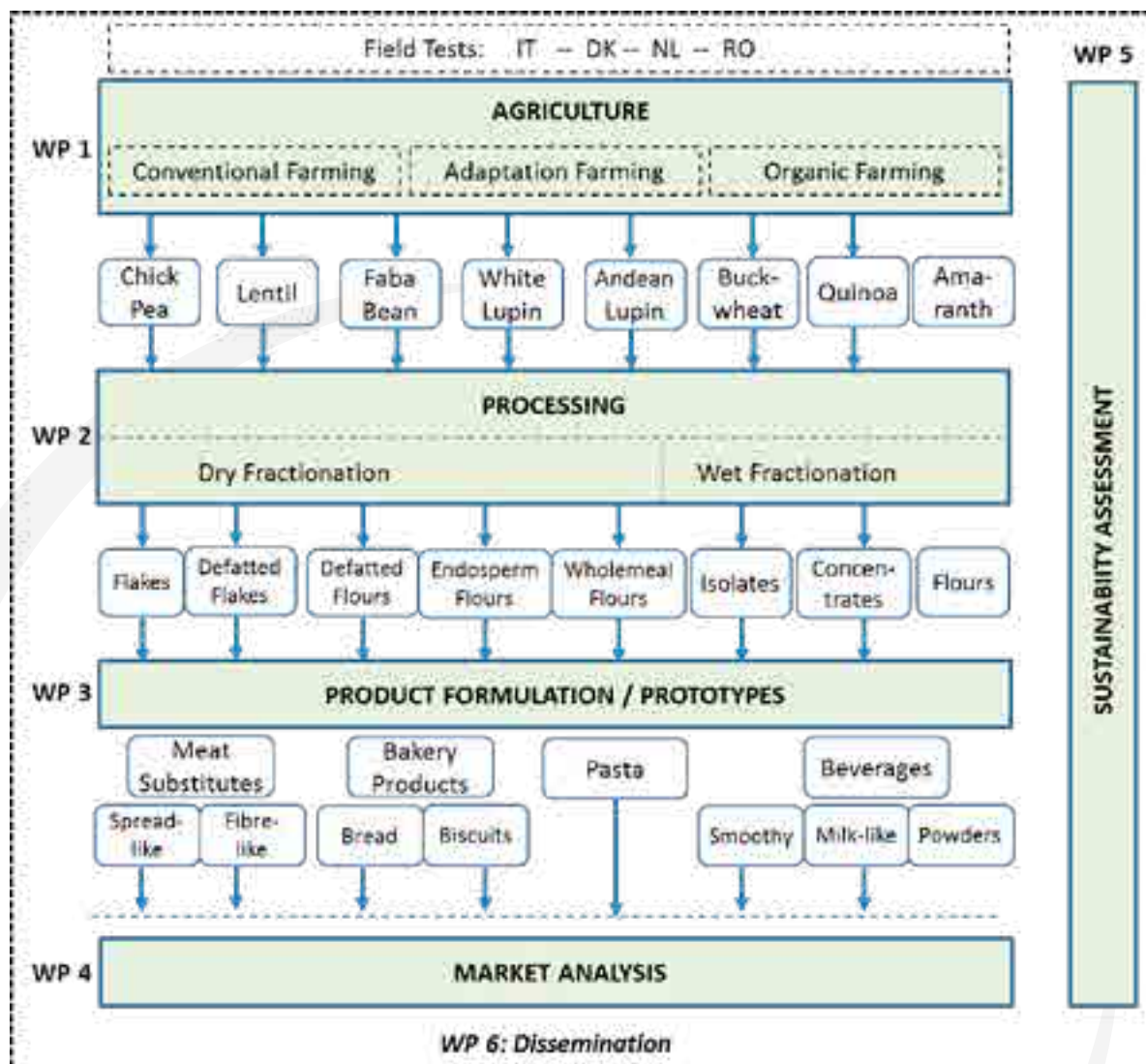
The project seeks to produce prototypes of a new product range of vegetarian products with high consumer acceptance, such as protein rich pasta and bakery products, vegan spreadable meat alternatives, extruded products (breakfast cereals and meat analogues) and infant food (see figure 1). It also comprises a sustainability assessment based on LCA and a socio-economic assessment. For this purpose several data and methodological issues have yet to be solved, like for instance:

- How to obtain adequate process data for protein extraction (e.g. protein isolates, protein concentrates) and protein-rich food production which are scarcely available, both in technical as well as in LCI literature
- How to transfer process data collected at laboratory or pilot scale by the project partners into LCI data that reflect commercial scale production
- How to benchmark the sustainability performance of the novel food product lines provided by PROTEIN2FOOD selecting
 - o a meaningful Functional Unit

- suitable reference products already today available in the stores
- The choice of socio-economic indicators relevant for and measurable at EU and global level

Furthermore, the poster will provide information on how to get connected with the project and/or the project partners.

Graph 1: Protein2Food project structure (WP= work packages)



This project has received funding from the Europeans Union's Horizon 2020 research and innovation programme under grant agreement No 63572

P58. Integration of nutrition as functional unit in Life Cycle Assessment

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Introduction

Defining the functional unit (FU) for agricultural systems is complex and its choice may influence results and conclusions. Recently, LCA studies have aimed at linking sustainable food supply with healthy eating and nutrition on the consumer side. While promising, such quantification of a food, meal, or diet through a nutrition-related FU may also result in misrepresentation and subsequent misinterpretation. Further, evaluating consumers' dietary intake is complex as food, meals, and diets serve multiple functions in a person's life. Finally, issues also exist in how nutritional data are derived, and this is ultimately dependent on the goal of the study.

Purpose

The purpose of this paper was to evaluate the usefulness of nutrition as FU in LCA and provide recommendations for the integration of nutrition as FU in LCA.

Methods

A literature review was conducted to synthesize advantages and disadvantages of using nutrition as FU in LCA. A three-prong approach was taken to 1) summarize methodologies commonly used as quality assessment of a food, meal, or diet, 2) review the studies that have attempted to integrate the concept of nutrient density, such as nutrient profiling, in LCA and 3) evaluate studies that have used composite health indicator scores and clinical endpoints linked to LCA.

Results

The application of nutrition as FU in LCA beyond what's considered common (e.g, single nutrient or energy), e.g., with the application of a diet quality score or nutrient profiling, may minimize nutritional reductionism but may be difficult to interpret and if not constrained (e.g., capped) to daily values, may misrepresent data and lend itself to misinterpretation. Recent approaches that express nutrition as part of a health indicator score or by clinical endpoints may be more useful to express diet's multi-functionality.

Variables	Description	Advantages	Disadvantages
Mass, Volume	Commonly used FU in LCA.	If adjusted for protein it accounts for quality.	Production at farm does not equate to what is consumed.
MJ, 100 kcal 1 kg, 100 g	Commonly used FU in LCA.	Quick and practical.	May under-represent meat compared with fruit and vegetables or vice versa.
Energy density kcal/100g	Commonly used in dietary studies.	Raises with fat and sugar and decreases with fiber and water.	Quality of fat is not differentiated. Sugary drinks have low energy density. May be added as variable but should not be used alone.
Protein, g	Commonly used in dietary studies.	Quick and practical in comparisons of animal and plant products in LCA.	Plant-based comparison needs to be adjusted to account for lower bioavailability of plant protein. Total protein should be capped to avoid over-representation of nutrients in animal protein, falsely elevating nutrient quality.
Nutrient profiling, nutrient quality, nutrient density indices	Arithmetic mean of nutrients in food, meal, diet. Expressed per 100 kcal or 100 g. Commonly, 6-9 positive and 3 negative nutrients to calculate score.	Considered less reductionist than representation via single nutrient. Literature already exists and indices are used in LCA studies. Can be used as FU or as a variable.	Time consuming. Nutrient density sometimes associated with higher GHG. Results vary based on index. Fruit and vegetables come out short due to nutrient underrepresentation (e.g., phytochemicals). Difficult to interpret. Essential to cap nutrients by daily value. Various other limitations.
Nutritional and Health indicators (composite scores)	Originated with healthy eating index to capture quality of people's diets. Developed based on dietary recommendations. Includes both nutrients and food and may add additional constructs such as economic impact.	Allows scientists to compute health and sustainability indicators, capturing multi-functionality. Considered less reductionist. May be expanded to include other functional aspects of nutrition such as social or psychological aspects.	Not used directly as FU. May still underrepresent nutrient density and multi-functionality of diets.
Physical endpoints (DALYs, morbidity, mortality)	Collected from epidemiological studies.	Provides tangible outcomes that relate to health and disease.	Not used directly as FU. Built on reductionism in nutritional sciences. May miss other aspects of diet and lifestyle.

Table 1. Advantages and disadvantages of using nutritional parameters as FU in LCA

P59. Mapping of food waste prevention actions into the Food Supply Chain for Life Cycle Management strategies

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Objective

The objective of the work is to map existing actions which prevent food waste (FW) along the Food Supply Chain (FSC) of food products, from a Life Cycle Management (LCM) perspective. The actions will be classified and analyzed in relation to different aspects: 1) in which stage of the food supply chain food waste is prevented; 2) type of action; and 3) environmental, social and economic consequences. Results are intended to give general trends about where and how these strategies are being implemented; and serve as a first step on impact pathways guidance for analyzing sustainability impacts of a prevention action. This would help design and assess LCM strategies in the food sector in terms of avoiding environmental impacts and adding social and economic positive impacts.

Methodology

Prevention of food waste activities databases (ReFED, 2016; Fusions, 2016; MAGRAMA, 2014)¹ were used as main source –complemented with Google tool- as well as for defining stages, type of action and consequences. Geographical coverage was worldwide and the languages of search were English, Portuguese and Spanish. This covered a period of two years (2014-2016), at the time of publication (the initiative itself could be older). The review is not intended to have absolute completeness or representativeness, but rather to give an idea of which are the mainstream activities designed worldwide to reduce food waste.

Results

¹ ReFED (2016) www.refed.com

Fusions (2016) Social Innovation Projects www.eu-fusions.org/index.php/social-innovations

MAGRAMA (2014) Catálogo de iniciativas nacionales e internacionales sobre el desperdicio alimentario

Stages of the supply chain, type of action and consequences are represented in figures 1, 2 and 3. From an initial 340 results found, 289 strategies were selected after eliminating duplicates, redundant or out of scope results (193 in Europe, 41 in North-America, 30 in South-America, 4 in Africa, 4 in Asia, 3 in Oceania and 9 worldwide).



Figure 1. Stages of the supply chain considered

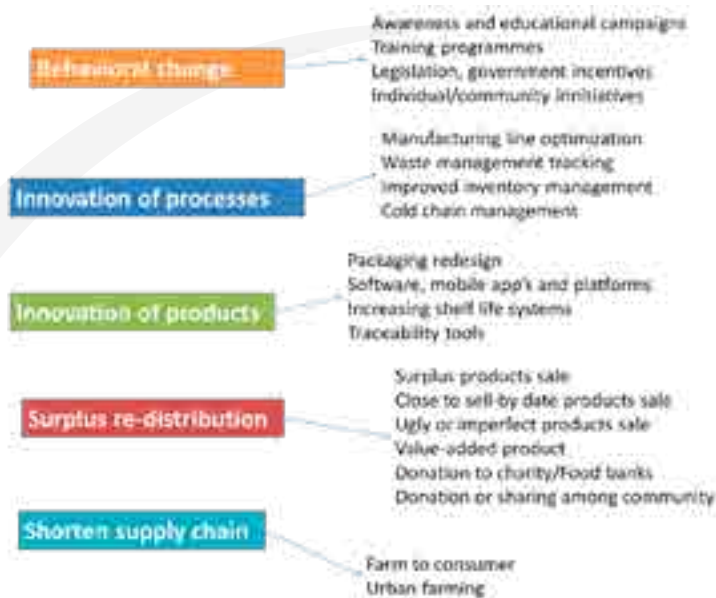


Figure 2. Type of initiatives classification considered in this study



Figure 3. Type of environmental, social and economic consequences considered in this study

P60. Macro-Environmental Analysis Of EU Food Supply Chain Losses Via Lca

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OBJECTIVE

The study aims at quantifying and evaluating the environmental sustainability of food waste (FAO 2011) associated to food production and consumption in the EU.

METHOD

The approach used in the study includes: the analysis of the typical European food consumption via the identification of an EU basket of products; a calculation of the loss and waste occurring during the life cycle of the various food and beverages of the aforementioned basket; and finally an extensive LCA study of the basket and respective waste and losses. The food categories used for the identification of the basket are meat and seafood, dairy products, crop based products, cereal based products, vegetables, fruit, beverages. The waste and losses of the basket products were estimated using data from the FAO report on such issues (FAO 2011).

MAIN RESULTS

The resulting data on waste and losses, occurring throughout the food supply chains and during food consumption, allow a classification to be drawn concerning the various environmental impacts and at the same time give an indication on the sustainability of food production, management and consumption in the EU. Specifically, the work highlights that the LCA results can change if a different accounting method is used that takes into consideration the lifecycle phases responsible for the generation of the food waste or loss.

IMPLICATIONS

Whilst in traditional food LCA the agricultural phase is the most burdening one, by considering the wastes and losses and their origin, such finding is not necessarily true.

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P61. Protein versus global warming potential – a meta-analysis of the global warming potential and protein ratios for human food

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This paper presents the results of a meta-analysis of LCA studies to identify the Global Warming Potential (GWP)/protein ratio for a range of human foods. Attempting to identify what foods offer the highest amount of protein, for the lowest GWP as 1) identifying alternate protein sources is cited as a key reasons limiting a shift away from a high meat based diet, and 2) Previous comparable studies identify a limited range of food types with respect to the diverse range of possible protein alternative.

The results were generated by expanding on Clune et al.'s (2016) meta-analysis of LCA studies to identify the GWP of 98 raw food types with a protein value higher than 3g/100g. These LCA figures were amended to enable a comparative figure for raw food with minimal packaging at the regional distribution centre. The GWP values were then divided by protein figures for raw food provided by the US dietary website. This created 88,994 GWP/protein scenarios which were statistically analysed.

The results of the paper indicate: 1) findings are generally consistent with the hierarchy identified in other papers that legumes and cereals offer the most satisfactory GWP/Protein ratio followed by non-ruminants, and then ruminants, 2) a very large diversity of results occurs between fish species, 3) Processed vegetarian meat alternatives were comparable with some non-ruminant meats, 4) the variation of results within individual food types in some categories is often high, and could be biased if you decided to cite only a limited number of LCA studies, 5) LCA studies are biased towards particular food types, many of the high protein/low GWP foods types have a low number of LCA studies identified, and would benefit from further attention given their potential role as a protein source in sustainable human diets.

Overall, the study provides a broader range of results for the GWP/protein ratio than previous papers to assist in informing sustainable human diets. The results also illustrate food types where limited LCA studies have been completed.

References

Clune, S., Crossin, E., Verghese, K., 2016. Systematic review of greenhouse gas emissions for different fresh food categories. *J. Cleaner Production* 10.1016/j.jclepro.2016.04.082

Table 1. GWP/protein ratio for select foods g CO₂eq/g protein figures

Food	Sub-food category	Median g CO₂eq/g protein	Stdev	Number of LCA/ protein scenarios utilised
Yellow peas, dry	Legume	0.88	0.03	2
Peas, dry	Legume	1.42	1.09	2
Cowpeas	Legume	2.06	0.54	4
Beans, french & runner	Legume	2.53	2.18	52
Oats	Cereal	2.78	1.17	27
Soybean	Legume	3.22	2.07	12
Peanuts/ground nuts	Legume	3.26	0.80	32
Haddock	Fish	3.41	0.08	4
Rye	Cereal	3.58	0.84	12
Chick peas	Legume	3.99	1.01	3
Lentils	Legume	4.06	0.16	4
Barley	Cereal	4.34	2.43	13
Wheat	Cereal	4.52	1.70	816
Emu	Non-ruminant	6.59	0.09	7
Herring	Fish	6.76	0.98	8
Almonds	Tree nuts	7.23	5.59	12
Cashew nuts	Tree nuts	7.90	2.81	4
Quinoa	Cereal	8.14	0.50	2
Mackerel	Fish	8.92	5.47	84
Tuna	Fish	9.22	6.07	30
Hake	Fish	9.77	3.93	7
Soy grilled pieces (tivall)	Meat alternative (V)	9.77	0.00	1

Pollock	Fish	10.43	3.99	6
Tempeh	Meat alternative (V)	12.94	0.00	1
Whiting	Fish	14.55	8.68	2
Salmon	Fish	16.98	7.03	168
Duck	Poultry	19.34	7.94	10
Chicken	Poultry	19.60	9.67	4,370
Kangaroo	Non-ruminant	19.68	0.73	2
Quorn	Meat alternative (V)	20.00	4.08	4
Tuna	Fish	20.42	5.47	60
Cod	Fish	21.61	8.03	32
Rabbit	Non-ruminant	22.33	4.99	4
Pork	Non-ruminant	28.01	8.40	7,998
Eggs	Poultry	28.03	9.79	37
Turkey	Poultry	33.29	10.49	273
Tofu	Meat alternative (V)	36.81	11.07	42
Rice	Cereal	37.12	19.34	108
Cow milk	Dairy	37.36	16.99	4,192
Cheese (combined avg.)	Dairy	37.58	15.80	1,632
Prawns/shrimp	Shellfish	57.31	90.89	11
Turbot	Fish	90.43	43.06	2
Beef	Ruminant	128.33	62.99	56,265
Lamb	Ruminant	141.80	65.41	4,144
Lobster	Shellfish	152.83	62.45	3

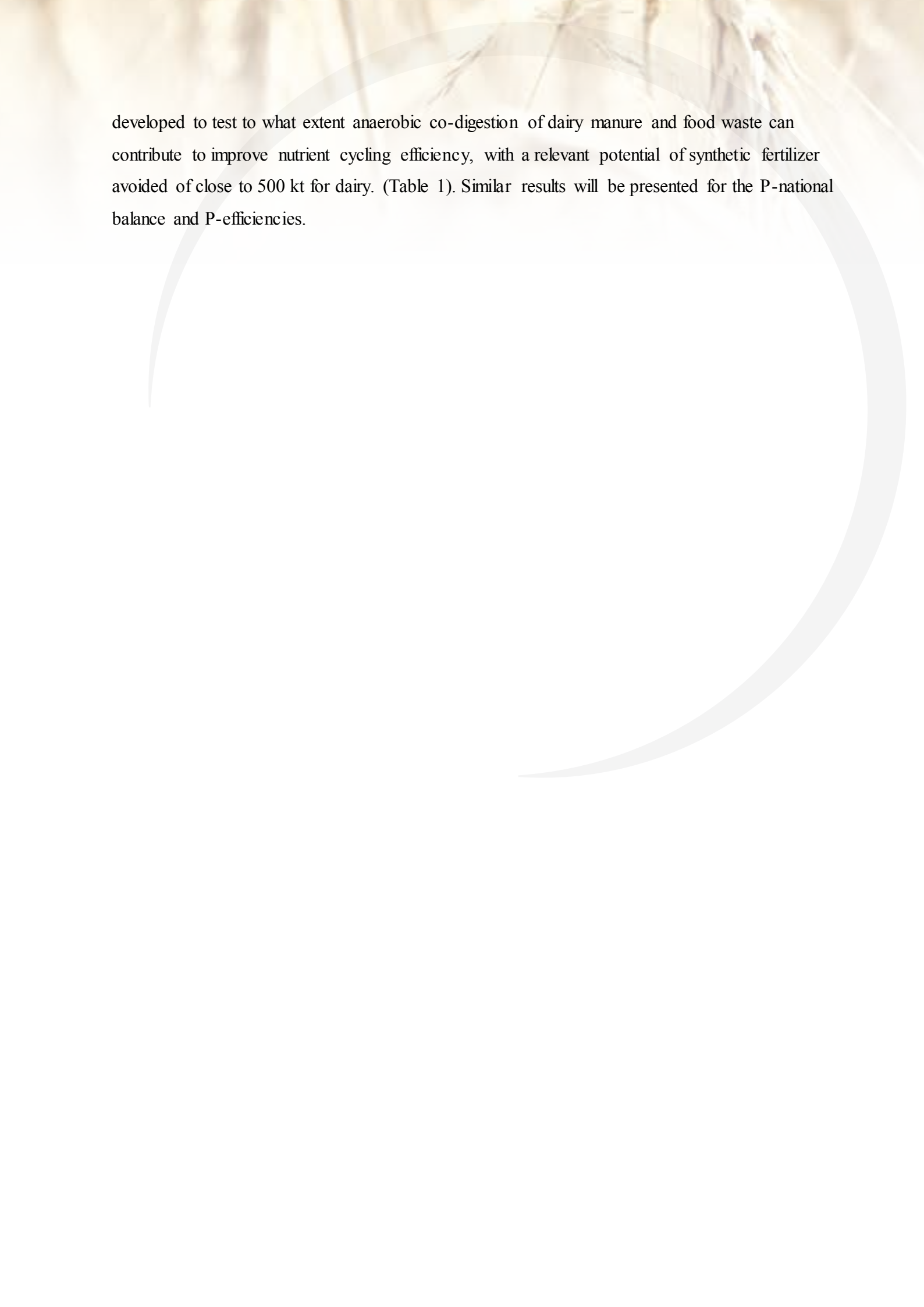
P62. A national balance approach to close the life cycle of nutrient and increase dairy production efficiency

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Analyses of global nitrogen flows have identified large potential for increased efficiency from livestock, and dairy, specifically has an important role in nutrient cycling and delivery of nutrients to consumers. This project aims to create national-level N and P budgets for the US, assessing the role of dairy production in national nutrient flows, quantifying the effects of dairy on nutrient cycling and associated impacts and identifying hotspots and potential improvements in inventory emissions.

Figure 1 shows that fertilizer and chemical industry is the major fixer of N_2 from air, with an annual fixation of 9,350 kt N/ yr. Agriculture is the second largest sector, with an N_2 fixation of 5,156 kt N/yr for US. Agriculture is the largest source of reactive nitrogen emission to air, predominantly due to ammonia emissions of 3,047 kt N from fertilizer application to field crops. Many feedback loops exist in both the dairy food supply chain and the 'non-dairy' food supply chain. For example, 104 kt N is produced as a by-product at the crop-based food industry and fed to the national dairy herd. N-efficiencies range from 71% for the poultry processing industry to 95% for the dairy processing industry. In total 1,203 kt N in synthetic fertilizer is used to produce 445 kt N in milk and 1 kt N in meat (boneless equivalent, veal calves only). Thus, 37% of total N applied with synthetic fertilizer ends up in milk supplied to the processing industry. In terms of nutrient use efficiency (NUE), the dairy processing industry is the most efficient part of the dairy food supply chain with an efficiency of 97%. Retail has then an NUE of 92%. The least efficient part of the dairy supply chain is the dairy herd, with only 21% of nitrogen fed to the herd recovered in milk and meat. The crop production stage and household stage have intermediate nitrogen use efficiencies of 78%. 'Hotspots' in the dairy food supply chain occur predominantly in the crop production stage and the 'dairy herd' (milk production) stage, suggesting that improvement should focus on reducing ammonia emissions in manure management systems and at fertilizer and manure application on the field. An LCA scenario was



developed to test to what extent anaerobic co-digestion of dairy manure and food waste can contribute to improve nutrient cycling efficiency, with a relevant potential of synthetic fertilizer avoided of close to 500 kt for dairy. (Table 1). Similar results will be presented for the P-national balance and P-efficiencies.

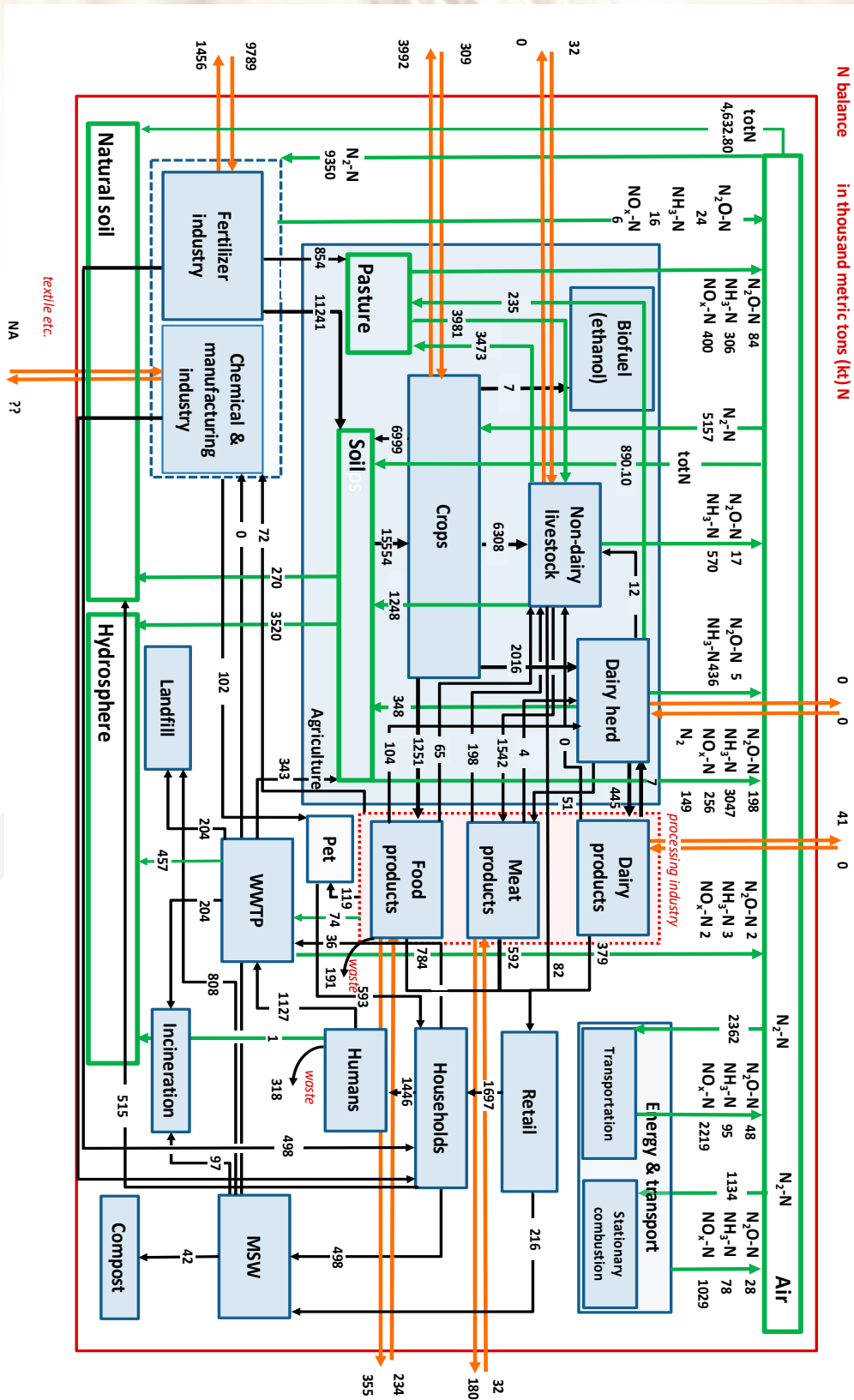


Figure 1. US National nitrogen mass-balance

Table 1 Total nutrient recovered by anaerobic co-digestion and potential fraction of synthetic fertilizer avoided

Nutrient	Total nutrient recovered (kt)	%Synthetic fertilizer avoided (tot. agric. prod.)	%Synthetic fertilizer avoided (dairy livestock)
N	496.9	4.1%	41.3%
P	151.3	8.6%	90%

POSTER PAPERS

CASE STUDIES: PRODUCTS

12. Case studies: Products

P63. From the LCA to the accounting for management control of the aggregated wine production process

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In general, the increase in production, quality and competitiveness undergone by the wine sector during the last years has been unfortunately reached at the expense of reducing the sustainability of traditional production processes. In fact, the amount of energy and materials required for wine production has increased considerably due to a higher consumption of water and plant treatments in the vineyards, the enlarged amount of bottled wine, the limited use of returnable bottles, the widespread use of industrial cooling equipment and the intensification in transport costs as a result of the increase in exports.

However, customers are little by little adding the environmental friendliness as strong criteria for their wine selections in the marketplace and many wineries are dedicating their efforts in a more eco-efficient production so an instrument is necessary to measure and solve problems.

As the higher environmental impacts in the wine production process are consequence of the energy consumption from fossil fuel, carbon footprint can be selected as an appropriate sustainability indicator able to be utilized in the winery decision-making.

The work described in this paper was performed within the framework of the Project “From the vine to the table: carbon footprint labelling of the aggregated wine production process”.

The project aims to characterize the energy related impacts not only of the production process in a Spanish winery but of the entire supply chain. In particular, the sources of information for the estimation of CO₂ emissions in the process of establishment and operation of the vineyard are particularly diffuse and protocols for its measurement were established.

A model for the effective calculation of these aggregated emissions, its accrual and percentage allocation for quantities of service and/or provided product in each case has been developed. The developed application will be integrated into the company management system, allowing the emissions accounting and its automatic association to systems such as the financial accounting, the product sheet, the supplier sheet, etc.

In this way the company could have a constantly updated record of the approximate volume of CO₂ emissions related to purchases or production units as well as timely information on the environmental performance of their suppliers/producers in terms of emissions and energy efficiency.

In line with the corporate commitment to reduce their environmental impact, the analysis of this data will allow the winery to detect potential improvements in the whole process of production of wine whose implementation can provide a considerable increase in the margin for manoeuvre.

And additional feature of the application is the added calculation of data on CO₂ emissions and energy efficiency which result would be ready to be included in the corporate information and communication system, as well as the allocation of the volume of emissions for each unit of product and its inclusion on the label of the product (wine bottles/box) showing the environmental commitment of the company, distinguishing positive corporate image, and facilitating access to international markets "green".

On the other hand, benefits directly associated to the implementation of initiatives for the measurement, accounting and labelling of emissions throughout the entire chain of value of the wine sector, are elevated and distributed throughout the entire chain. Knowing the levels of emissions generated in the primary activity in the fields, from the different agricultural processes involved, allows incorporating measures for improvement that means lower costs in energy consumption with consequent savings and increase in industrial competitiveness.

P64. A methodology for sustainability evaluation of food supply chains: Example of organic potato in Northern France

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This study aimed at developing and testing a method to assess the economic and environmental impacts of different local food supply chains. This method should easily test the sensitivity of economic and environmental indicators to new technical choices. It was tested by evaluating the supply chain of organic potato in Picardy (Northern France).

The environmental assessment was performed by the Life Cycle Assessment. The system boundaries were from cradle to selling location's gate (including the distribution phase). Environmental impact was calculated with Recipe method for climate change, acidification and eutrophication, with cumulative energy demand method for energy demand and with UseTox method for ecotoxicity. Economic assessment was based on the calculation of financial accounts for each economic activity along the supply chain (agricultural production, storing, packaging ...). Financial accounts represent resources and workforce required for the production throughout the entire business life cycle and allowed to calculate the following indicators by activity: investments, profitability (Net Present Value), return on investment, employment, value added and its distribution. The latter determine the distribution of wealth created amongst national economic actors (figure 1): workers (wages), financial institutions (interest charges), local and national administration (taxes, social charges) and non-financial enterprises (Gross operating surpluses). A sector study was conducted to identify all the economic activities encompassed by the value chain and to quantify the physical and monetary flows between each step.

Table 1 gives an example of comparison between mechanical and thermal haulm crushing. Little differences appear between both practices on environmental and economic indicators. For organic potatoes supply chain, more than 20 different scenarios (with different practices on crop production and distribution system) were assessed. The tests show the links between economic and environmental indicators for each activity of the supply chain. This method is reproducible for other food supply chains and can be easily used to assist the decision making process and the system optimization by giving a wide overview of different scenarios quickly.

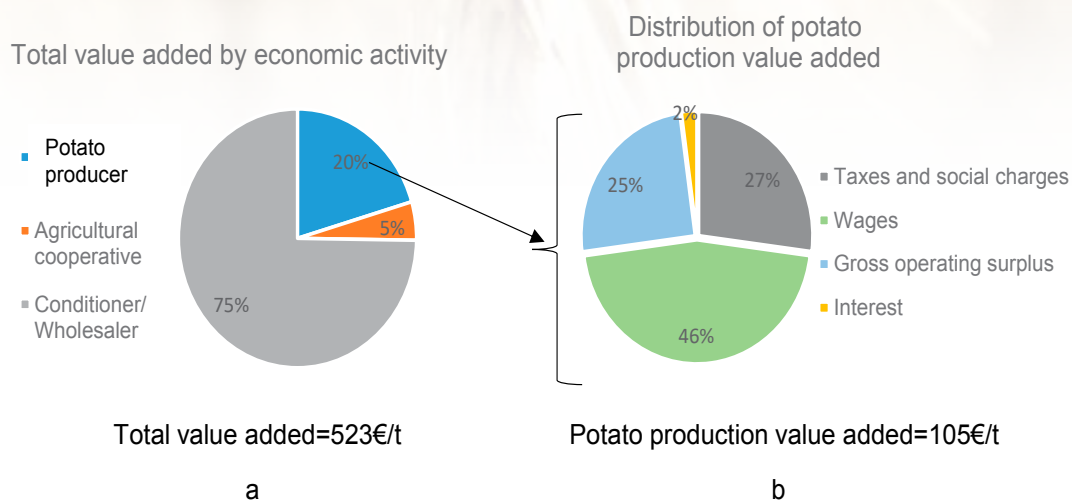


Figure 1 : Value added distribution:

- Among each economic stakeholder along the supply chain (left)
- Among potato production (agricultural activity only) (right)

Table 1 : Variation on environmental and economic indicators for 1 ton of potato at farm gate: example of the substitution of mechanical haulm crushing by thermal haulm crushing.

Dimension	Indicators	Variation (thermal haulm crushing/mechanical haulm crushing)
Environmental	Climate change	+4%
	Eutrophication	0%
	Acidification	+2%
	Ecotoxicity	0%
	Energy demand	+7%
Economical	Value added	-3%
	Gross operating surplus	-6%
	Net present value (on 10 years)	-11%
	Time of return on investment	+30%
	Employment	0%

P65. Environmental and Eco-efficiency Assessment of Artisan Cheese Production with Protected Designation of Origin in the Mediterranean Island of Minorca

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I. Objective

Islands are hotspots in the provision of local, unique products and the integration of the environmental dimension is especially interesting to ensure their self-sufficiency and resource protection. The goal of this study was to determine the life cycle environmental impacts of the artisan cheese with Mahón-Menorca Protected Designation of Origin (PDO) through an eco-efficiency analysis of four local companies (E1 to E4).

II. Method

A Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) of the cheese production (secondary sector) was performed using data from the companies. Background data were retrieved from ecoinvent v3 through Simapro 8; we used the ReCiPe (H) method to estimate the environmental impacts. The functional unit was the production and sale of 1 kg of cheese with PDO. We integrated the LCA and LCC results to assess the eco-efficiency of the secondary sector based on ISO 14045:2012 (Global Warming Potential depicted in Figure 2). The artisan sector was compared with an industrial producer that operates in the same area. We applied literature data to calculate the environmental impacts of the primary and tertiary sectors. In this case, only the GWP (100 years) was included due to data availability.

III. Main result

The total estimated GWP of the traditional cheese industry was 11.7 kg CO₂eq/kg (Table 1), 91% of which was associated with the primary sector due to the feeding needs of the cattle and manure production. When comparing different types of companies (Figure 2), E3 was the most eco-efficient, as it used a natural drying chamber. In contrast, E4 had the highest environmental and economic costs because it applied more energy-intensive processes. Although cheese is exported to countries such as the United States, sales are not the major challenge (Table 1), and, due to the relevance of the primary sector, artisan and industrial production had similar results.

IV. Implications

This study highlights that the impacts of the agri-food industry located in islands are not always associated with the transportation to continental areas. In this case, there is a need for improving the processes related to milk production in order to reduce the footprint of cheese and preserve the resources of the island.

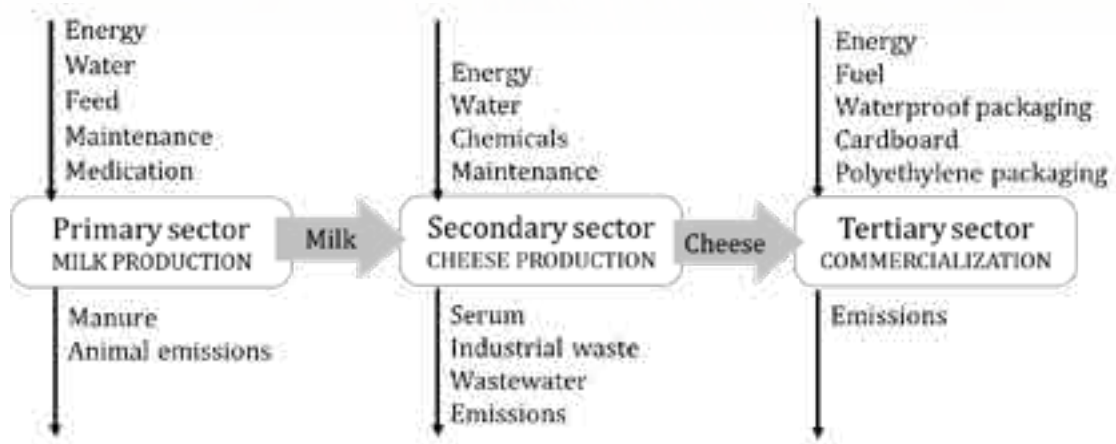


Figure 1. Life-cycle stages of the traditional cheese sector in Minorca

Table 1. Global Warming Potential of the artisan and industrial cheese sector in Minorca and main contributors

Sector	Artisan production		Industrial production ^a	
	kg CO ₂ e./kg cheese	%	kg CO ₂ e./kg cheese	%
Primary	10.6	91	10.6	90%
Secondary	1.0	8.6	0.4	4%
Tertiary	0.05	0.4	0.7	6%
TOTAL	11.7	100	11.7	100

^a Data on the secondary sector from Llopis (2010), Evaluation of alternatives for reducing the environmental impacts of the large-scale production of D.O Mahón-Menorca cheese through Life Cycle Assessment (Evaluación de alternativas para la disminución del impacto ambiental del proceso de elaboración de queso D.O. Mahón-Menorca a gran escala mediante Análisis de Ciclo de Vida). Universidad Politécnica de Valencia

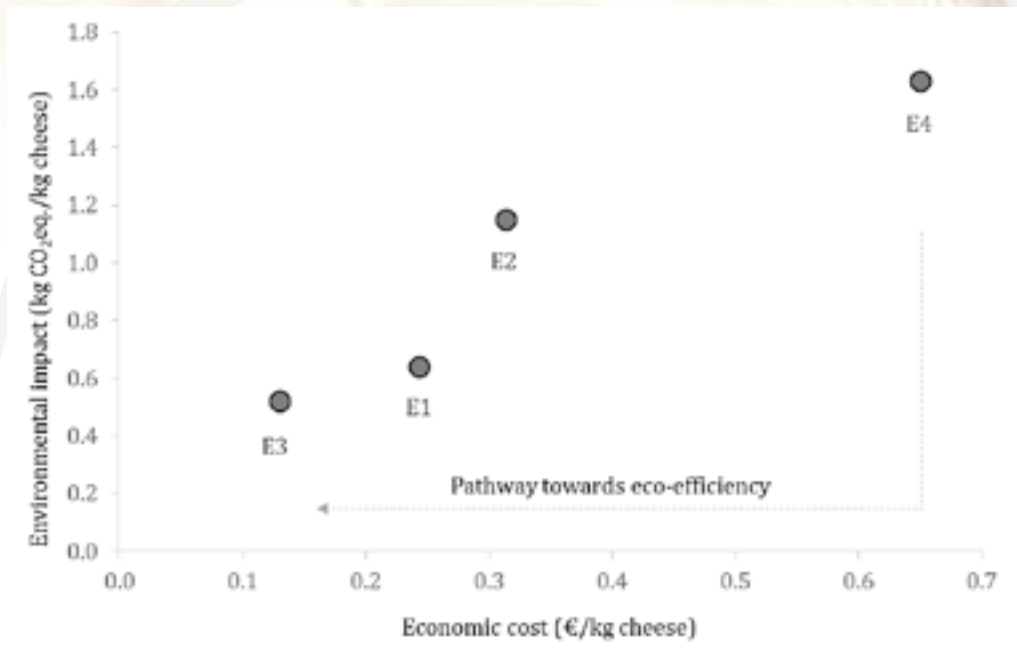


Figure 2. Eco-efficiency benchmarking of the secondary sector considering the companies under analysis

P66. Environmental impacts of duck *foie gras* production in 3 contrasting systems

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5ITAVI, 4CTCPA, 2ASSELDOR, 3INRA, 1INRA

The *foie gras* sector has established two processes to certify production conditions for consumers: Red Label (RL) and Protected Geographical Indication (PGI) which differ of standard (STD) system by a longer rearing and overfeeding periods and specification on geographic origin of resources used (only for PGI). *Foie gras* is a luxury product whose image should not be tarnished by high environmental impacts. The aim of this work was to compare with LCA method, the environmental impacts of duck *foie gras* production in these 3 systems at the slaughter house gate.

Attributional LCA were based on average production systems (experimental and bibliographic data or survey for primary data, see Table 1; INRA and Ecoinvent database v2.2 for secondary data, SimaPro 8.1.0.60 software). Seven impact categories were calculated using mainly CML2 baseline v2.04 method for 1t of *foie gras* and using an economic allocation approach: eutrophication potential (EP, kg PO₄³⁻ eq.), global warming potential (GWP, kg CO₂ eq.), acidification potential (AP, kg SO₂ eq.), terrestrial ecotoxicity (TE, kg 1,4- DCB eq.), primary energy use (PEU, MJ; CED v1.05 method), Water Use (WU, m³) and land occupation (LO, m² .year).

The potential environmental impacts of *foie gras* production are rather similar for STD and PGI systems (<5% of difference for all impacts) but higher for RL system (+9 to +24% depending on impacts, Table 1), except for AP and CED (difference with STD or PGI system <5%). This is due to longer production process and higher feed intake. Regarding the steps of the production process, the major contribution to the potential impacts are rearing (36 to 63%) and/or overfeeding (28 to 57%) periods. Considering the class of inputs, the major contribution is feed (>56%) except for AP, mainly explained by manure management (>75%).

Present results suggest that the RL system offers to consumers a product with higher sensory quality but this is linked to slightly higher environmental impacts. On the opposite, the PGI system guarantees to consumers the respect of specifications concerning production process without increase in environmental impacts. This work shows that the management of feed and manure during the rearing and overfeeding periods are the most relevant ways to reduce environmental impacts of duck *foie gras* production.

Table 1. Main characteristics and performances in the 3 production systems

	Production system		
	STD	PGI	LR
Characteristics and performance			
Feed intake during rearing period (kg)	15.5	16.0	17.3
Corn intake during overfeeding period (kg)	8.7	9.0	9.5
Mortality during rearing period (%)	2.1	2.5	3.0
Mortality during overfeeding period (%)	3.2	2.5	1.5
Age at slaughter (days)	89	93.5	105
Live weight at slaughter (g)	5.5	5.7	5.9
Weight of <i>foie gras</i> (g)	538	546	548
Price of <i>foie gras</i> at slaughter house gate (€/kg)	12.9	14.5	16.2
Environmental impacts for 1t of <i>foie gras</i>			
Global warming potential (kg CO ₂ -eq.)	823	838	795
Eutrophication potential (kg PO ₄ ³⁻ -eq.)	370	374	404
Acidification potential (kg SO ₂ -eq.)	41 532	40 644	46 698
Terrestrial toxicity (kg 1,4-DB-eq.)	3 633	3 655	4 369
Primary energy use (MJ)	66 127	66 561	78 509
Water use (m ³)	504 387	479 609	520 500
Land occupation (m ² .year)	3 394	3 412	4 202

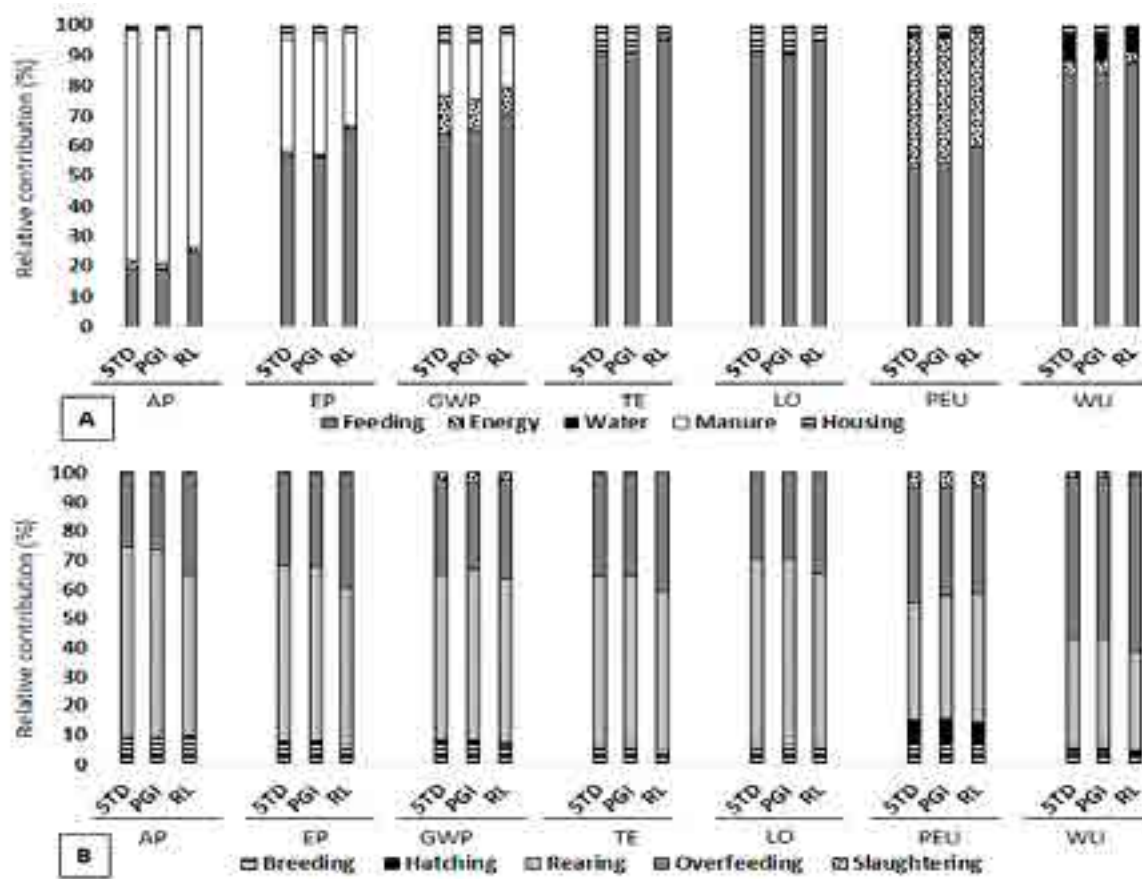


Figure 1: Contribution of different classes of inputs (A) and of different steps of the production process (B) to the potential environmental impacts of duck *foie gras* production in 3 systems.

P67. Carbon footprint of three organic beverage products of an Italian Company

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- Objective of the work

The objective of this work was to evaluate the greenhouse gas emissions of three different organic beverage products made of oat, rice and almond, provided by a north-east Italian company, highlighting what are the life cycle stages responsible for the most part of the final emissions of each product.

- Methodological detail

The carbon footprint assessment was performed according to the GHG Protocol Life Cycle Product Accounting and Reporting Standard (Product Standard), adopting a life cycle approach from cradle to grave. The function unit was 1000 ml of packaged beverage in each case. The impact assessment method adopted was the IPCC 2013 GWP 100a.

- Main Results of the study

According to the values listed in absolute terms in table 1 and in percentage terms in figure 1, results showed minimal variations among the three products for the life cycle stage of packaging, production, use and disposal. On the contrary, significant variations were observed for the raw material stage (0,220 kg CO₂eq for oat drink, 0,809 kg CO₂eq for rice drink, 0,177 kg CO₂eq for almond drink) and distribution stage (0,123 kg CO₂eq for oat drink, 0,183 kg CO₂eq for rice drink, 0,199 kg CO₂eq for almond drink). While distribution stage was mainly affected by location of customers to which products are delivered, with different distances and different kind of transports, variation of results among the three products for the raw material stage were due to different impact of the main agricultural ingredient used to realize each drink, corresponding to 0,205 kg CO₂eq for oat, 0,782 kg CO₂eq for rice, 0,163 kg CO₂eq for almond.

- Implications, meanings, conclusions

This study highlights that raw material and distribution together were the life cycle stages responsible for more than 50% of the total carbon footprint of each product analysed. The main ingredient of each product highly affected the final impact, mainly because of the high level of emissions generated by

farming operations, underlining the importance to have high quality data especially for raw material life cycle stage.

Life cycle stage	Unit	Oat drink	Rice drink	Almond drink
Raw materials	kg CO ₂ eq / F.U.	0,220	0,809	0,177
Packaging	kg CO ₂ eq / F.U.	0,106	0,106	0,117
Production	kg CO ₂ eq / F.U.	0,114	0,115	0,112
Distribution	kg CO ₂ eq / F.U.	0,123	0,183	0,199
Use	kg CO ₂ eq / F.U.	0,025	0,025	0,025
Disposal	kg CO ₂ eq / F.U.	0,012	0,012	0,012
Total	kg CO ₂ eq / F.U.	0,599	1,250	0,642

Table 1: Carbon Footprint results according to the different life cycle stage in terms of kg CO₂eq over the functional unit of each product.

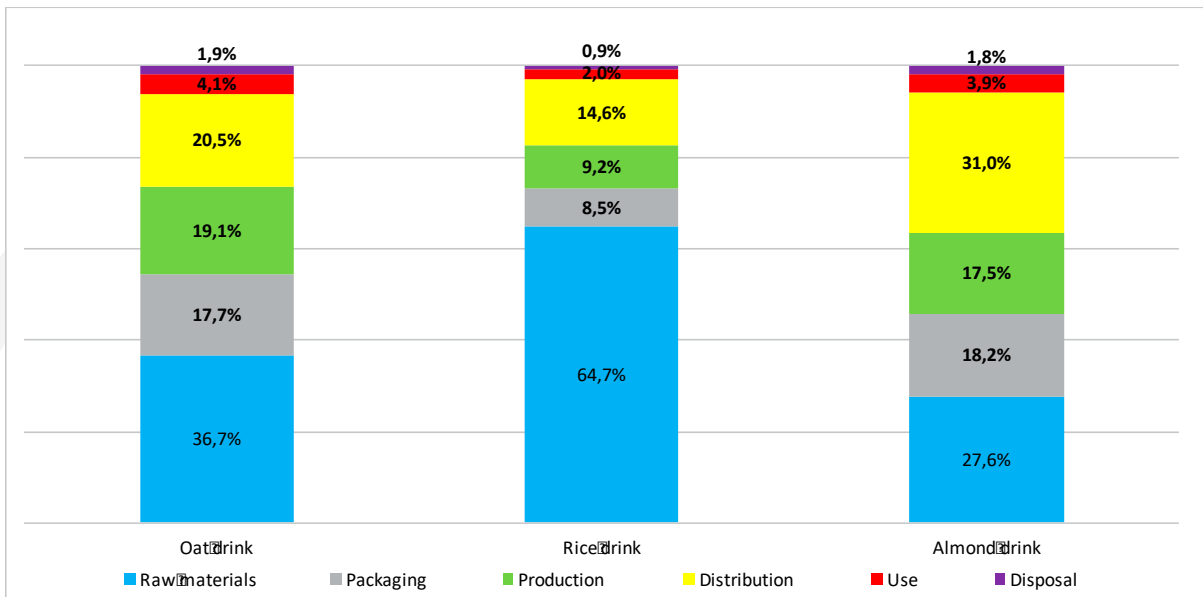


Figure 1: Carbon Footprint results according to the different life cycle stage in terms of percentage incidence on the total value of each product.

P68. Improved fertiliser management reduces the product carbon footprint of green coffee: a case study from Vietnam

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Product carbon footprints (PCFs) are a tool for estimating and reducing greenhouse gas emissions related to products and supply chains. Here we report the results of a case study on the PCF of green coffee production in Vietnam which was part of a Public-Private-Partnership (PPP) project on smallholder farms. The aim was to quantify the climate mitigation potential of changing the fertilisation management from common farmer practice to a more balanced plant nutrition program.

Two fertilisation strategies were compared using paired plots on 21 farms: 1) the baseline was characterised by common over-fertilisation, the use of soil acidifying fertilisers and a lack of micro-nutrient applications; 2) the more balanced nitrate based PPP crop nutrition program reduced nutrient application rates and included micro-nutrients.

The PCF analysis included all relevant processes up to the farm gate and the transport of the green coffee beans to the next supply chain partner. In addition, the impact of changing the fertilisation program on yields and farmer incomes was analysed.

The results show that changing the fertiliser management decreased the PCF of green coffee within the case study sample of farms by an average of 16% (Figure 1). Average yields and net farmer incomes increased by 11% and 14%, respectively.

Thus, improved fertilisation achieved important co-benefits and represents a significant opportunity for addressing the challenge of reducing greenhouse gas emissions while at the same time increasing yields and improving farmer incomes.

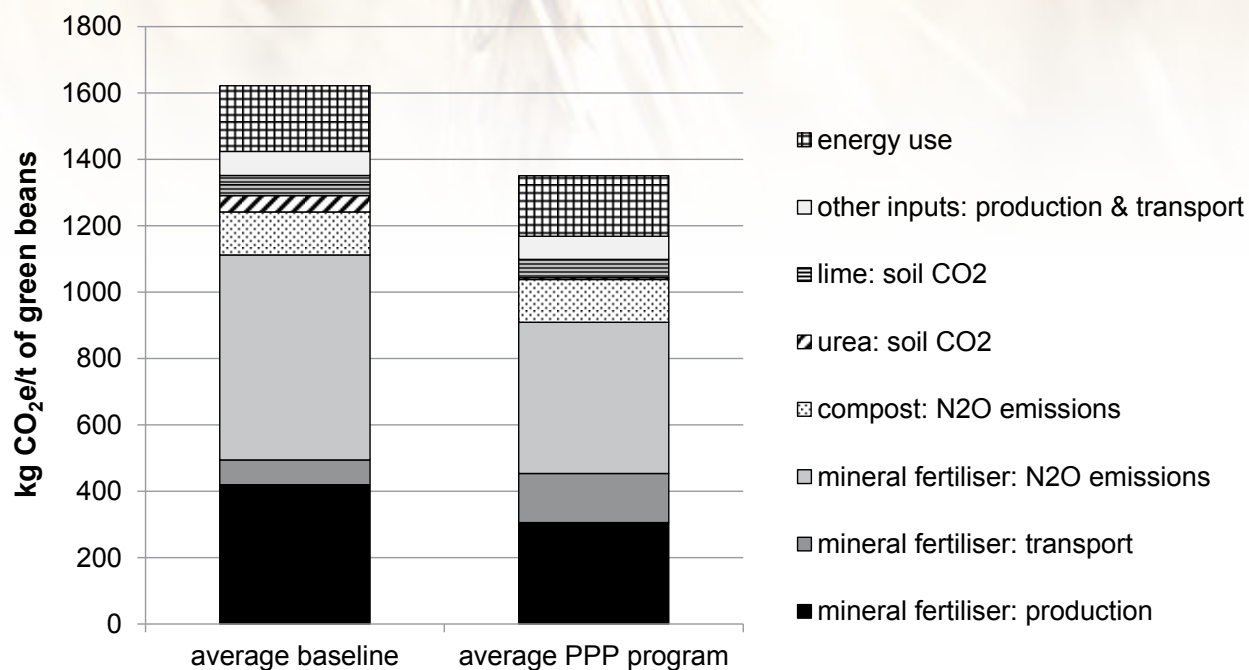


Figure 1: Average product carbon footprint (kg CO₂e/t of green beans) for two contrasted fertilisation programs (“baseline” and “PPP program”), based on a sample of paired plots on 21 farms in Vietnam and including all relevant processes up to the transport of the green beans to the next supply chain partner.

P69. Comparing the environmental impact of home grown tomatoes with supermarket products

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Goal and Scope

This study investigates the environmental impact of three different scenarios for the supply of tomatoes to households: (a) home grown tomatoes from own plantation in a pot, (b) field grown tomatoes from supermarket and (c) tomatoes from supermarket, grown in a greenhouse. Data include the entire life cycle from field to household. Food losses are taken into account.

Life Cycle Inventory

The life cycle inventory for home grown tomatoes includes: clay pot (20 l, 10 year lifetime), garden mould (20 l/a and pot), purchase of seedlings, fertilizer usage according to packaging instructions, pesticide usage corresponding to the average use in private gardens, tap water (40 l/season and plant), yield (3 kg/pot and season). An average Swiss transport scenario for consumer purchases is used for transports to home. The inventory analysis is based on data in the ecoinvent database v2.2 [1] and in the ESU-database [2].

Life Cycle Impact Assessment

The environmental impact is assessed with the Ecological Scarcity Method 2013 [3] and summarized to ecological scarcity points. **Error! Reference source not found.** shows the environmental impact per cultivation method and kg of tomatoes. The overall environmental impact of the different cultivation methods is subdivided by the source of the impact. Field grown tomatoes from the supermarket show a lower impact than home grown tomatoes. The usage of garden mould and its transport to home cause the most relevant environmental impact of home grown tomatoes. The environmental impact of tomatoes grown in a greenhouse is mainly caused by heating.

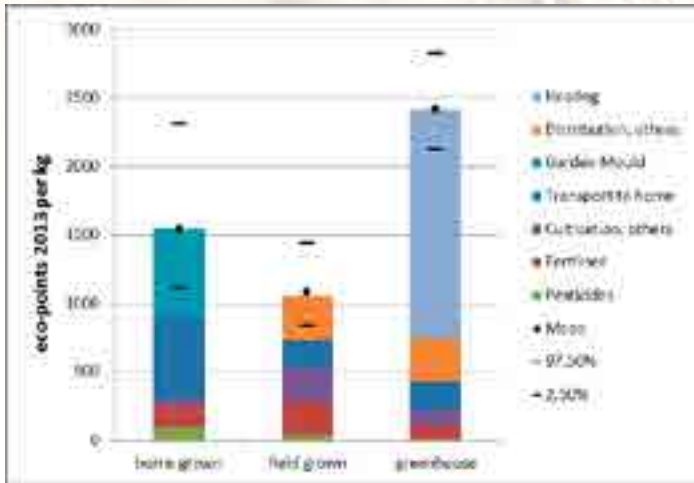


Fig. 1: Environmental impact of different tomato cultivation methods per kg of tomatoes at home (eco-points 2013 per kg) with confidence interval of 5 percent.

The impact categories “global warming”, “main air pollutants and particulate matter” and “water pollutants” show the highest variability regarding their influence on the overall impact (see **Error! Reference source not found.**). The share of the other impact categories does not relevantly vary between the different cultivation methods. The share of main air pollutants and particulate matter is highest for home grown tomatoes. This is caused by the high amount of garden mould per kg of yield in comparison to the other cultivation methods. The relatively high amount of garden mould has also an effect on the environmental impact caused by home transport. The share of water pollutants in the total environmental impact is highest for field grown tomatoes. This is because the highest amount of nitrogen fertiliser is calculated for that cultivation method. The relatively high environmental impact of tomatoes grown in a greenhouse compared to other cultivation methods is due to heating. Other environmentally related impacts tend to be lower than in the open-ground cultivation.

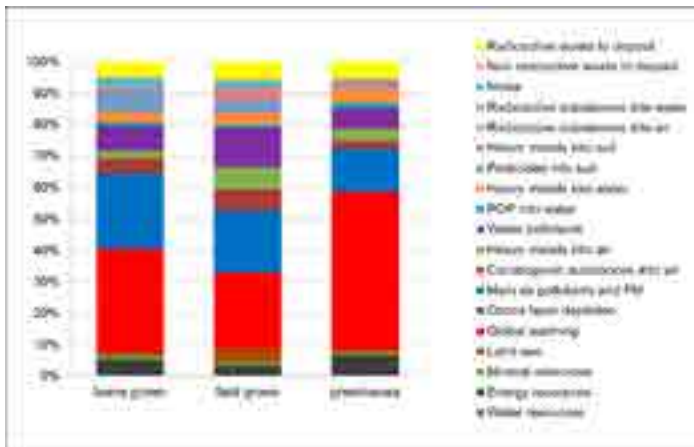


Fig. 2: Percentage of impact category in overall environmental impact per cultivation method (Ecological Scarcity Method 2013).

Interpretation

According to the assumptions in this study home grown tomatoes do not cause lower environmental impacts than seasonal tomatoes from the supermarket. It has to be considered

that the performance of this cultivation method is very much depending on the individual cultivation behaviour. The yearly usage of garden mould can be reduced by the reuse of material. This is also related to a reduction of transport weight. In addition, transport by bicycle or by other means of transport (train, bus) could reduce the impact. Fertilizer and pesticides usage can also be reduced. However, this measure may lead to lower yield results. Considering the impact reduction potential it seems possible to cultivate home grown tomatoes which cause a lower environmental impact than the other cultivation methods. But, on the other side it can be feared that many home gardeners do not perform as well as professional tomato growers.

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P70. Environmental profile of Brazilian sugarcane

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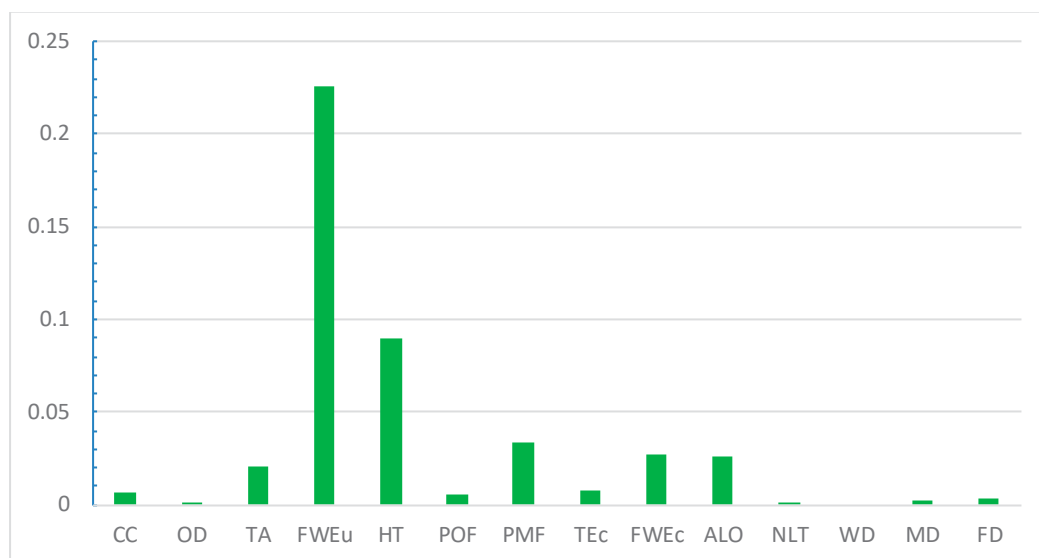
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The sugarcane ethanol is a very important fuel used for vehicles in Brazil. About 32 million vehicles of the country's fleet has a flexible-fuel technology, and are supplied by sugarcane ethanol, petrol or a mixture of both. Sugarcane is also used for the production of sugar and the industrial waste from both processes is used on electricity generation, representing 15.7% of the national energy matrix. In the 2013/2014 season, 8.81 million ha were cultivated, producing 659 million tons of sugarcane. Therefore, it is possible to figure out the potential environmental impact of the Brazilian sugarcane cultivation. This study characterized the sugarcane cropping systems of nine producing regions in the country considering specific technical parameters of the agricultural process and specific input parameters of the estimative models of emission. A major effort was undertaken to determine the land use changes related to sugarcane cultivation and their emissions. An environmental profile was generated as a result of the Brazilian sugarcane modal cropping system, identifying its hot spots. The agricultural production phase is the main contributor to eight of the impact categories analyzed. The most significant categories were Freshwater Eutrophication and Human Toxicity, the first being caused by phosphorus emission into surface water due to erosion, and the second caused by heavy metals emitted into soil, substances arising from fertilizers. Particulate Matter Formation was mainly caused by the burning of straw, practiced when the harvest of sugarcane is manual. This practice also contributed to Terrestrial Acidification, due to NO_x emissions, which along with NH₃ emissions, derived from the application of nitrogen fertilizer, accounting for 94.2% of this impact. For Freshwater toxicity, substances that cause impact were copper (35.8%), present in fertilizers, and fipronil (25.3%), a pesticide. The energy balance of the sugarcane life cycle - ratio of renewable

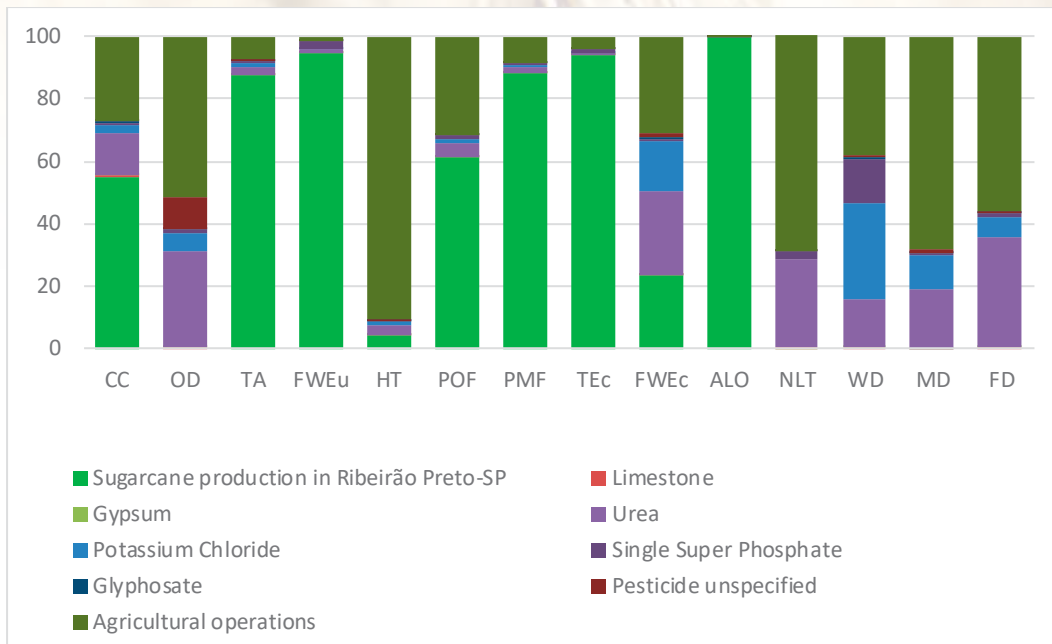
energy produced and fossil energy consumed - was highly favorable, 26.4, precisely because of the biomass production by this crop. Cumulative emissions of greenhouse gases were equivalent to 54.13 kg CO₂ eq per ton of sugarcane. The main emissions were NO₂ (32.2%) and CO₂, the latter derived from the land use change (34.2%) and from the application of fertilizers (27.2%). The improvement in the environmental performance of this crop should focus on to avoid clearing of new areas for agricultural use, avoid planting in high slope areas (where the harvest can not be mechanized) and adopt the rational use of fertilizers.

Figure 1. Normalized environmental impacts of Brazilian South-Central sugarcane production.



Impact categories: CC, Climate Change; OD, Ozone Depletion; TA, Terrestrial Acidification; FWEu, Freshwater Eutrophication; HT, Human Toxicity; POF, Photochemical Oxidant Formation; PMF, Particulate Matter Formation; Tec, Terrestrial Ecotoxicity; FWEc, Freshwater Ecotoxicity; ALO, Agricultural Land Occupation; NLT, Natural Land Transformation; WD, Water Depletion; MD, Metal Depletion; FD: Fossil Depletion.

Figure 2. Environmental profile of the cropping system of the largest producing region in Brazil (Ribeirão Preto).



Impact categories: CC, Climate Change; OD, Ozone Depletion; TA, Terrestrial Acidification; FWEu, Freshwater Eutrophication; HT, Human Toxicity; POF, Photochemical Oxidant Formation; PMF, Particulate Matter Formation; Tec, Terrestrial Ecotoxicity; FWEc, Freshwater Ecotoxicity; ALO, Agricultural Land Occupation; NLT, Natural Land Transformation; WD, Water Depletion; MD, Metal Depletion; FD: Fossil Depletion.

P71. LCA of rapeseed oil in Portugal: endogenous vs. imported feedstock

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Rapeseed is an important feedstock for feed and energy purposes. The main goal of this article is to present an environmental life-cycle assessment of rapeseed oil produced in Portugal comparing alternative pathways of endogenous and imported rapeseed. A life-cycle inventory and model was implemented addressing land-use change (LUC), cultivation, oil extraction and transport. The functional unit adopted was 1 kg of rapeseed oil. LUC was assessed for two alternative scenarios. A comprehensive inventory of different rapeseed cultivation systems was implemented based on endogenous data collected from eight farms in Portugal together with data (from the literature) for imported rapeseed from five countries. Rapeseed oil extraction was modeled based on data collected in two Portuguese mills. The extraction is a multifunctional process, producing both rapeseed oil (0.42 kg kg⁻¹ seed) and rapeseed meal (0.58 kg kg⁻¹ seed). Emission factors for chemicals, field operations, energy and transport were adopted from databases and literature. Fertilization emissions (nitrous oxide, ammonia, nitrate, nitrogen oxides, phosphate and phosphorous) were calculated based on the IPCC Tier 1 approach (2006) and SALCA-P models. Impacts (ReCiPe method) were assessed for global warming (GHG intensity), terrestrial acidification, ozone depletion, photochemical oxidation, marine and freshwater eutrophication. The results show the importance of LUC and cultivation on rapeseed oil life-cycle impacts, but significant differences were observed for the alternative LUC scenarios and cultivation systems assessed and there is a high-level of uncertainty due to field emissions from fertilizer application. Despite the high uncertainty, results point to the importance of promoting endogenous rapeseed to displace imports to produce oil with lower environmental impacts. A sensitivity analysis to alternative allocation procedures (price, mass and energy allocation) was also performed, showing that considerably lower impacts are calculated for mass allocation, whereas similar results were obtained for energy and price allocation.

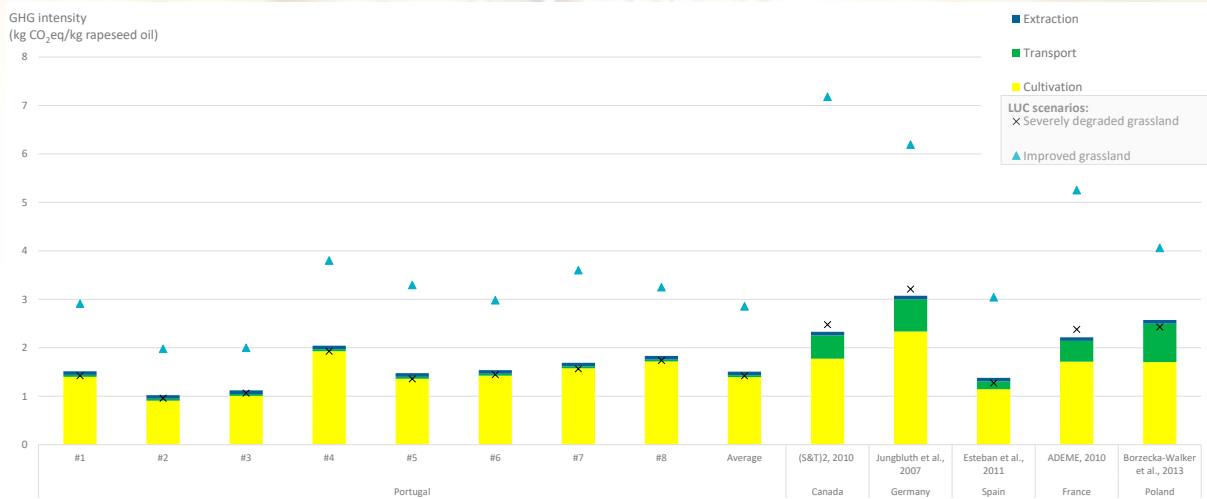


Figure 1. GHG intensity of rapeseed oil (price allocation) for two LUC scenarios using endogenous (eight farms in Portugal) or imported rapeseed.

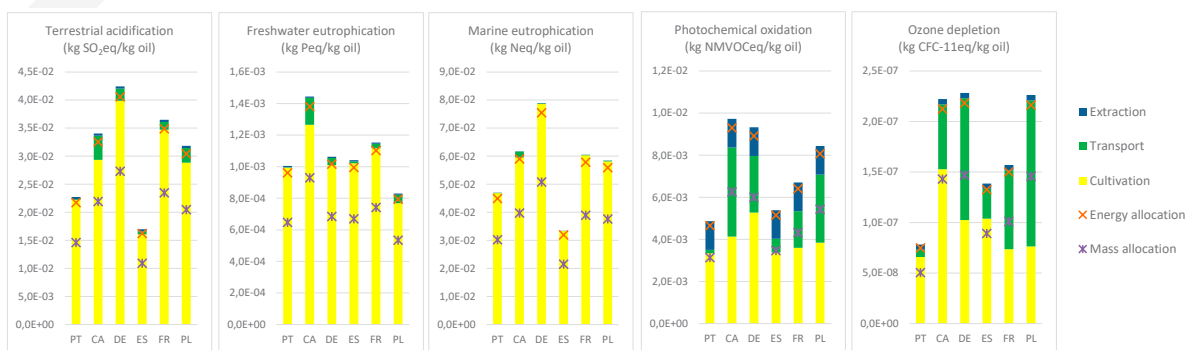


Figure 2. LC environmental impacts of rapeseed oil using endogenous or imported rapeseed (price, mass and energy allocation).

P72. Scenario-based Construction of Life Cycle Inventories for Agricultural Production Systems: Case Studies from the Asian Rice LCA Initiative

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An important purpose of applying life cycle assessment (LCA) to agriculture is to establish sustainable agricultural production systems under site-specific conditions; therefore, scenario-based life cycle inventory (LCI) data is crucial. However, construction of scenario-based data differs from that of national average data used sometimes in environmental labelling policies. In the present study, we examined the possibility of constructing scenario-based LCI data for rice production systems in Asian countries.

As a preliminary step, we determined whether the construction of LCI data for rice production in Japan and Korea, both of which have extensive statistical publications on rice production and LCI data for general industrial background processes, was possible by using government statistics. Then, we assessed the possibility of constructing LCI data on the basis of the detailed scenarios of rice production techniques in each country.

We confirmed that life cycle greenhouse gas (GHG) emissions from rice production systems in Japan and Korea could be calculated using government statistics (Table 1). However, the reason for GHG emission differences was difficult to specify. In contrast, we recognised that scenario construction of foreground processes based on detailed production scenarios (Table 2) was feasible in each country, although the development of inventories for background processes is necessary in the future.

The results indicate that in making production systems comparable with each other, we should use explicit production scenarios. Therefore, if we use averaged statistical information, the technical details on agricultural practices disappear and comparisons between agricultural production systems become difficult. Further development of LCI data on agricultural production and related processes,

including agricultural inputs, should be considered to clarify the potential of comparative LCA using explicit scenario generation.

Table 1. Life cycle GHG emissions from rice production systems in Japan and Korea and data sources used for the calculation.

	Japan	Korea
Life cycle GHG emissions (kg CO ₂ eq./kg)	1.48	1.14
Direct emissions from paddy fields (kg CO ₂ eq./kg)	0.69	0.86
Crop yield (kg/ha)	5,830 (MAFF, Japan)	4,830 (RDA, Korea)
Data for the amount of agricultural inputs	Statistical Survey on Farm Management and Economy	Food, Agriculture, Forestry and Fisheries Statistical Yearbook
Data and methods used for calculating direct GHG emissions	National Greenhouse Gas Inventory Report of Japan	Guidelines for Local Government Greenhouse Gas Inventories

Table 2. Information gathered for constructing rice cultivation scenarios in each country

No.	Item
1.01	Crop name
1.02	Variety (cultivar)
2.01	Location
2.02	Soil type
2.03	Average yield
2.04	Transplanting or direct seeding
2.05	Maximum depth of water
2.06	Proceeding crop (if rotated)
2.07	Following crop (if rotated)
3.01	Plowing and land preparation
3.02	Border coating
3.03	Raising of seedlings
3.04	Basal fertilizer application
3.05	Application of other organic materials

- 3.06 Start of flooding
 - 3.07 Soil puddling
 - 3.08 Transplanting
 - 3.09 Weeding
 - 3.10 Midseason drainage
 - 3.11 Water management after midseason drainage (intermittent irrigation)
 - 3.12 Additional fertilizer application
 - 3.13 Fungicide application
 - 3.14 Insecticide application
 - 3.15 Herbicide application
 - 3.16 Herbicide application (ridges between rice fields)
 - 3.17 Weeding in ridges between rice fields
 - 3.18 Harvesting
 - 3.19 Drying
 - 3.20 Management of rice straw
-

P73. Comparing the environmental performances of craft and industrial beer: strengths and weaknesses of Life Cycle Assessment

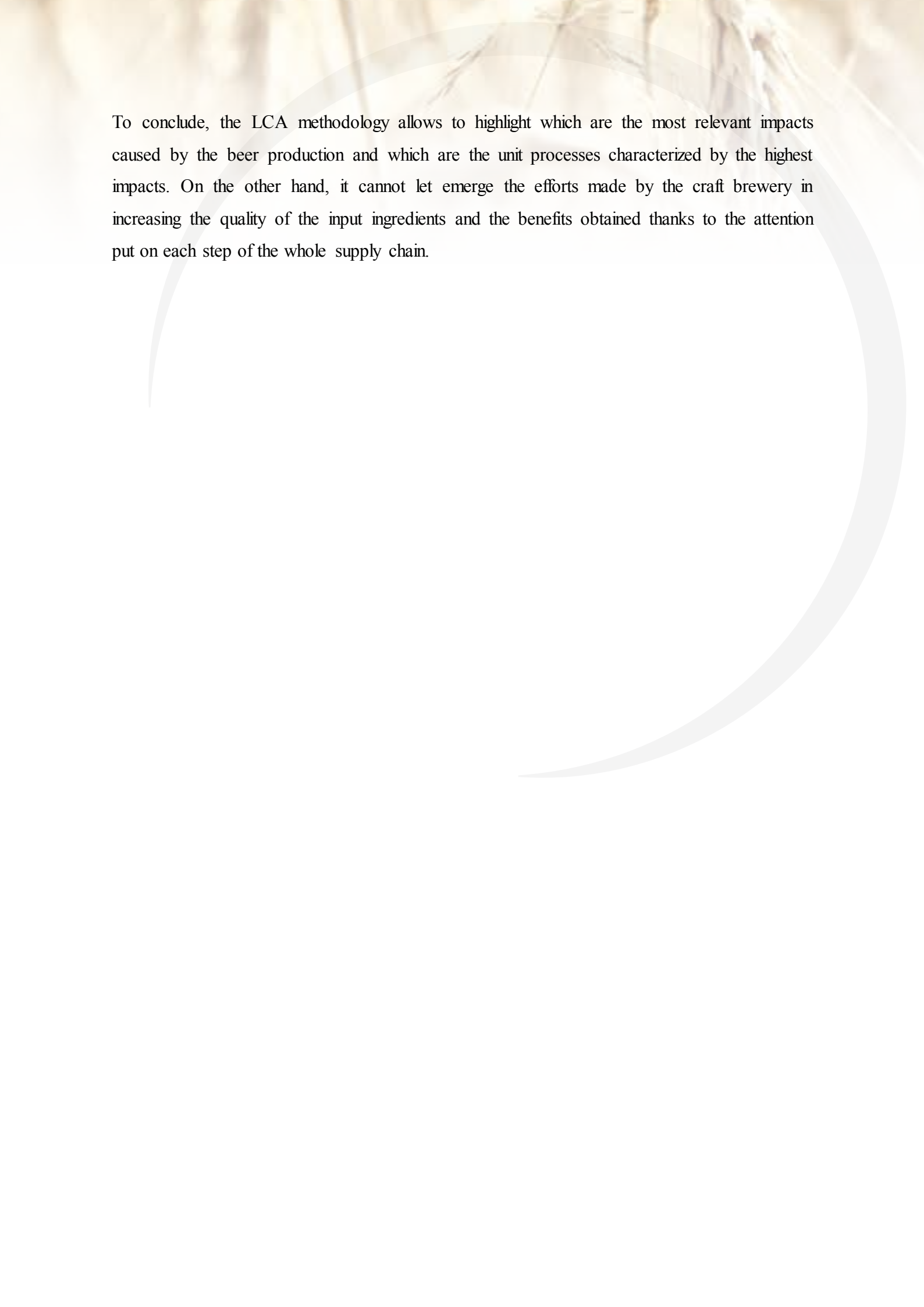
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The European brewing industry put the environmental performance of its product in the foreground and consequently beer was selected among the Pilot Products for the Product Environmental Footprint (PEF) by the European Commission. Given the relevance of this ancient beverage in the European context, this work aims to assess the environmental performance of two different types of beer production: the industrial and the craft beer, focusing on the Italian context.

The environmental burdens are assessed through the LCA methodology, and the functional unit adopted is one hectoliter of beer. Concerning the industrial representative beer, the list of ingredients and materials created by the European Commission together with the stakeholders of the European beer sector for the PEF pilot phase is used as input data. The list represents an average European beer recipe. Concerning the craft beer, an Italian beer by an emerging brewery located in Novara has been analyzed. The brewery directly provided primary data used in the assessment. The results of these two LCAs are obtained using the Simapro 8 software and CML-IA-baseline characterization method. Moreover, a preliminary evaluation of water footprint is made evaluating and comparing the water depletion category included in the ReCiPe Midpoint (H) V1.11 method.

Comparing the results obtained from the two analysis, it emerges that the hectoliter of industrial beer is characterized by higher environmental performance in all the categories considered. This outcome is mainly evident in the GWP category, where the industrial beer causes 31.4 kg_{CO₂eq.}, while the craft one causes 67.4 kg_{CO₂eq.}. The larger use of cereals in the recipe of the craft beer and the lower energetic efficiency of the plant represent the main causes of the obtained results. Moreover, the industrial processes adopt different techniques in order to make the production faster and more efficient. For instance, one of the most effective procedures is the substitution of barley malt with sugars and other unmalted cereals (e.g., maize). It allows reducing the time and energy required for wort production and accelerate the brewing process itself, but the quality of the final product is affected.



To conclude, the LCA methodology allows to highlight which are the most relevant impacts caused by the beer production and which are the unit processes characterized by the highest impacts. On the other hand, it cannot let emerge the efforts made by the craft brewery in increasing the quality of the input ingredients and the benefits obtained thanks to the attention put on each step of the whole supply chain.

Supporting data:

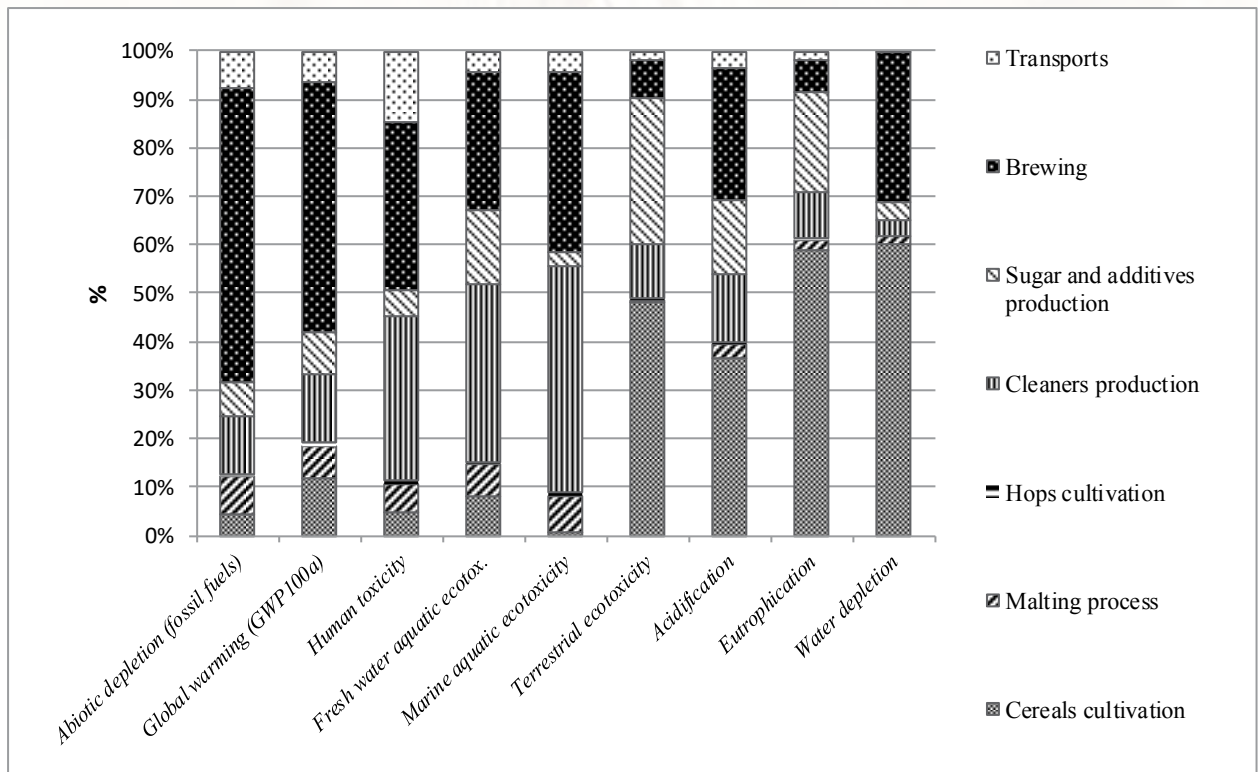


Figure 2: LCIA results of the representative industrial beer (CML-IA baseline method, ReCiPe Midpoint V1.11 method for the Water depletion)

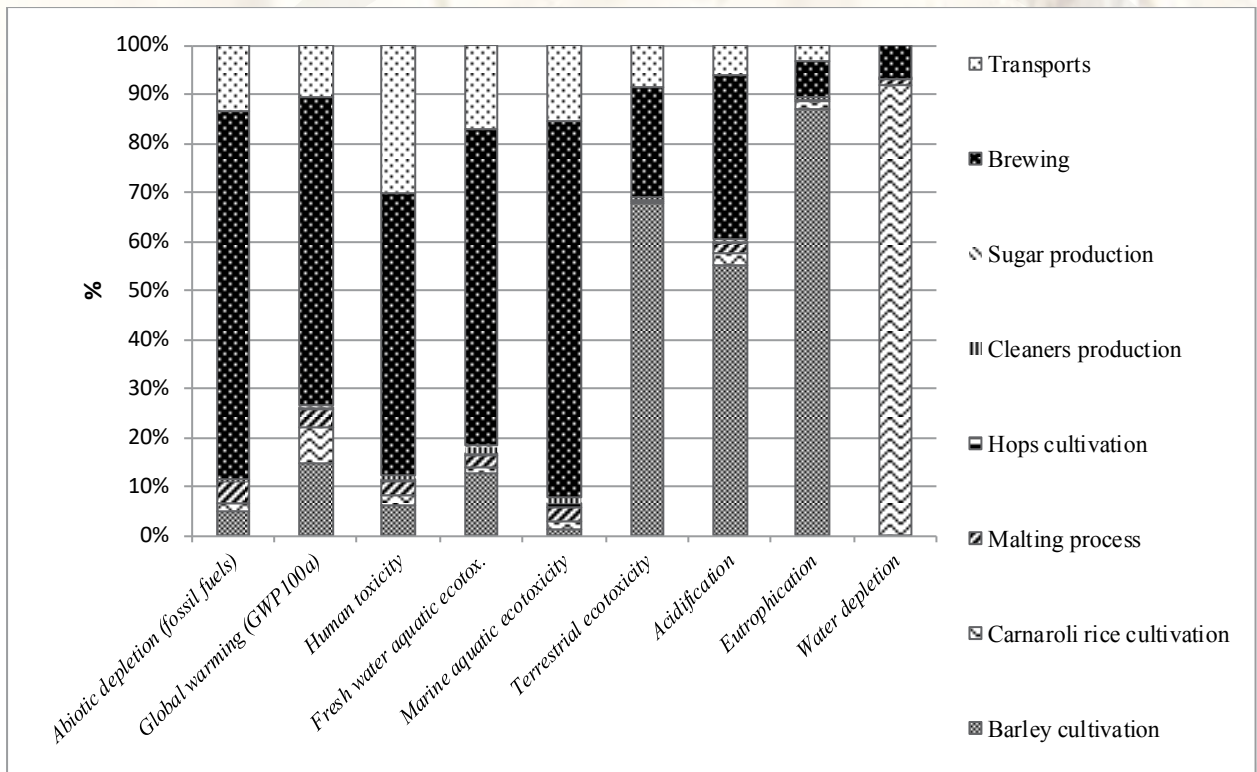


Figure 3: LCIA results of the Italian craft beer (CML-IA baseline method, ReCiPe Midpoint V1.11 method for the Water

P74. CFP of Spanish clementines: contribution of post-farm stages and variability assessment

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The goal of this study is to calculate the carbon footprint (CFP) of Spanish clementines. The contribution of the postharvest treatment and transport are also assessed extending the conventional approach focused on the farming stage. Furthermore, a variability assessment is carried out to determine the distribution of the average CFP.

Data from 21 farms (10 organic and 11 conventional) together with data from 1 postharvest central were gathered to calculate the CFP of 1 kg of packaged clementines following the PAS 2050. For each type of farm, the bootstrap distribution of the CFP mean was built by taking 10,000 Monte Carlo (MC) samples of the same size as the number of farms of each type. The variability of the main parameters of the post-farm stages (transport distances, degreening and refrigeration time, % marketable clementines) was assessed by using a MC simulation. In this way, the bootstrap distribution of the post-farm stages CFP was also obtained. The farm bootstrap CFP distribution was summed to the post-farm one obtaining a joint distribution from cradle to wholesaler.

Plotting the histograms of the bootstrap distribution of the mean CFP for conventional and organic clementines allows the qualitative difference in the means to be ascertained (Fig. 1 and Fig. 3), that is, their relative proximity. The histograms show the most likely distribution of the population average CFP for each farming system. The average CFP of the conventional oranges is 0.51 kg CO₂ eq. while the one of the organic farms is 0.34 kg CO₂ eq. The farming stage of the conventional citrus is the one contributing the most to the CFP (0.29 kg CO₂ eq.), followed by the transport to Europe (0.13 kg CO₂ eq.). For organic clementines, the average contribution of the farming stage (0.12 kg CO₂ eq.) is comparable to that of transport.

The results stress the high variability of the CFP of clementines, which could be extrapolated to other fruits. Farmer practices have a great influence on the farm CFP. Nevertheless, product seasonality (which mainly determines the need for degreening treatment and the % marketable clementines) and market characteristics (which affects the refrigeration time and

the transport distance) are the main drivers of post farm CFP, aspects that cannot be easily controlled. Finally, it must be highlighted that this study introduces a way to assess the variability of CFP when the reliability of data sources is low.

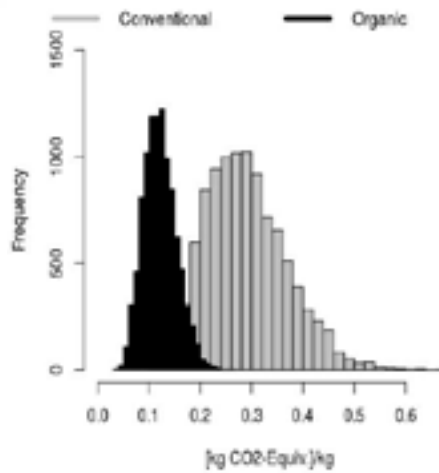


Figure 1. Histograms of the distribution of the mean of the farming stage CFP for the bootstrapped samples

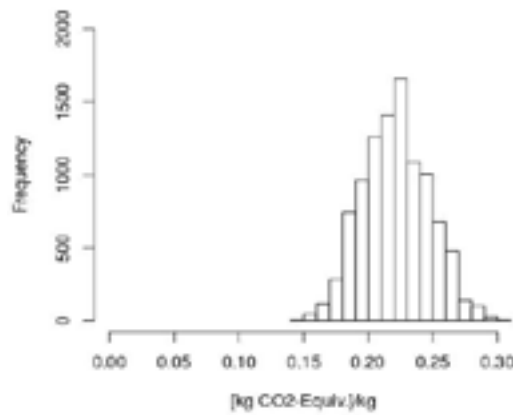


Figure 2. Histograms of the distribution of the mean of the post-farm stages CFP for the bootstrapped samples.

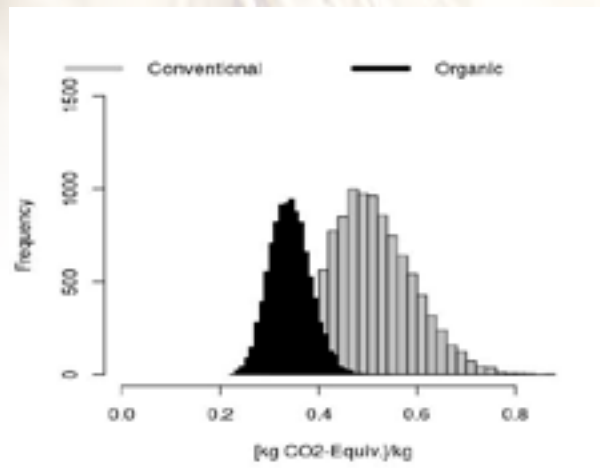


Figure 3. Histograms of the distribution of the mean of the clementines CFP for the bootstrapped samples.

P75. Environmental assessment of strategies for production of stabilized yeast

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Yeast are widely used for producing fermented (bread, beer...) and health benefit (probiotics) products. The production of stable and active yeast involves fermentation, concentration, protection, drying (stabilization) and storage. During the stabilization and storage steps, the cells face numerous stress which may deteriorate functional properties and cause cell death. Different strategies can be used to preserve cell survival, such as changing growth medium for fermentation or adapting process conditions (time, temperature).

This work aims at i) performing environmental analysis of production process of stabilized yeast and identifying hotspots; ii) comparing different scenarios of varying conditions of fermentation (growth medium with vs without cysteine) and of drying (45 °C during 90 min vs 60 °C during 60 min). SimaPro (PRé consultant) has been used for the Life Cycle Assessment modeling with ILCD 2011 method. With the purpose of meaningful comparisons, the impact scores were weighted by the final yeast survival, quantified by cell cultivability.

Fermentation appeared as the main hotspot due to its energy and water consumptions. The addition of cysteine improved the yeast survival leading to less yeast necessary to produce, and consequently decreased the total environmental impact of the system. Drying at 45 °C during 90 min had a higher environmental impact than drying at 60 °C during 60 min because drying temperature did not impact energy consumption as much as drying duration.

The comparison between microorganisms' stabilization scenarios highlighted the relevance of a life cycle approach to identify hotspots and suggested options for decreasing the environmental impact.

P76. The work conditions in the Brazilian sugarcane industry: an application of Social Life Cycle Assessment (S-LCA)

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Brazilian sugarcane production has grown more than 500% in last three decades, setting Brazil as a world leading producer and exporter of sugar and ethanol. Notwithstanding, there are many studies stressing social issues related to working conditions.

We used S-LCA in compliance with UNEP guidelines to analyze working conditions in three phases of the sugar cane supply chain in the Mato Grosso do Sul state to the year 2012: Agricultural stage, Ethanol and Sugar production. We use the National Household Sample Survey (PNAD) database and our study covered the production and related processes from the sugar cane supply chain.

The subcategories assessed and the main results are in Table 1. To the subcategory “Freedom of Association and Collective Bargaining”, workers' “rate of unionization” was the indicator adopted. As expected, the lower rate of unionized workers occurred at the agricultural stage (16.98 %). Most of the rural workers enrolled in the cultivation and harvest of sugar cane have a lower education level than those workers working downstream. The higher rate of unionization identified was at the sugar production (40.02%). In Brazil, the Consolidation of Labor Laws states that the minimum age to work is 14 years old, young people up 14 years old may work as apprentice under special conditions. Given this, to the subcategory “Child Labour”, two range of age was assessed, 5-9 years old and 10-13 years old. Neither the agricultural nor the industrial sectors studied presented registers of children under 14 working. Concerning the subcategory “Fair salary” we compared the workers mean wage to the national minimum wage to the year 2012, (circa US\$ 338.67), so all three sectors have been paying wages above the national minimum wage. In addition, we were able to observe that the agricultural stage paid lower salaries compared to the industrial stages. To analyze the subcategory “Hours of work” we chose as parameter the national weekly allowance according to the Consolidation of Labor Laws; where the maximum working hours may not exceed 8 hrs per day or 56 hrs per week. The sector with the lower weekly working hours was the ethanol production (41.5 hrs), followed by sugar production (45.8 hrs) and agricultural stage (45.9 hrs).

Our results showed that the sugar cane supply chain accomplished the Consolidation of Labor Laws regulation to the categories and year studied herein. There are some differences in the same supply chain in relation to social issues among sectors; e.g. the worst performance of agricultural stage in most subcategories.

Table 01: Results of subcategory assessment - Stakeholder worker

Subcategory	Inventory indicator	Unit of Measurement	Results by sector		
			sugarcane cultivation	ethanol production	sugar production
Freedom of Association and Collective Bargaining	Rate of unionization	percentage	16.98%	27.25%	40.02%
Child Labour	Rate of child labor 5 to 9 years	percentage	0%	0%	0%
	Rate of child labor 10 to 13 years	percentage	0%	0%	0%
Fair Salary	Mean wage from main job	US\$	576.4	854.11	1175.75
	Minimum wage required by law	US\$	450.2	450.2	450.2
Hours of Work	Maximum hours of work permitted by law	hours	8 hours day or 56 hours/week	8 hours day or 56 hours/week	8 hours day or 56 hours/week
	Number of hours worked per week in main job	hours	45.9	41.5	45.8

POSTER PAPERS

LAND USE AND EUTROPHICATION

13. Land Use and Eutrophication

P77. Using the direct LUC tool and Agri-footprint crop model to understand carbon footprint trends in oil crops.

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Abstract:

Blonk Consultants has developed tools to generate life cycle inventory data on crops for Agri-footprint and to generate data on the emissions of land use change due to crop cultivation. The research presented in the paper builds on these tool to gain insights in the temporal sensitivity of the carbon footprints of three major oil crops. The research question is two-fold: how do choices on time-scales affect the results, and how did the carbon footprint of oil crops develop over time?

For the expansion of crop areas (and potential related reduction of forest area) the time horizon used plays an important role. An often used timeframe is 20 years, which is somewhat arbitrarily chosen. Our analysis shows that the relative expanded crop area indeed depends on the timeframe, but leads to different outcomes for the different crops when it is varied. The graphs (page 2) show the % of the crop area that is changed due to expansions of crop area in a country using various timeframes. The Direct Land Use Change Assessment Tool uses this percentage to calculate the loss of carbon from deforestation which leads to the CO₂ emissions from land use change at that given year.

- 1) For oil palm fruit from Indonesia, if longer time frames are chosen, nearly all of the crop is essentially considered to have been expanded in that time frame and the impact mainly depends on over how many year it is averaged, so the impact decreases with longer timeframes.
- 2) For soybeans from Brazil, the picture is the same, but the outcomes are more variable, depending on the reference year and time frame.
- 3) For rapeseed in France, there has been a relatively steady linear increase of production area, with large year-on-year fluctuations. The time frame has relatively little influence, but the data variability explains the outcomes.

In the paper that we intend to submit to the conference, these results will be put into a broader context by considering other CO₂ related trends in crop cultivation (such as the quantities of N-fertilizers applied), to gain a better insight in the overall carbon footprint of crops in the last decades.

	Percentage harvested area that has expanded (different timeframes)	Area harvested and production
<p>Oil palm (Indonesia)</p>		
<p>Rapeseed (France)</p>		
<p>Soybean (Brazil)</p>		

P78. Evaluation of the Integration of Regionalized Land Use Aspects in LCA for Bio-based Products

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One of the central objectives of the bioeconomy concept is the substitution of fossil materials and energetic resources with biogenic ones. The project “Evaluation of regionalized land use and biodiversity aspects in Life Cycle Analysis (LCA) for bio-based products” funded by the government of Baden-Württemberg, Germany represents a considerable effort at improving and refining methods to integrate land use effects in life cycle impact assessment.

We present a LCA study of corn grains and short rotation coppice (SRC) with poplar at the same geographic location aiming 1) at demonstrating the impact category land use and 2) evaluating the effects of two different functional units, mass and area, on the impact category land use. Impact assessment includes the categories climate change, acidification, eutrophication, as well as the land use impact categories calculated with LANCA[®] (Land Use Indicator Value Calculation in Life Cycle Assessment) erosion resistance, mechanical and physicochemical filtration, groundwater replenishment, as well as soil organic carbon (SOC). Figure 1 shows the results of the impact categories climate change, acidification and eutrophication per kg dry matter (DM) of corn and per kg dry wood from SRC. Considering the impact categories climate change, eutrophication and acidification, SRC is favourable due to low pesticide and fertilizer applications as well as low management effort. Figure 2 presents the erosion resistance, occupation impact category measured in kg soil loss. Two figures are shown for corn and SRC: First the impacts per m² and second per kg dry matter. As the yield of SRC per hectare for the considered location is lower than the yield of corn, the impacts of corn are lower considering the functional unit of 1 kg DM.

The results show that especially for agricultural products the impact categories related to land use can differ from the classic emission-based impact categories such as eutrophication or acidification, even more so when taking yields into account and not only e.g. 1 ha. In our example SRC (FU 1 kg DM) is in favour regarding climate change, acidification and eutrophication but corn is in favour regarding erosion resistance. Therefore, any LCA of

agricultural products including land using processes must include land use impact categories, e.g. as part of the development of new products and processes, as is recommended by ILCD.

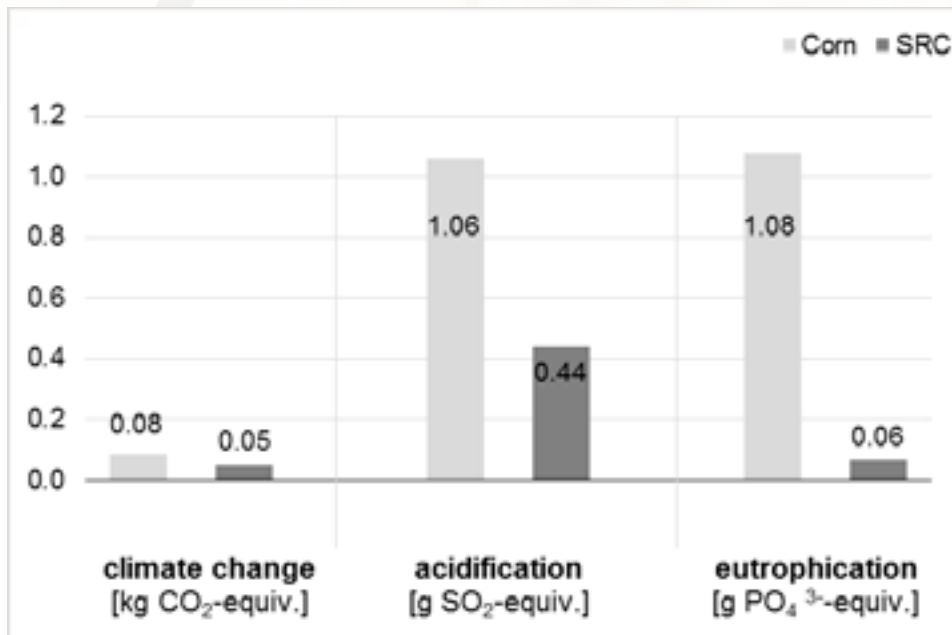


Figure 1: Impact categories climate change, acidification and eutrophication per 1 kg dry matter corn and short rotation coppice (poplar wood)

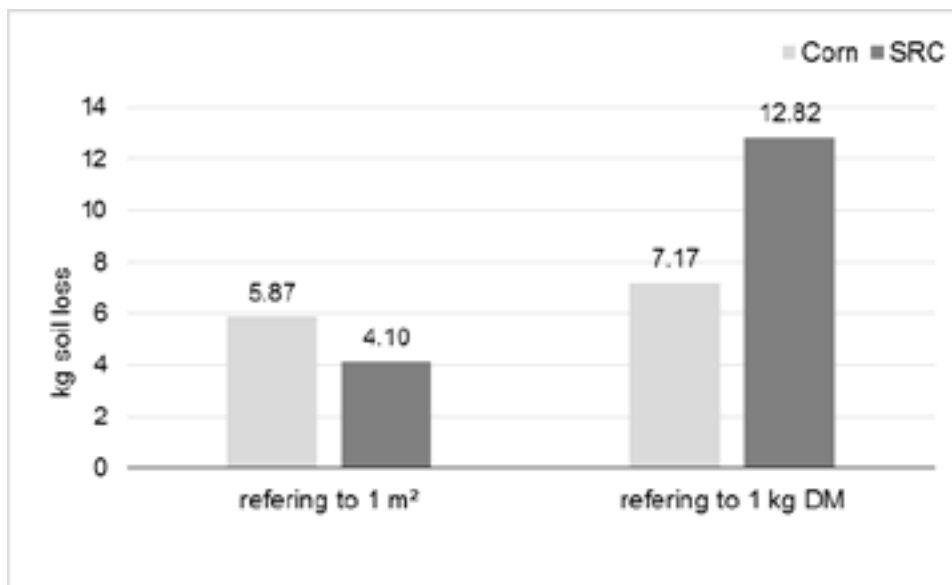


Figure 2: erosion resistance, occupation impact per 1 m² cultivated area corn and poplar wood and per 1 kg dry matter (DM) corn and poplar wood

POSTER PAPERS

FOOD PROCESSING

14. Food Processing

P79. The Dairy Product Environmental Footprint project - lessons and visions from the pilot project

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Objective of the work

The DAIRY PEF aims to develop both a specific methodology – known as Product Environmental Footprint Category Rules or PEFCR – and communication guidance on the environmental footprint of different dairy products. The project examines a broad array of environmental issues, including the carbon footprint, water use, other different emissions, land use change and allocation questions, in order to create a comprehensive assessment of their environmental impacts over their full life cycle. In addition, it will develop communication vehicles for the PEFCR.

Methodological detail

The project assessed existing footprinting methodologies in the dairy sector and defined the scope of five subcategories of dairy products (liquid milk, dried whey, cheeses, fermented milk, butterfat). A modelling framework and screening allowed identification of the most relevant life cycle stages & processes (see figure 1), the elaboration of the detailed PEFCR for the five categories of dairy products and the most relevant impact categories. The project also includes public consultations, open to all stakeholders, supporting studies as test of the methodology on real products, a communication phase with testing of different tools to improve the footprint, and undergoes approval by legislators and NGOs at different stages, including methodological choices as e.g the inclusion of food waste (see figure 2).

Main results of the study

From the study emerged that the most relevant life cycle stages are raw milk production (relevant processes: feed production, enteric fermentation, manure storage), dairy processing (relevant processes: energy carriers, water use, wastewater treatment), packaging (relevant processes: raw materials manufacturing), and distribution (relevant processes: chilled transports, chilled storage at retail, transport by the consumer). Besides, the relevant impact categories are climate change, water resource depletion, freshwater eutrophication, marine and terrestrial eutrophication, freshwater ecotoxicity, land use and acidification, plus additional information required on biodiversity.

Conclusions

The methodological part has built the new basis for all future LCAs in the dairy field and beyond. The results from the real product studies will further define usage options and validate tools for communicating on dairy LCAs. The PEFCR may help consumers to make more «ecological» choices, manufacturers to reduce the impacts associated with dairy products and communicate their environmental merits, and policy makers to refine environmental

legislation and elaborate incentive measures. On the downside, LCA methodology is quite complex to use and has some limitations (impact categories, data availability, nutritional component); it will still need overall evaluation to show if the methodology is ripe for e.g. use in numeric product comparison, or if the methodology could be of more general benefit to the dairy sector and its environmental knowledge and performance.

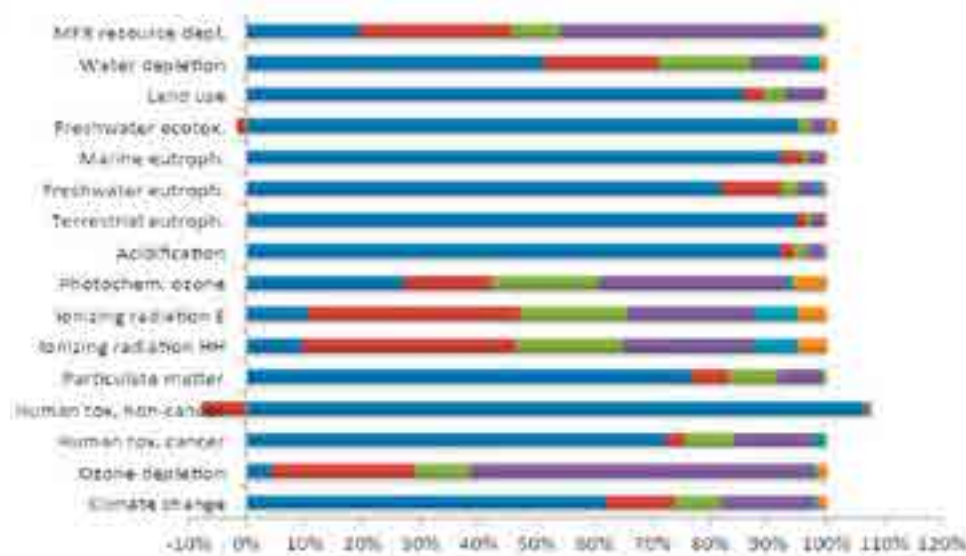


Figure 1: Characterisation results for the entire life cycle (per life cycle stage) - here for the subcategory of fermented milks

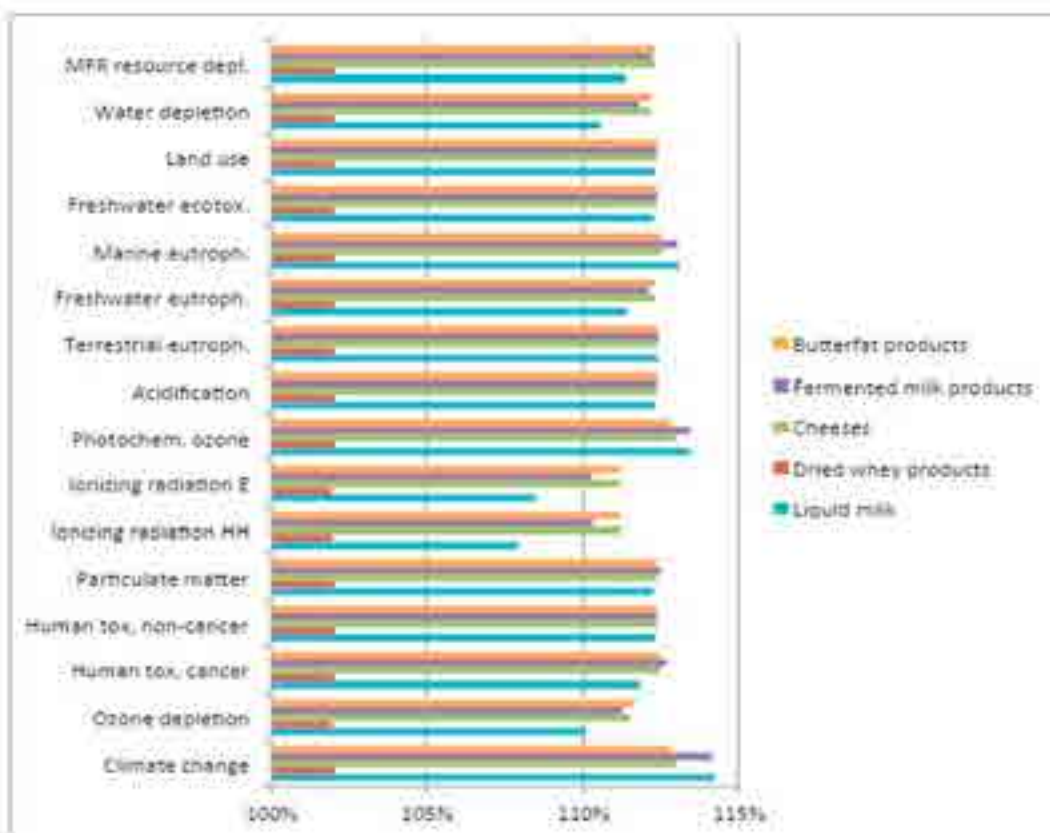


Figure 2: Impact increase due to food waste compared to default scenario for different dairy products

P80. Carbon Footprint of Grana Padano PDO cheese in a full life cycle perspective

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The aim of the present study was to assess the carbon footprint of Grana Padano PDO (Protected Designation of Origin) Cheese with a from cradle to grave approach. The study was submitted for an external critical review in order to verify the compliance with the ISO/TS 14067 (ISO, 2013).

The study involved two dairies (A and B) with 8 and 16 farms respectively. The functional unit (FU) was 1 kg of cheese aged for 12 months. Farm data were collected through farmer interviews, while the other data (from milk collection to end of life) were provided mainly by the dairies. On-farm GHG emissions were estimated following the IPCC Guidelines (IPCC, 2006a, 2006b), the off-farm and the total GHG emissions were assessed using the SimaPro 8.0 software (PRé Consultants, 2015) by the 100-year GWP (IPCC, 2013). The allocation between cheese and by-products (i.e. cream and whey) was done as proposed by the PCR: Yoghurt, Butter and Cheese (International EPD[®] System, 2013).

The results obtained were 16.02 and 15.84 kg CO₂.eq. kg⁻¹ FU for A and B respectively. The main contributor to the impact was the production of raw milk (75.3% and 73.9% respectively). The other process (from the raw milk collection through the product processing to the end of life) had a smaller impact. The use phase (cooled storage and product waste) played the main role in the GHG emissions of the post-farm steps.

Through the review of an accredited and independent body the two dairies obtained the ISO certification.

Table 1: CF of Grana Padano cheese [Kg CO₂-eq. FU⁻¹]

	Farm	Milk Collection	Milk Processing	Packaging	Retailing	Use	End of Life	TOTAL
A	12.06	0.29	0.49	0.13	0.18	2.83	0.04	16.02
B	11.71	0.10	0.86	0.19	0.08	2.81	0.08	15.84

Figure 1: Contribution of the different process steps [% of CO₂-eq. FU⁻¹]

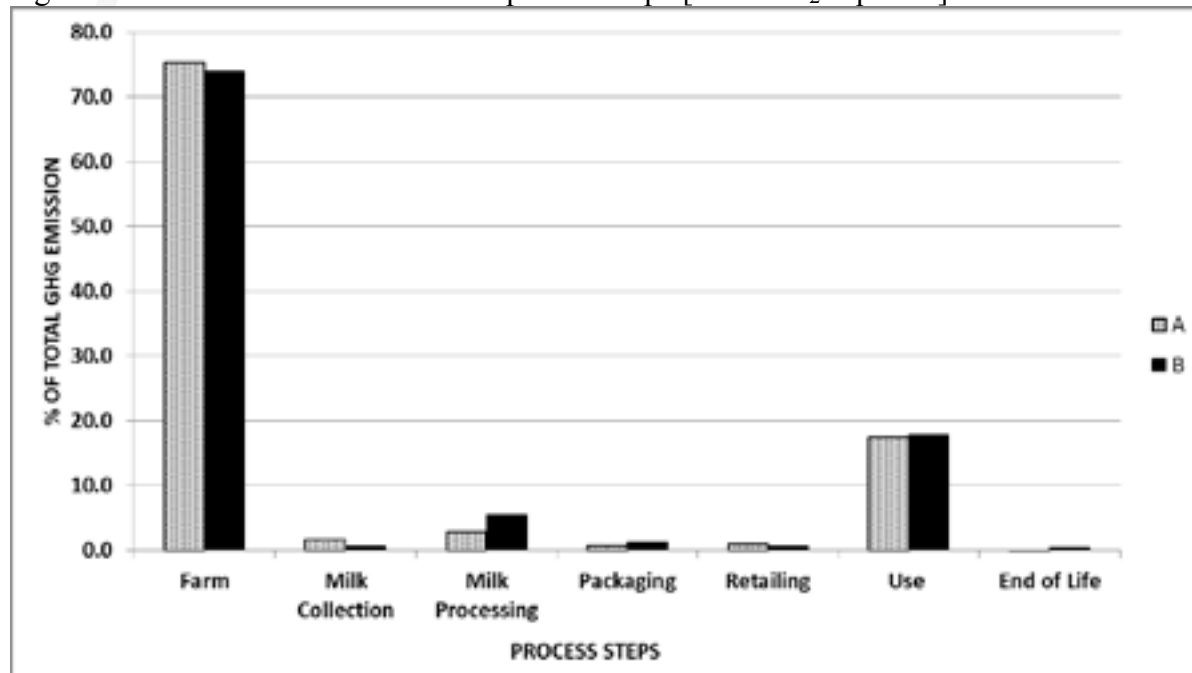


Figure 2: Contribution of the different substances and of the different compartments to total farm GHG emissions (% of total CO₂-eq. of farm activities).

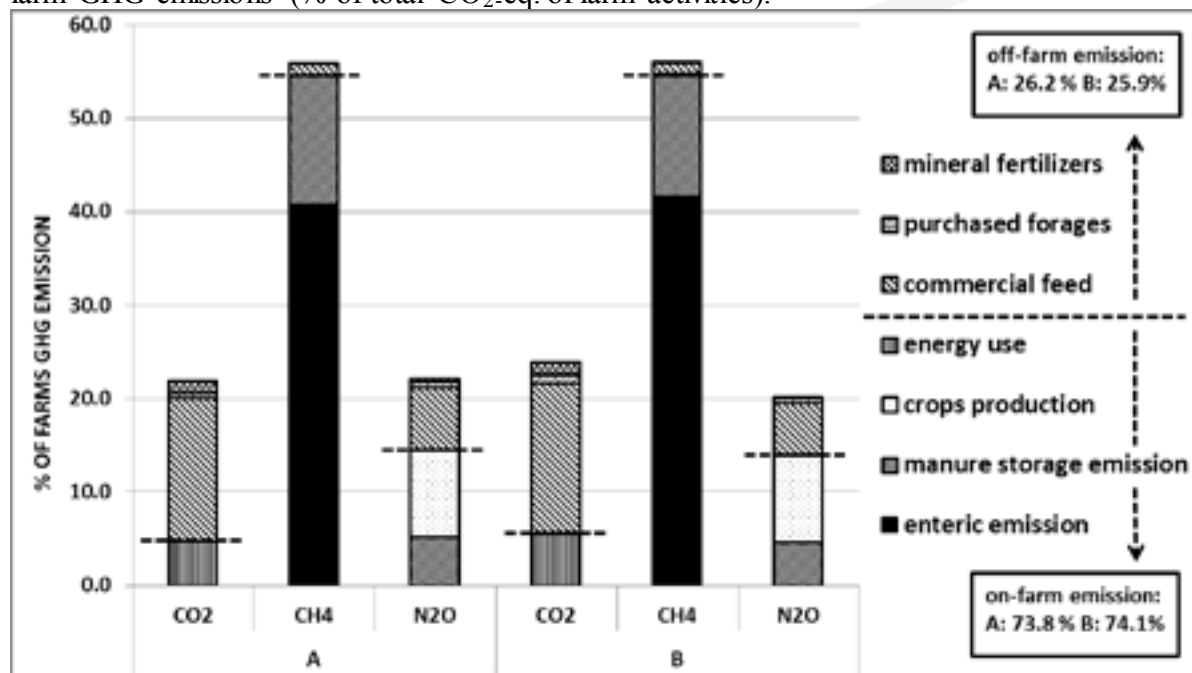
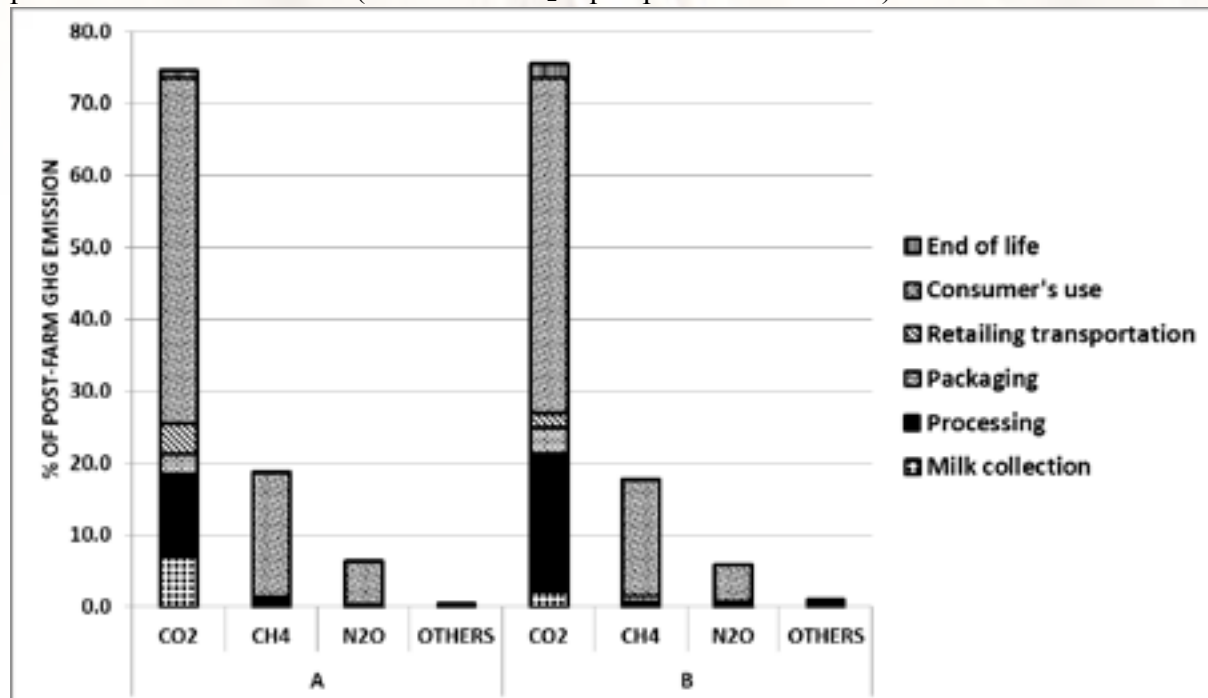


Figure 3: Contribution of the different substances and of the different compartments to total post-farm GHG emissions (% of total CO₂-eq. of post-farm activities).



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P81. Life Cycle Assessment of Grana Padano cheese production

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Objective: the aim was to evaluate the environmental impact of Grana Padano PDO cheese through a “cradle to cheese factory gate” Life Cycle Assessment.

Methods: system boundaries included all activities from initial producing and processing of raw materials (crop production, animal feeds, etc.) through milk production to cheese manufacturing. Primary data were directly collected in a representative dairy farm and in a cheese factory that produces about 3.6% of the total Grana Padano (tab. 1). The functional unit was 1 kg cheese. Dairy farm gas emissions were calculated using IPCC (2009) and EEA (2009) equations; impact categories were evaluated using midpoint ILCD method. An economic allocation was applied at cheese factory level among cheese, whey, butter, buttermilk.

Results: Global Warming Potential (GWP) per cheese kg was 15.7 kg CO₂ eq. (tab. 2) higher than GWP reported for fresh and semi-hard cheeses. Grana Padano has a low cheese yield (12.8 milk kg per cheese kg) due to the production process that requires curd cutting and cooking (until 56°C) and long ripening (>9 months), with high liquid losses. Over 90% of GWP, Acidification and Eutrophication were caused by milk production and upstream processes. In particular, 66.7% of GWP came from dairy farm (CH₄ and N₂O from enteric fermentations and slurry) whereas 36% derived from feed both produced on-farm and purchased.

Implications: mitigation strategies have to be searched primarily at farm level. Mineral and fossil resource depletion (MFRD) can be moderately mitigated reducing both purchased feed and milk transportation. Considering Grana Padano high nutritional value (high protein and fat content), impact can be profitably expressed per 50 g serving portion (GWP: 0.78 kg CO₂ eq.).

Table 1. Inventory and economic data from cheese factory (2015)

Input	Unit	Value
Milk	t/year	95882
Rennet*	g/wheel	17.5
Lysozime*	g/wheel	10
Salt	t/year	120
Natural gas [§]	MWh	8655
Electricity [§]	MWh	4747
Cleaning detergents [§]	t/year	31.5
Output		
Cheese wheels	N	183443
Average wheel weight	kg/wheel	35.9
Cheese	t	6592
Butter	t	206
Whey	t	61806
Buttermilk	t	1104
<i>Economic value</i>		
PDO Cheese	€/kg	7.30
Whey	€/kg	0.04
Butter	€/kg	3.34
Buttermilk	€/kg	0.18

*not included in LCA; [§] for cheesemaking and ripening

Table 2. Environmental impacts of the production of 1 kg Grana Padano PDO cheese and percentage contributions of inputs and activities

Impact category	GWP	Acidification	Terrestrial eutrophication	Freshwater eutrophication	Marine eutrophication	MFRD*
Unit	kg CO ₂ eq	molc H ⁺ eq	molc N eq	g P eq	g N eq	kg Sb eq
Total	15.7	0.35	1.54	2.13	66.2	0.12
<i>Percentage contributions:</i>						
Purchased concentrate feeds	22.7	7.11	6.38	56.1	52.6	57.3
Purchased forages	3.70	0.44	0.40	2.38	2.05	2.40
Bedding materials	0.91	0.44	0.35	1.34	2.25	0.79
In-farm feed production	9.61	28.8	29.1	24.9	29.3	17.6
Farm energy use	3.08	1.03	0.93	7.39	2.03	16.4
Housing and enteric emissions	36.9	31.3	31.9		5.06	
Slurry storage emissions	17.1	29.7	30.3		5.22	
Milk transportation (average 40 km)	1.25	0.37	0.39	0.76	0.83	4.31
Natural gas in cheesemaking	1.82					
Electricity in cheesemaking	2.16	0.55	0.19	3.74	0.43	0.34
Detergents in cheesemaking	0.32	0.17	0.03	2.82	0.17	0.42

Natural gas in ripening	0.13					
Electricity in ripening	0.32	0.08	0.03	0.56	0.06	0.05

*MFRD: mineral and fossil resource depletion

P82. More value with less harm: scrutinizing sustainable pathways for rest resource handling in food manufacturing empirically

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With an increased emphasis on circular economy, food manufacturers need to examine their material and energy flows and utilize every useful resource efficiently. Not only is resource scarcity an overhanging threat for several important resources, but maximizing economic output and minimizing environmental harm will be ever more important in coming years. Food producers cannot focus solely on their main product and treat the rest as waste but must examine the best ways to manage every input to their facilities. The main research question of this study is “what are the most sustainable ways to handle rest resources in food production systems”? Sustainability in this context encompasses the economic, the environmental and the social dimensions and all three will be evaluated simultaneously. The study looks into the dairy, brewery and slaughterhouse sectors in Norway and possibly Sweden and Denmark to investigate empirically the amount and composition of rest resources. It investigates existing and proposed combinations of rest resources and processing routes to create new products or raw materials for other industrial processes. These products are then compared to traditional ways of acquiring the same products or raw materials. Life cycle assessment forms the basis for evaluating economic (LCC), environmental (LCA) and social indicators (SLCA) and developing and choosing the right indicators is part of the study. The aim is thus to find quantitative measures of sustainability of rest resource handling, but the qualitative descriptions of how companies perform rest resource handling and how they evaluate different options are an important part of the study. The inclusion of different sectors within the food industry allows for cross comparisons to check for differences and similarities and learn across sectors. The study is based on a systematic search of available rest resources and existing and possible process technologies in the dairy, brewery and slaughterhouse sectors. The general research model is shown in figure 3.

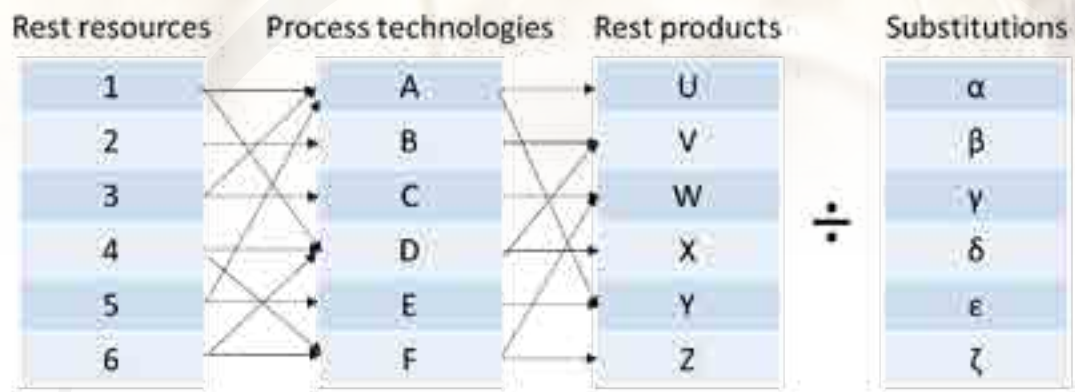


Figure 3 A sketch of the research process to find sustainable ways to handle rest resources in food manufacturing.

P83. Assessment of the Potential Improvement of Cheese Environmental Performance Achievable through the Use of an Innovative Real-time Milk Classification Service

Trevisan M1, Corrado S1

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The present work is a task of the Eco-innovation project More cheese from less milk: eco-innovative real time classification technology for optimized milk use (MilkyWay). It aims at assessing the potential environmental benefits achievable through the use of an innovative real-time Milk Classification Service (MCS), produced by Afimilk, an Israeli company. Thanks to optical observation, the MCS analyses milk coagulation properties on real time and allows to divide milk in two fractions, one suitable for cheese-making and one for fermentation and other fluid milk products. The use of the separated milk for cheese-making allows to increase the cheese yield and, therefore, to reduce the amount of milk necessary to produce a certain quantity of cheese, with a potential enhancement of the overall environmental performance of cheese production.

The study is carried out following the LCA methodology, according to ISO standards 14040:2006 and 14044:2006. The potential improvements of the environmental performance of cheese are assessed through the comparison of two systems: (i) base-case, without the use of the MCS (Figure 1) and (ii) improved, with the use of MCS (Figure 2).

The functional unit of the study is 1 kg of cheese and the system boundaries are set from the primary agricultural production to the gate of the dairy. Three farms are involved in the project. Primary data are collected from the farms and the cheese-making where the cheese from separated milk is produced. Emissions of methane from manure fermentation and manure management are estimated according to IPCC Guidelines for National Greenhouse Gas Inventory, whereas emissions of ammonia, nitrous oxide and nitrogen oxides due to the management and the application of manure and mineral fertilisers are estimated according to EMEP/EEA Air Pollutant Emission Inventory Guidebook 2013.

The trials on cheese-making with separated milk are still on-going, therefore complete results are not available yet and they will be presented at the conference.

Figure 1: Representation of the base-case system, without the MCS

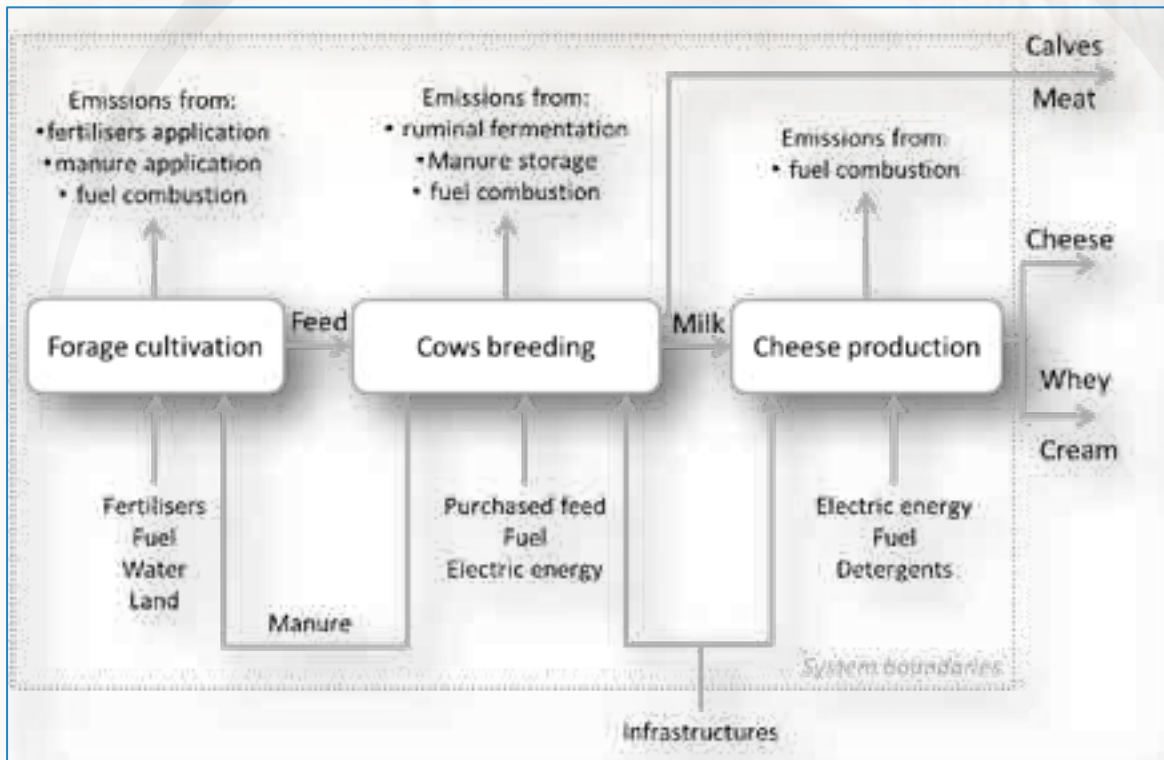
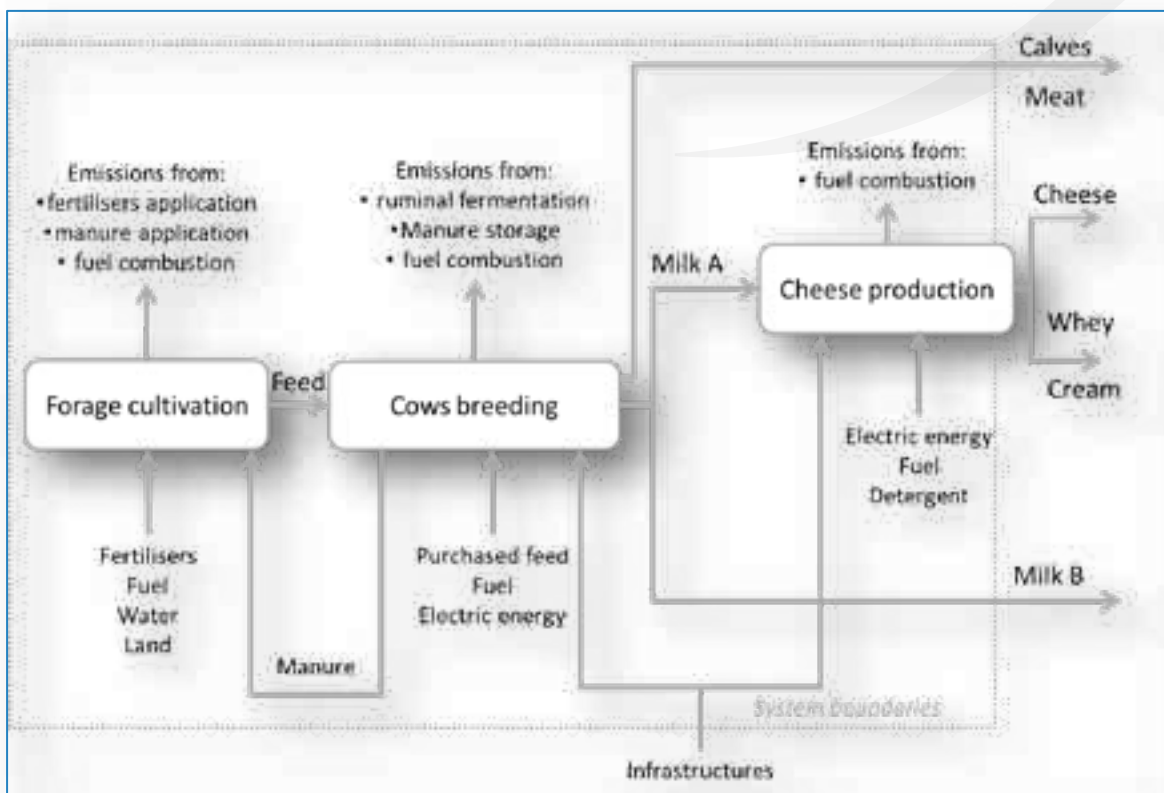


Figure 2: Representation of the improved system, with the MCS



P84. Benchmarking of Irish dairy processing with LCA

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The dairy industry is a vital part of the Irish agri-food sector, accounting for 29% of all food and drink exports in 2014 with a value of €3.06 billion. With the abolishment of European Union milk production quotas in April 2015, the dairy sector is expected to boom and will provide enormous opportunities to the Irish economy. However, the dairy industry faces increasingly stringent environmental regulations, for example, greenhouse gas emissions, energy and water efficiencies, and waste reduction. Until now, there has been no detailed study concerning the LCA of the Irish dairy processing industry and relevant studies are limited. As the industry is geared up for expansion, it is critical to understand how to maximise the value of additional dairy products without deteriorating environmental sustainability.

The objective of this work is to build an LCA model to analyse the energy, water and greenhouse gas emission intensities of Irish dairy processing. Data from a number of dairy processors have been undergoing through site visits and communication with site managers. System boundary is gate to gate, i.e. from milk leaving the farm gate to the moment when products are leaving the processor's gate. Major foreground (on-site) processes include raw milk transportation from farm to the processor, milk intake, product-specific processing, utilities, and wastewater treatment plant. Background (off-site) processes include electricity generation and transmission, fuel production and deliver, chemicals, packaging materials. Functional unit is defined as 1 kg of final product as produced, including butter, buttermilk, cheese, casein (and caseinate), whey products, and skimmed milk powder. Process specific energy and materials will be used where possible, otherwise allocation of inputs among multiple outputs will depend on product mass. Simapro 8.1 will be used in the LCA modelling. Ecoinvent 3.0 database will be used for background processes.

POSTER PAPERS

WASTE AND BI/CO-PRODUCTS

15. Waste and bi/co-products

P85. Quantification of food waste in the primary production sector in the Nordic Countries.

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The main aim of the project was to develop a framework for the of food waste in the primary production sector by developing and adapting methods for food waste quantification and define food waste in primary production. Calculation of total food waste amounts (including food waste treatment, reasons and reduction possibilities) in primary production in Denmark, Finland, Norway and Sweden was another aim.

Method development was done through case studies of eight products (carrot, onion, green pea, yellow pea, wheat, rye, aquaculture rainbow trout). The methods used in the case studies were questionnaires, interviews, and direct measurements in the field. Additionally, national statistics, previous studies and scientific publications were used to quantify food waste in the Nordic countries.

Primary production was defined as agriculture, aquaculture and fisheries, starting from when plants are ready for harvest, when farmed fish and animals are born and when milk is drawn and eggs are laid by bird. The system ends when the product is sent to processing or wholesale/retail. The term *side flow* was used instead of food waste to give a more holistic approach, differentiate the approach from the existing terms (Table 1). *Side flow* was defined as the edible part of product from primary production, produced with the intention to be eaten by human, but instead used for other purposes or was sent to waste treatment.

The total side flow amount in the four Nordic Countries was estimated as: 1030 tonnes, thus, 4 % of the total primary production (26 110 tonnes). It includes only the edible part of food that is produced for human consumption. Our study gave valuable experience of suitable practices and side flow case studies (Table 2), and resulted in a framework for suitable methods and definitions for future food waste studies in primary production.

Table 1. Differences in the terminology and definitions between this project and the FUSIONS (*) project

	This project	FUSIONS project
Term for wasted produce	Side flow	Food waste
Primary production	Includes the period from birth to slaughter in domesticated animals and fish. Includes the period from harvest to pre-processing.	Starts when domesticated animals and fish are ready for slaughter, otherwise the same. Includes the period from harvest to pre-processing.
Side flow/food waste	Includes only edible parts. Includes all produce intended for human consumption but not used as such.	Includes edible and inedible parts. Includes only produce sent to waste treatment, disposed in other ways or used for energy

(*) FUSIONS (Food Use for Social Innovation by Optimising Waste Prevention Strategies) is a European research project about working towards a more resource efficient Europe by significantly reducing food waste. See: <http://www.eu-fusions.org/>.

Table 2. Case studies results

Product	Total primary production side flow amount (% relative to total production)
Carrots	17,6 %; 20,8 %; 25,8 %; 13-31 %
Onions	8,4 %; 11,4 %; 21 %; 17-33 %
Green and yellow peas	16,5 %; 17,6 %; 17,7 %
Wheat	5,0 %; 6,6 %; 23 %
Rye	4,2 %
Char and Rainbow trout	5,6 %

P86. Accounting for impacts of production and disposal of food waste along the supply chain in an eco-design tool

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Objective To examine the ways to account for and present information on the impacts of food waste through the supply chain so that packaging designers and manufacturers can optimize the balance between packaging impacts and the product protection function of packaging

Background The FAO have estimated food waste to be in the range of 1.3 billion tonnes globally (FAO 2013). The UN sustainable development goal 12.3 is to halve per capita food waste by 2030 at retail and consumer level. PIQET (Packaging Quick Evaluation Tool) is a packaging eco-design tool which has been used by consumer goods companies for more than 10 years. Until now the tool has not directly included the impacts of product waste as part of the packaging assessment. In the new version of the tool, due out in May 2016, the impacts of manufacturing product which is wasted is included in the assessment using the product sector definition and environmentally extended input output data (Suh 2004). The tool also includes the impacts of disposal of wasted food and packaging along the supply chain.

Recommendations The inclusion of product waste at each stage of the supply chain requires an assessment of the manufacturing impacts up to that point in the supply chain. For example loss at filling leads to only loss of manufacture product, while loss at retail represents the loss of product, its package and the transport throughout the wholesale and to retail chain.

The impact of food waste disposal needs to include options for reuse of product in seconds markets, recycling into animal feed, and disposal routes such as composting or landfill. The impacts of potential food waste are presented to the packaging designer during the design process is the same environmental indicators that are used for packaging design i.e. global warming, eutrophication, land use, water use, resource depletion etc. Figure 1 & 2 show an example of greenhouse gas impacts of waste compared to product along the supply chain.

Implications and next steps. It is hoped that greater understanding of the relative impacts of food inside the packaging and the package itself will lead to a better optimization of packaging. Future developments of the tool aim to include more specific estimates of product impact based on company specific data, rather than relying on input output data.

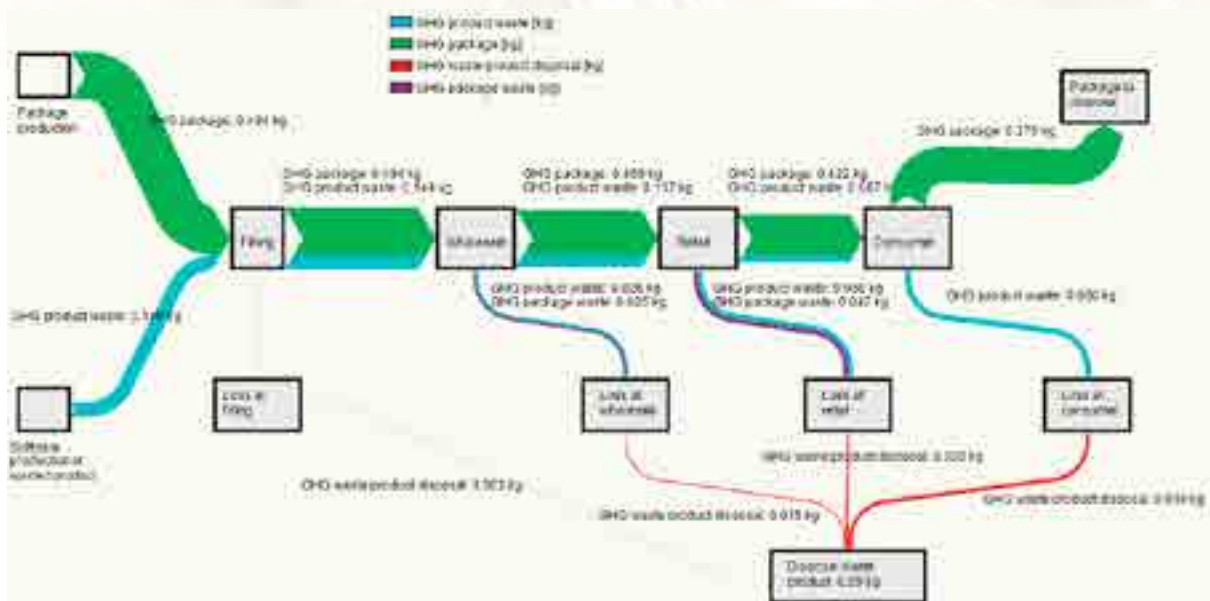


Figure 1. Fictitious example of and waste product GHG emissions.

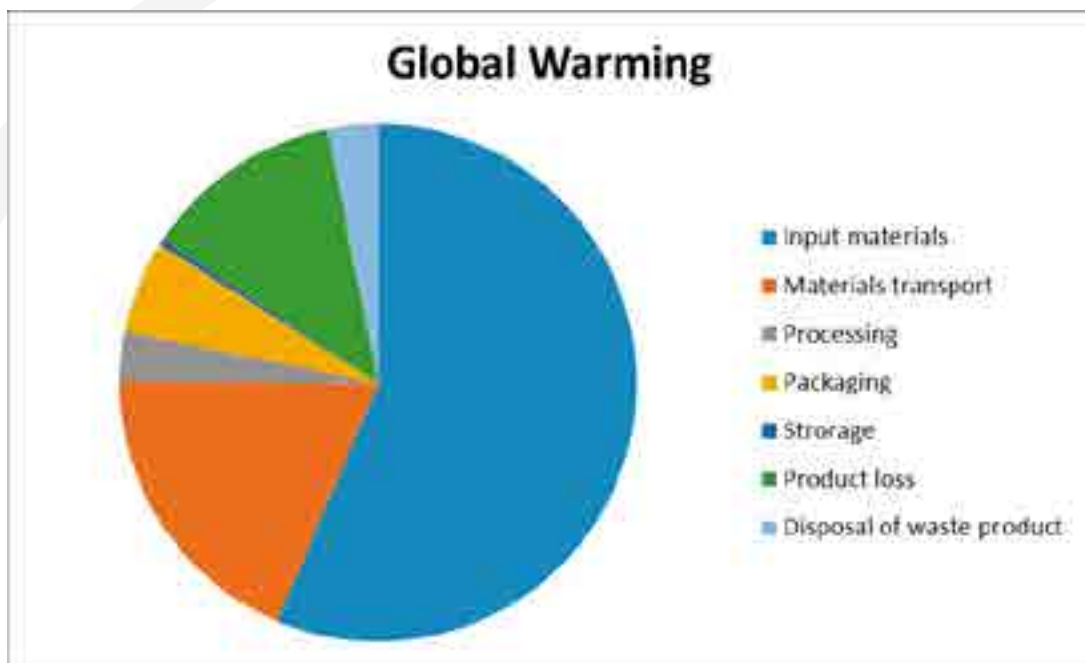


Figure 2. Fictitious example of global warming indicator contribution analysis of product, packaging, and product waste impacts.

P87. Bioeconomy and regionally contextualised LCA for biomass production

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With many environmental burdens associated with biomass production occurring and being induced at the regional level, there is a need to produce more regional and spatially representative life cycle assessments of biomass based systems and bioeconomy regions.

The RELCA modelling approach (O’Keeffe et al., 2016b) using a “within regional” life cycle scope (O’Keeffe et al., 2016a) was implemented to determine the regional distribution of GHG (greenhouse gas) emissions related to biodiesel production. RELCA combines geographical modelling with life cycle software, through the use of catchment delineation to assess the potential GHG of regional bioenergy configurations.

The results showed that the regional GHG variability cannot be captured with a simple regional average value and mitigation potential depends on production location within the region. Additionally, assessing both biomass and conversion plant configurations are needed for mitigation strategies.

With biomass foreseen to play a greater role in the future biobased economy (i.e. the use of biomass for food, materials, chemicals and energy); the ability to account for the heterogeneous geographical characteristics found within a region, using life cycle approaches, will be important to support more regional resource management..

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P88. Handling Of Organic By-Products And Residues In Lca Methods – A Case Of Five Food Industry Sectors

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2. Ostfold Research, Ostfold, Norway

This study investigates how organic by-products and residues from processing in five different food industry sectors are handled within LCA methods. By mapping LCA methods for handling residues within sectors of dairy, brewery, fishery, slaughterhouse and bakery, we aim to provide a more nuanced alternative to “zero burden” assumptions for residues, with implications for inventories and future studies of residue utilization. Figure 1 shows the idea that the upstream lifecycle is important to studies of residue utilization.

Relevant academic literature within the five sectors is identified (thus excluding grey literature). Studies from 2010-2016 including explicit handling of residues in LCA method are analyzed further, primarily regarding means for handling multi-functionality.

Results show that relevant literature is unevenly distributed between the industrial sectors (Table 1). The number of impact categories studied in each publication also varies. Allocation based on physical or economical properties appears to be the most common approach for handling multi-functionality in studies that include several (by-)products of economic value (e.g. dairy), or where a substantial amount of organic material or energy is found within residual products (e.g. fishery). This is despite the fact that half of the studies follow the ISO method guidelines which prioritizes system expansion. Where there is one main product in terms of mass and economic value, substitution seems to be more common (e.g. brewery). Choice of allocation basis is problematized in several papers.

In terms of inventory data we use sweet whey from dairy processing, where allocation is predominant, as an example. GHG emissions allocated to dry whey range from approximately 0.2 to 13 kg CO₂-equivalents per kg. The main difference is due to exclusion or inclusion of raw milk production. Currently, whey is seldom a problematic by-product, but methodological issues and considerations can inspire studies of other residual materials. The choice of environmental impact data for by-products and residues for use in future studies is not straightforward. With increased valorization and processing of residues and by-products for new products, representative and fair data will become increasingly important. Transparency regarding environmental burdens assigned to residues, or benefits credited to main products, appears an important point for improvement.

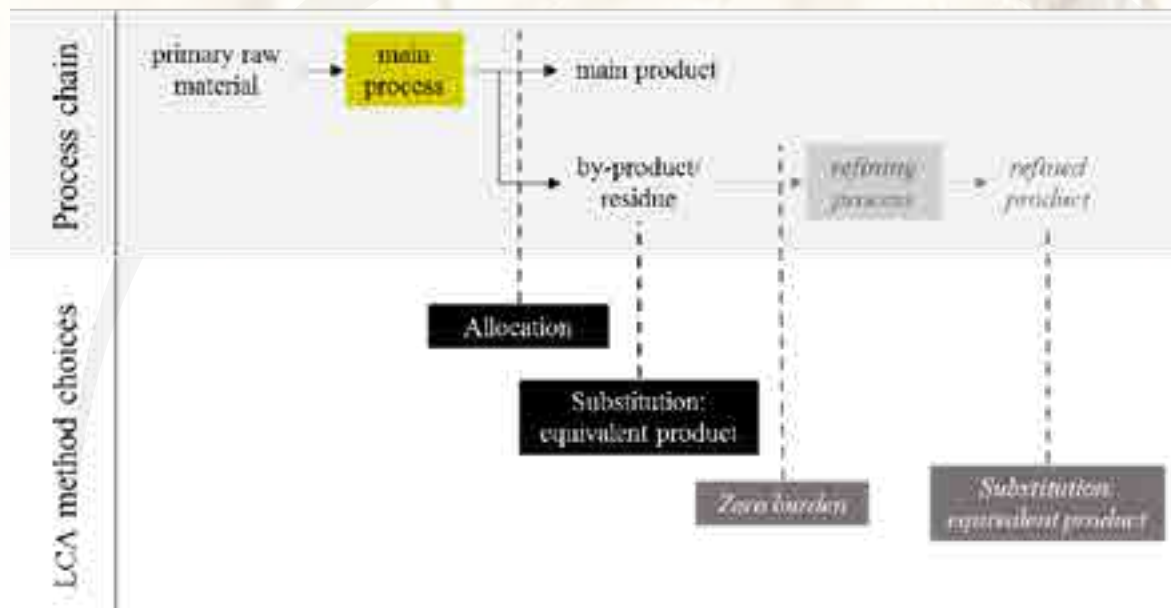


Figure 1. Schematic illustration of process chain for organic by-product or residue, including upstream and downstream life cycle phases, and relevant LCA method choices. A residual resource can be allocated impact from the main process, or provide benefits to the main process by substituting products of equivalent function. Both aspects could be important to consider in studies of further residue processing, in order to avoid sub-optimization and potentially misleading assumptions of “zero burden” resources.

Table 1. Overview of studies included in the analysis.

	Bakery	Brewery	Dairy	Fishery	Slaughter
No. of identified publications ¹	1	2	11	6	8
... whereof focus on methodology related to residues or by-products, or residue valorization	0	0	2	1	2
No. of publications applying					
... only allocation to residues	0	0	9	5	4
... only substitution to residues	1	2	1	1	2
... both allocation and substitution for residues	0	0	1	0	2
No. of publications assessing					
... more than 2 environmental impact categories	0	2	4	4	5
... only GHG emissions	1	0	6	1	2
... only energy	0	0	1	1	0

¹Only including studies where processing residues are handled in calculations.

P89. Environmental Trade-offs between Food Packaging and Food Waste: Waste Rate Data Challenges

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Food packaging has long served a role in protecting and preserving both perishable and shelf-stable foods, but sustainability efforts aimed at reducing the environmental impact of packaging often overlook this critical role. Food waste can represent a significant fraction of the food life cycle's overall environmental burden. This presents an important research and design question: can investments in resources and associated emissions due to improved packaging technologies be justified from an environmental standpoint if they contribute to reductions in food waste? Here we use LCA to demonstrate the trade off in environmental impact between food packaging and food waste for a number of specific case studies in a U.S. context. Emphasis is placed on demonstrating the challenges faced in establishing empirical food waste rate data.

A LCA model was developed with specific attention to food waste effects. The functional unit is mass of food *eaten*. Impact of food production and processing relies on previously published LCA studies. Packaging manufacturing, product distribution, retail energy use, and home refrigeration are modeled using processes from the Ecoinvent database. Food and packaging disposal follows the EPA's WARM model. While consumer level food waste is included, scenarios are differentiated by retail-level waste rates, which are derived from empirical data from food retail partners.

Establishing food waste rates, especially at the particular product level necessary for comparing packaging scenarios, is extremely challenging. We discuss approaches ranging from national level estimates to interviews with individual store managers to company-wide inventory data. Table 1 compares waste rates from opposing ends of this spectrum. Further, consumer-level food waste plays a pivotal role in life cycle impacts, but sound empirical data doesn't exist. We discuss potential approaches to acquiring such data. With these waste rate uncertainties in mind, we explore the trade-offs in energy use and greenhouse gas emissions between food packaging and food waste through a number of case studies in the U. S. One such example is shown in Figure 1.

Packaging has a role in reducing food waste and improving overall system environmental performance. Current data availability limits the extent to which this role can be quantified.

Table 2. Retail-level food waste rates for a select number of food categories, comparing empirical values based on sales and waste data from a US retailer (averaging multiple products in each food category) and values available through USDA's Loss Adjusted Food Availability dataset.

Food category	Waste rate data from Retail Partner ^a					LAFAs ^e
	number of products averaged ^b	total sales of all products over 2 years	mean waste rate ^c	Standard deviation ^c	sales weighted average ^d	"% Loss from retail/institutional to consumer level"
Orange Juice	14	\$28,758,141	1.37%	4.16%	0.62%	6.0%
Potatoes	5	\$5,647,066	1.50%	2.56%	0.42%	6.5%
Cheese	8	\$1,470,846	3.83%	4.45%	3.42%	6.0%
Milk	21	\$279,594,866	1.09%	1.25%	0.39%	12.0%
Pork	6	\$28,698,870	4.08%	1.68%	3.86%	4.4%
Spinach	3	\$14,328,032	2.61%	3.19%	0.87%	14.4%
Eggs	5	\$27,415,887	0.0007%	0.0005%	0.0005%	9.0%
Tomatoes, chopped, canned	4	\$2,360,999	0.18%	0.07%	0.19%	6.0%
Tortilla Chips	6	\$6,204,647	0.84%	1.47%	0.42%	6.0%
Beef	9	\$45,853,048	6.97%	7.95%	2.80%	4.3%

^abased on sales and "throwaway" tracking from an anonymous US retail chain, totaled over multiple storefronts and 2 years of sales.

^bnumber of individual products of given food category included in estimates

^cArithmetic mean (and standard deviation) of retail-level waste rate of the products included in the category

^dAverage retail-level waste rate, weighted by the total sales of each product in the given category

^efrom the Loss Adjusted Food Availability (LAFAs) data, maintained by USDA ([http://www.ers.usda.gov/data-products/food-availability-\(per-capita\)-data-system.aspx](http://www.ers.usda.gov/data-products/food-availability-(per-capita)-data-system.aspx))

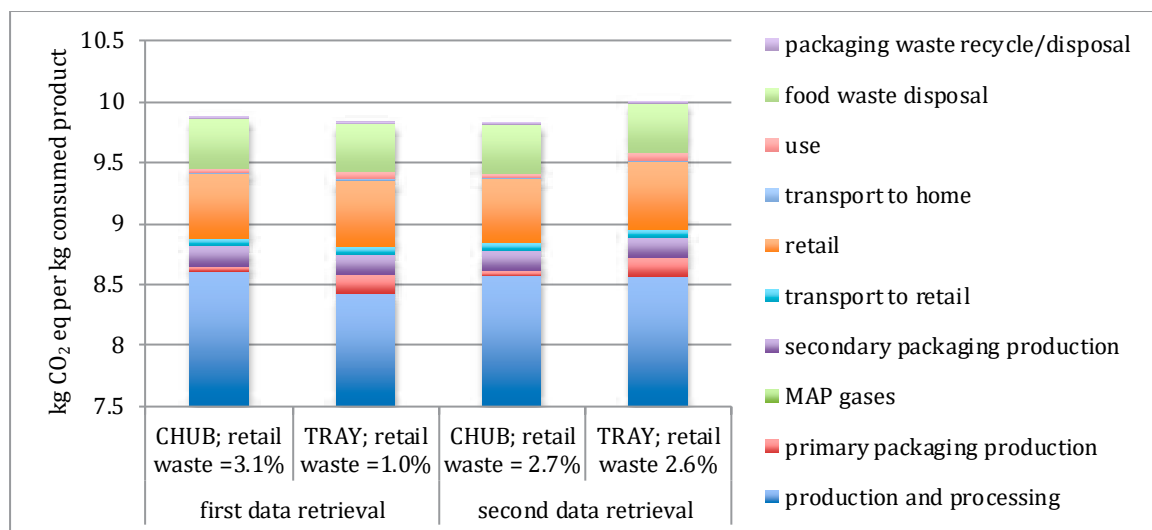


Figure 1. Example result of food waste/food packaging trade-off. Here, the case is ground turkey packaged in a 3 lb (1.4 kg) tube or "chub", compared with ground turkey packaged in a 3 lb (1.4 kg) sealed, Modified Atmosphere Packaging (MAP) tray. An initial waste rate data retrieval from retail partner showed reduced retail waste for the MAP tray, which was sufficient to compensate increased emissions from supplying the tray. A second data gathering nine months later indicates that the initial difference in waste rate between the packaging configurations did not hold, emphasizing the sensitivity of such comparisons to waste rates and the importance of high quality waste rate data.

P90. Environmental Advantages of Using Turkey Litter as a Fuel

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Systems-model based life cycle assessment (LCA) was used, which included all the processes of turkey production from feeds and farm fuels to litter management (Williams et al., 2015). The functional unit (FU) was 1000 kg live weight at the farm gate. Impact categories of global warming, eutrophication and acidification potentials and cumulative primary energy demand were included. Activity data came from industry partners and a company that generates electricity centrally from renewable materials. Changes in soil C were quantified with the RothC model (Figure 1).

The key differences between the litter management methods were that conventional management substitutes for NPK fertiliser, but incurs large ammonia losses and energy for land application while litter as a fuel substitutes for thermal energy, loses N, but retains PK fertiliser (Table 1). Diverting litter from land prevents a repeated source of C from soil leading to emissions of CO₂ from land. The main net effects were reductions in acidification and eutrophication potentials and cumulative primary energy demand of 70%, 55% and 14% respectively (Figure 2). Loss of soil C resulted in little effect on net GHG emissions.

The analysis is of two contrasting avoided burdens with one approach mitigating much ammonia loss, at the expense of losing N as plant nutrient. This has benefits for poultry production by enhancing its ability to help meet international obligations to meet acidifying gas emission targets. The loss of soil C eliminated possible climate change benefits and this must be recognised when diverting biomass from traditional destinations in soil. Although this analysis was applied to turkey, similar results can be expected with broilers, which represent much more litter production in total.

Reference Williams, A.G., Leinonen, I., Kyriazakis, I., 2015. J. Clean. Prod. doi:10.1016/j.jclepro.2015.11.044

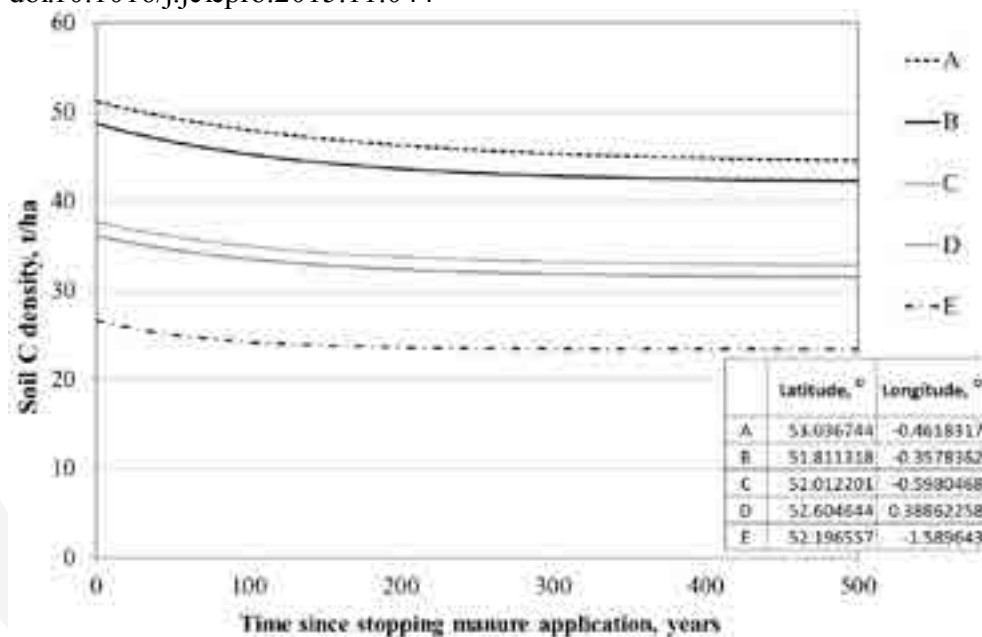


Figure 2 Simulated loss of soil C by stopping application of litter to land

Table 3 Main burdens of producing 1000 kg live weight up to the farm gate using conventional litter management

AP, kg SO ₂ Equiv.	EP, kg PO ₄ ³⁻ Equiv.	CPED, GJ	GWP, kg CO ₂ Equiv.
84	32	21	4500

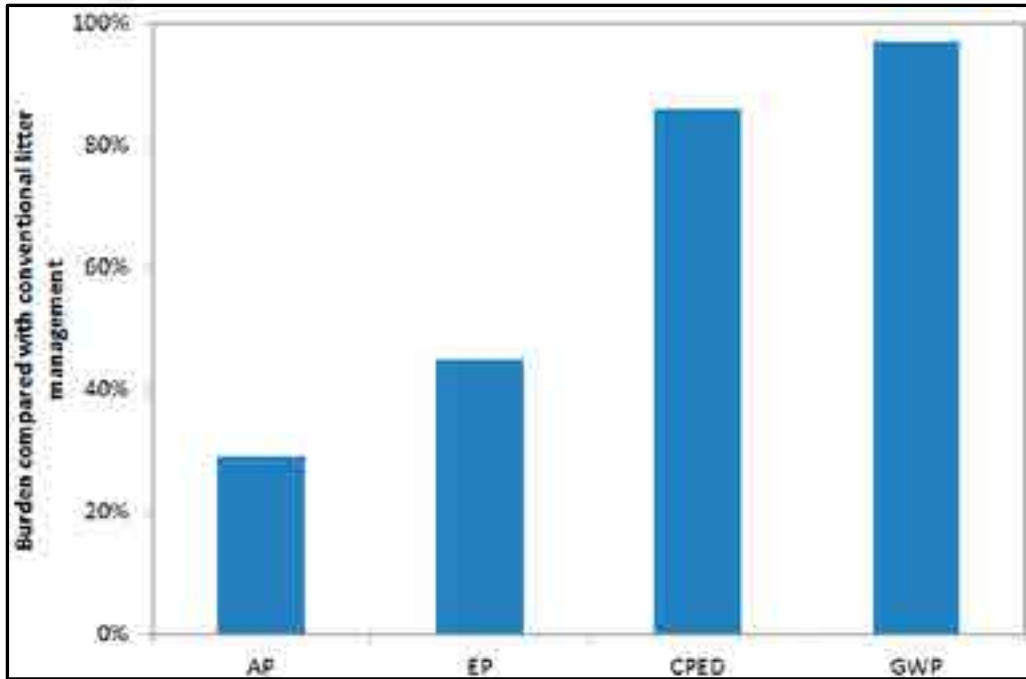


Figure 3 Burdens of producing turkey with litter as a fuel compared with conventional management as a fertilizer.

P91. Environmental Impacts of Food Waste in a Life Cycle Perspective

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The food production system has been acknowledged as a problem that needs to be addressed in order to achieve a sustainable society. Retail is an important player in the food supply chain and its influence spreads both upstream to suppliers and downstream to consumers. Therefore, this research aims to contribute to reduction of the environmental impacts related to food waste in retail, by identifying products with high environmental impacts. This study has two main goals. The first is the quantification of food waste produced by the supermarket and the second is to examine the environmental impacts of some products in order to assess the impacts generated by the waste production. For the first objective all food waste generated by the supermarket in one year was considered. For the second goal, a selection of products was made according to the following criteria: high level of disposal and high frequency on the generation. Frequency of generation means the number of days in the year that the product was wasted; this parameter was used in order to highlight products that were wasted in large quantities due to a single occurrence, such as a breakdown of equipment. The selected products for this stage of the research were beef, pork and chicken meat products; bread; strawberries, bananas, tomato, lettuce, potato, carrot, cabbage and apple.

A supermarket located in the city of Borås, Sweden, provided the data of the wasted food products. The store is considered to represent a typical mid-size urban store, with a sales area of approximately 400 m². The research assessed the gravimetric composition of solid waste from the supermarket in order to quantify the production of organic waste. The functional unit chosen was the yearly food waste disposed by the supermarket. The system boundary includes all relevant process, such as primary production, packaging, industrial processing, transportation, retail and waste treatment.

It can be observed from Figure 4 that bread was the category with the higher amount of waste and the majority was fresh bread produced at the supermarket chain bakery.

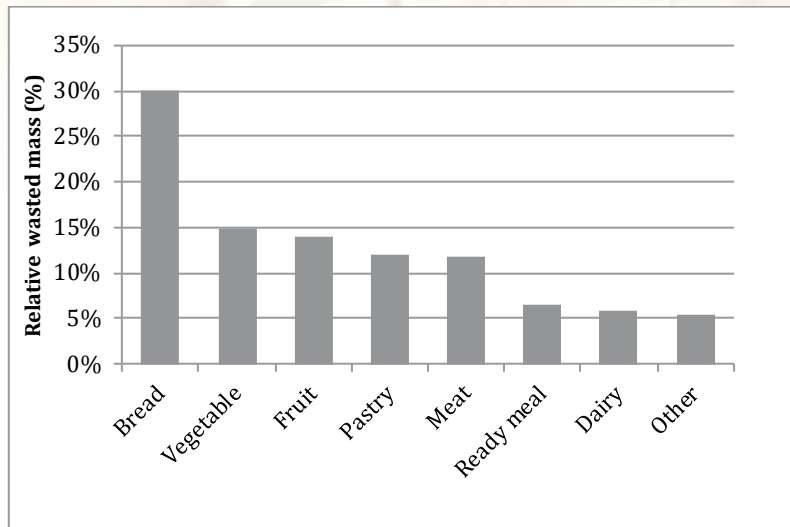


Figure 4 - Share of food waste from different categories in the store.

The difference between the distribution of the environmental footprint and the wasted mass for each product (Figure 2) highlights the importance of not only measuring the food waste in terms of mass, but also in terms of environmental indicators and costs.

The relative contribution of mass, costs and selected characterization impacts for waste generated in one year of the five different products categories are presented in Figure 5. Overall, the products with higher environmental impacts in the analysed categories were beef and bread. Bread has a great importance in terms of environmental impacts and cost. It is the product with higher contribution in several categories such as ozone depletion and resource depletion. Beef leads in categories such as climate change, eutrophication and acidification.

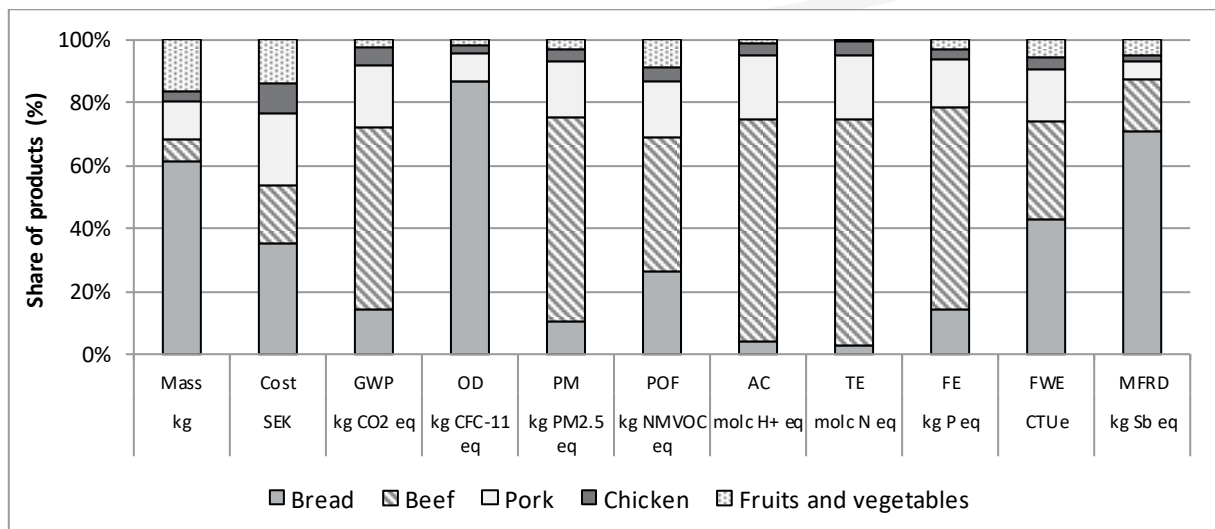


Figure 5 - Relative contribution of wasted mass, cost, climate change (GWP), ozone depletion (OD), particulate matter (PM), photochemical ozone formation (POF), acidification (AC), terrestrial eutrophication (TE), freshwater eutrophication (FE), freshwater ecotoxicity (FWE), and resource depletion (MFRD).

P92. Life cycle assessment of tomato pomace use options

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The California tomato processing industry produced approximately 388,856 t of tomato pomace in 2014. Whilst currently used as an animal feed ingredient, tomato pomace might also be valorized as a soil amendment for biosolarization, a biological pest control technology for agricultural soils. Primary Energy Demand (PED) and Global Warming Potential (GWP) equivalent emissions were calculated for two valorization pathways: (i) feed for cattle (BaU); and (ii) biosolarization. In order to make these two valorization pathways comparable three hypothetical scenarios were constructed whereby each part of the system was satisfied, i.e. a pest management system and cattle feed system. The scenarios were (1) tomato pomace used for cattle feed and soil pest control using fumigant Telone II and herbicide glyphosate; (2) tomato pomace used for cattle feed and soil pest control using solarization; (3) alternative cattle feed (cottonseed, canola pellets and wheat straw) and soil pest control using biosolarization with tomato pomace. Scenario 2 and 3 resulted in a reduction of GWP and PED compared to the business-as-usual option (scenario 1). Among scenarios, the GWP impact ranged from 64-99 kg CO₂-e and 1505-2261 MJ for PED per tonne of pomace. The collocation of tomato processing facilities and farms that can utilise biosolarization in lieu of fumigation and herbicide application. Such an approach would further reduce carbon and energy impacts and facilitate the development of a circular bioeconomy in California. The majority of the impacts were shown to be outside of the tomato processors' immediate control, therefore encouraging the diversion of tomato pomace to biosolarization may be desirable. For California, the GWP and PED saving could be as large as 5.5 million kg CO₂-e and 173,000 GJ annually.

POSTER PAPERS

CASE STUDIES: DIET

16. Case studies: diet

P93. Canteen or Private Kitchen – Which Lunch is more Sustainable?

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Objective of the work: Due to the pace of modern life people eat out more often than preparing meals at home. The European food sector accounts for 17 % of greenhouse gas emissions and 26 % of natural resource use in the final consumption, but there is a lack of data differentiating the ecological impact of meals consumed in canteens and at home. Up to now only few suggestions have been made concerning sustainable food preparation in canteens, mostly without analysing primary data. Thus, we posed the question: Is it more sustainable to eat at home than in a canteen? This study is based on the first results from the German NAHGast¹ project, which aims at supporting and promoting transformation processes for more sustainability in the food sector.

Methodological details: A meal of spaghetti Bolognese prepared in a canteen kitchen (assuming 1000 meals per day²) and in a private kitchen (four people) is regarded. A life cycle assessment to calculate the material footprint per person and meal considering the ingredients, preparation, cooling and food waste was conducted. The material footprint was chosen as an indicator for the ecological impact, as abiotic and biotic resource use are major factors that affect the impact of meals on the environment. The data was derived from a case study within a hospital canteen in Germany, but as the study will continue more case studies (n=5) will be available.

Main Results: The first results indicate that the preparation of lunch within canteens is more resource efficient. For instance, the material footprint for a meal in a canteen is about 2.8 kg including food waste, whereas a meal at home has a resource use of about 3.6 kg per portion. The main reason for this is the use of gas-fired equipment instead of electric stoves in the canteen. However, even though the actual preparation is done using more energy efficient equipment, additional preparation steps (e.g. chilling, keeping warm of the meals) partly diminish this advantage.

¹ NAHGast aims at developing, testing and the distribution of concepts for sustainable production and consumption in the field of out-of-home catering.

² Within the analysed setting typically 800-1000 meals per day are served.

Implications and conclusion: The first results show that canteens are an essential part of a sustainable development in the food sector. However, the impact of e.g. how the food is processed and food waste should not be underestimated. As only the data of one case study was available, further research and data is needed to evaluate the advantages of canteen meals in more detail.

Table 1 – Comparison of material intensity

Canteen		Private household	
Steps	Material Footprint per step per meal	Steps	Material Footprint per step meal
Ingredients	1.86 kg	Ingredients	1.86 kg
Meat is stored in cooling chamber until needed	<0.01 kg	Meat is stored in refrigerator until needed	0.01 kg
1. Spaghetti are cooked in cooking kettles using gas	0.02 kg	1. Spaghetti are cooked in a pot on an electric stove	0.36 kg
2. Spaghetti are put in a chiller to cool	0.01 kg	2. Sauce is prepared in a second pot on an electric stove using all ingredients	1.40 kg
3. Cooked spaghetti are stored in a cooling chamber for one day	0.07 kg		
4. Spaghetti are put in combi-steamers (using gas) to heat up	<0.01 kg		
5. Spaghetti are ready to serve and are kept warm	0.05 kg		
6. Sauce is prepared using pressure kettles (using gas)	0.07 kg		
7. Sauce is kept warm and ready to serve	0.16 kg		
food waste about 25 %	(additional) 0.56 kg	food waste 0 -12 %	(additional) 0 - 0.46 kg
result	2.79 kg		3.63 - 4.06 kg

P94. Climate Impacts of European Diets – Special Focus on Data Harmonization and Uncertainties

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The primary goal of our study (part of SusDiet-project: SUSFOOD-ERA-Net, 2014-2017) was to produce greenhouse gas emission -estimates (GHGE-estimates) for average diets in Finland, Sweden, UK, France, and Italy. The overall objective was to compare foods (including drinks) in the diets and to find more climate friendly diet options.

We used categorization of 151 food categories and gave GHGE-estimates for each food category. Altogether, we used 80 indicator products to represent the 151 food categories. GHGE-estimates of food product categories were based on statistics and existing literature (LCA-based studies) where we included agricultural production - and processing steps. We went through numerous studies (mostly scientific, peer reviewed articles) and chose literature-references with similar methodologies. Furthermore, we used literature- and expert evaluation -based GHGE-defaults for cooking, storing and packing, and left transportation and consumers' travels to grocery stores outside our system boundaries. Additionally, since the used dietary data was based on food consumed (daily food consumption data reported by participating consumers), we used conversion factors to convert the GHGE-estimates of produced food to match the weight of food consumed. As uncertainties of the GHGE-estimates were high, uncertainty ranges for the GHGE-estimates of the 151 food categories were also produced.

The GHGEs of the average diets in the five countries varied from 4.1 to 5.0 kg CO₂-eq. per person per day but no significant differences were observed (Figure 1). Food categories that contributed the most to the GHGEs of the average diets were: several "Composite food"-categories that contain meat, "Beef", "Cheese", "Coffee", "Milk" and "Poultry". When the 151 food categories were aggregated into 18 food categories the aggregated categories contributing the most to the GHGEs of the average diets were "Meat and meat products" and "Composite food" (Figure 2).

The GHGE-estimates of our study are to be further used in optimization models where GHGE-emissions and nutritional quality of diets are jointly assessed to explore and compare the potential of substitutions between food categories.

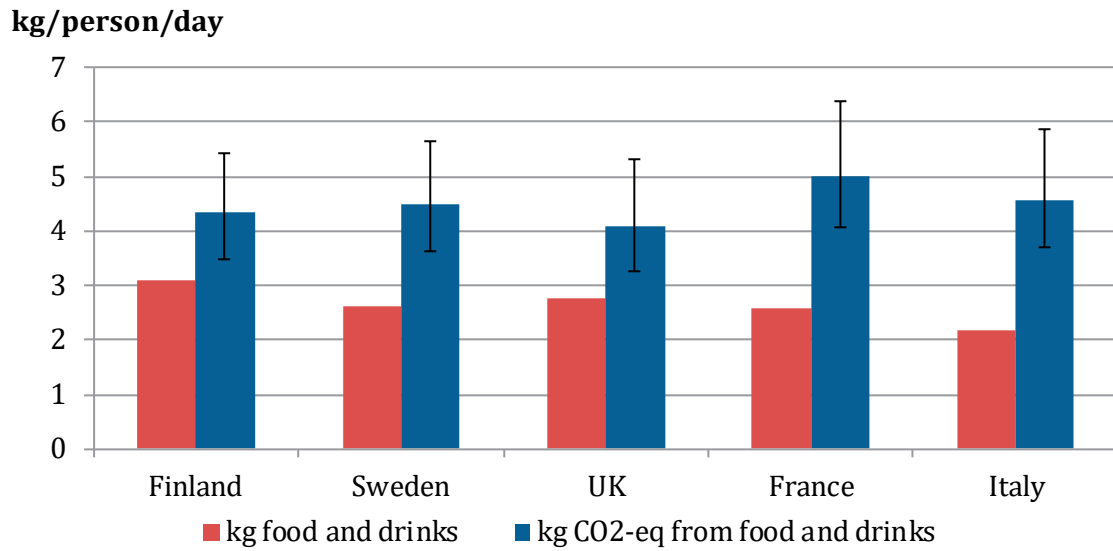


Figure 1. Food consumption amounts and GHGEs (kg CO₂-eq) of the food consumed, and uncertainty ranges of GHGEs in Finland, Sweden, UK, France, and Italy.

**kg CO₂-eq./
person/day**

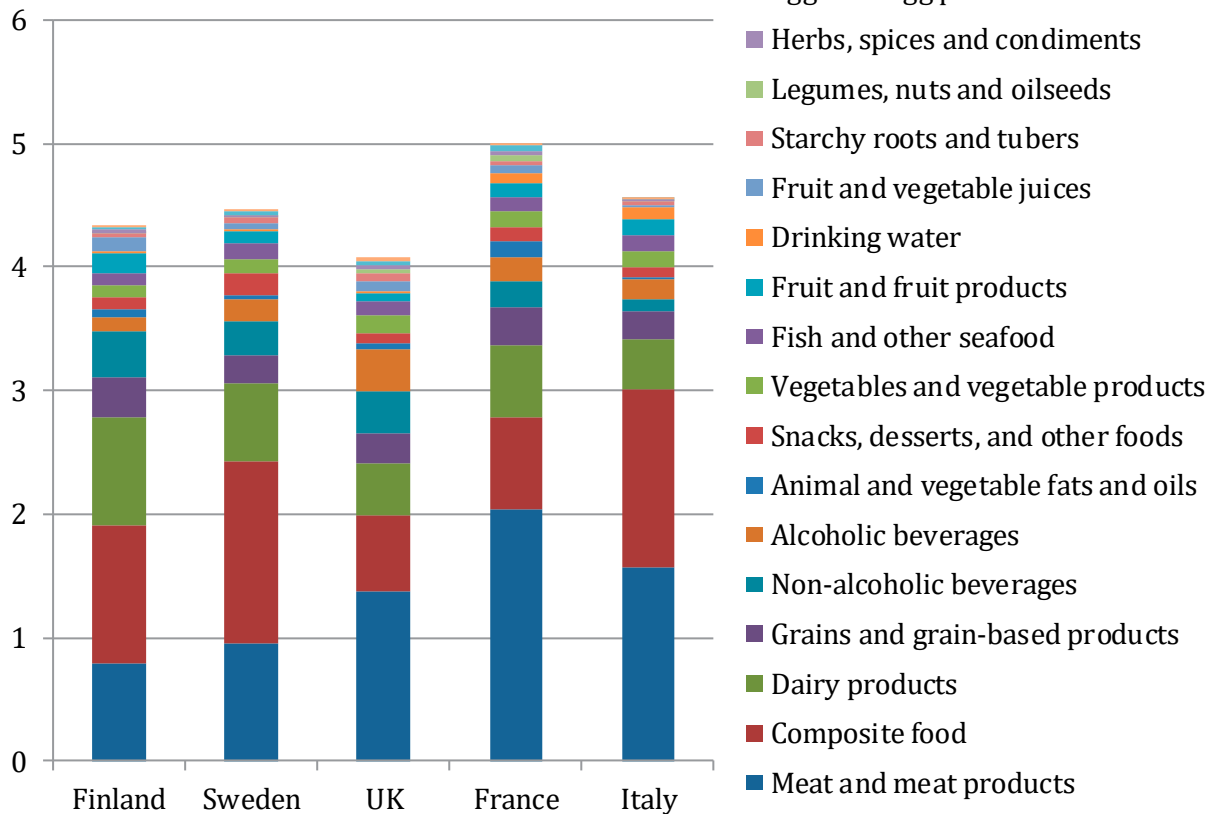


Figure 2. GHGEs (kg CO₂-eq) of the average diets in Finland, Sweden, UK, France, and Italy divided into 18 aggregated food categories.

P95. Considering cascade effects in freshwater consumption impacts: the case of dietary patterns of two Mediterranean countries

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The type of diet consumed in a given region (how foods are combined) influences environmental impacts considering how and where foods are produced. The MEDINA project focuses on the durability of national diets of 2 countries: France and Tunisia.

Considering environmental impacts LCA offers a suitable global methodology provided that freshwater deprivation and land use impacts are considered. This study focuses on freshwater consumption and the first objective is to compare indicators already proposed by the scientific community for freshwater deprivation midpoints. The second objective is to discuss the importance of cascade effects i.e. considering deprivation effects in the downstream parts of a given watershed due to freshwater consumed in the upstream part of it. Cascade effects were highlighted by Loubet et al. (2013) who propose a method to compute them.

Amounts and origin of raw products were computed from national food enquiries managed by the Medina-study group coupled with world trade data (UNComtrade database) and a first evaluation of freshwater deprivation impact was computed at the country level. Then mondial vectorial maps were built for 3 indicators at the watershed level for the level 6 in Pfafstetter classification (Verdin & Verdin, 1999). The 3 indicators are: the water scarcity indicator (Pfister et al., 2009), the consumption-to-availability ratio defined by the Water Footprint Network (Hoekstra et al., 2012) and the AWARE (Available Water REmaining) indicator proposed by the UNEP-SETAC WULCA working group. Indicators were re-computed to fit with Pfafstetter watersheds and cascade effects was then applied.

Maps are given for annual and monthly values of the indicators (13 maps per indicator). Freshwater consumption impacts were computed for French and Tunisian diets with the 3 indicators (example in fig.1). Global impact of freshwater withdrawals due to food consumption of one Tunisian during one year is in average 7 times higher than for a French. Importance of considering cascade effect (example in Fig.2) is discussed at the country level for the two national diets. A spatial focus is made on products grown in specific regions considering their importance in the freshwater consumption impacts in order to analyse the results at the watershed level. Considering monthly rather than annual values is also discussed.

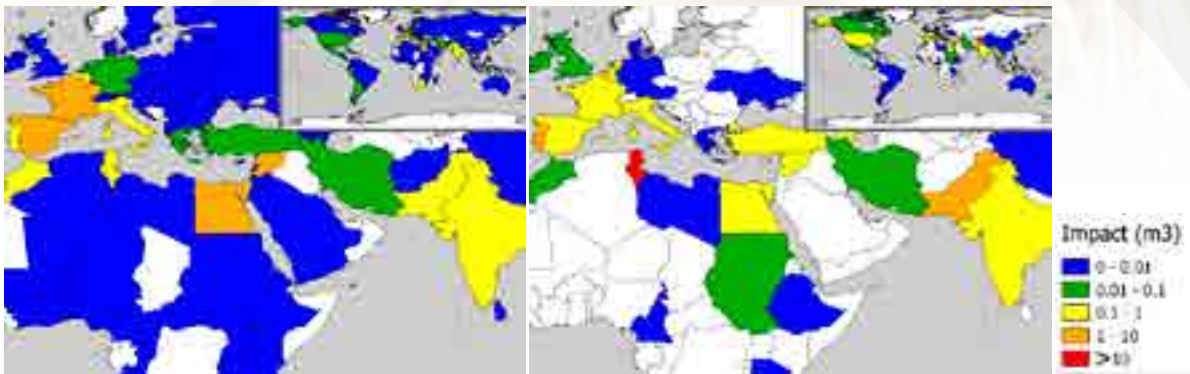


Figure 1. Freshwater impacts of French (left) and Tunisian (right) diets all around the world (using WSI). Amounts given in equivalent m^3 for food consumption of one person during one year.

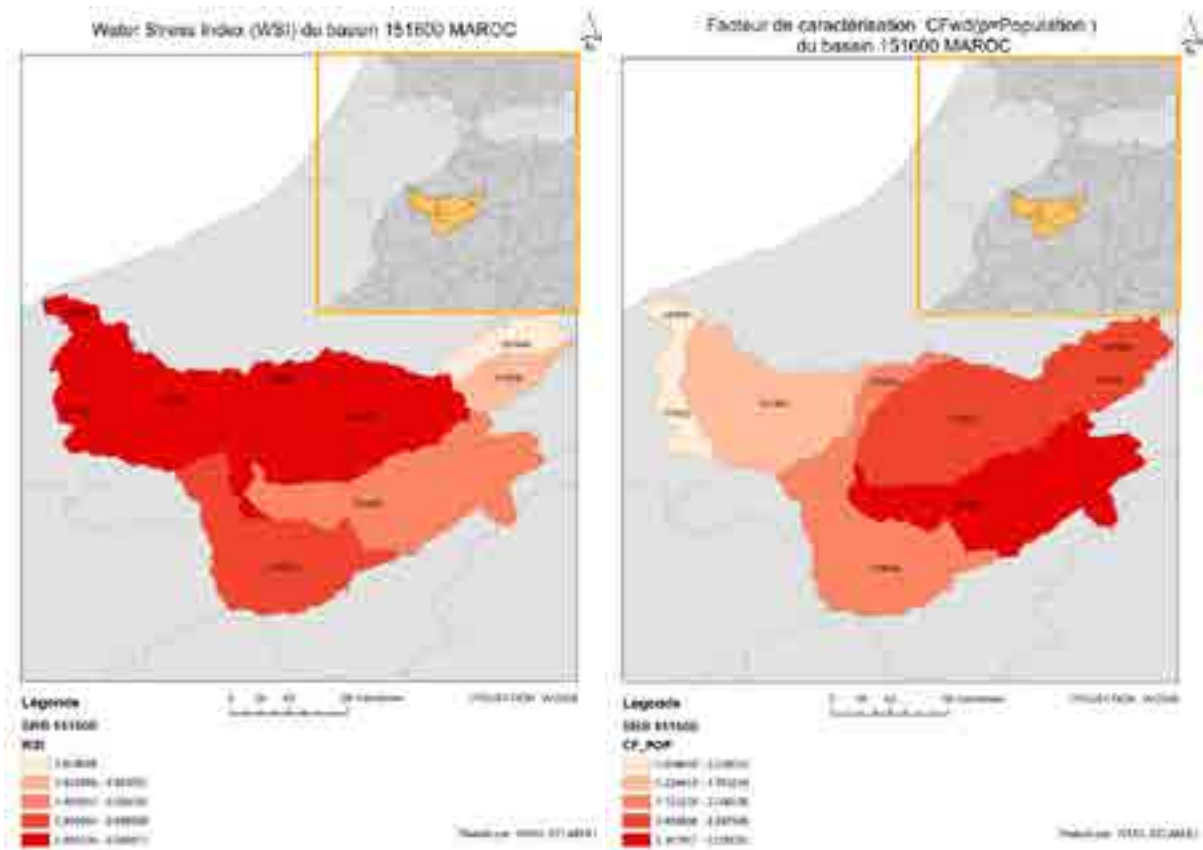


Figure 2. Midpoint indicator (WSI) on a Moroccan watershed with (right) and without (left) cascade effect

POSTER PAPERS

IMPACTS

17. Impacts

P96. Cow manure management: closing the loop with anaerobic digestion and crop rotation

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Abstract: Agriculture has traditionally been an example of closing loops, being manure management systems necessary to close livestock production on sustainable criteria. The objective of this study was the environmental assessment of using digested manure in maize production. The system included manure anaerobic digestion (AD) and maize-raygrass crop rotation fertilized with digestate. Raygrass was cultivated after maize harvest to reduce nitrogen excess in soil, and next was used as co-substrate in AD. In this LCA, the functional unit was 1 ton of maize. The system boundaries excluded livestock production and included the processes: the manure AD, the liquid-solid fraction mechanical separation of the digestate (LF and SL), the maize-raygrass rotation, and transports (Fig 1). For the by-products of biogas plant we used the substitution method, with electricity country mix and chemical fertilizers. Crop emissions were calculated adapting international methodologies (IPCC, EMEP/EAA and SALCA) to pedoclimatic local conditions. The LCIA method was ILCD 2011 Midpoint. Results showed the benefits of using digestate LF, due to renewable electricity production in manure processing. Direct emissions because of fertilizer application were the major contributor to impact categories (Fig 2). Diesel used in labour operations was an important contributor in climate change and ozone depletion. If livestock production and allocation of manure processing between livestock and maize production would have been included in the system boundaries, then the digestate contribution in maize production might decrease. The study demonstrated the importance of emission accounting. Therefore, more specific methods to calculate emissions for agricultural systems are necessary.

P97. Gas Emissions from Manure Management: Measurements vs Estimates.

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Data on GHGs emissions obtained through experimental results conducted in Italian dairy farms were compared with emissions from manure management estimated using the IPCC *Tier 2*.

CH₄ and N₂O concentrations were seasonally measured in three dairy cattle buildings, characterized by different manure management systems, using a photoacoustic gas monitor according to the “dynamic chamber method”. Measurements were acquired during two years from different shed components. Meanwhile, the farm owners filled a questionnaire useful for the IPCC *Tier 2* implementation.

The IPCC approach does not distinguish between emissions originating from the building and from the storage. Differently, field studies are more frequently focused on a particular aspect of the whole manure management system, giving partial emission factors. For CH₄, the choice of the proper Methane Conversion Factor (MCF) is crucial for the representativeness of the final result, due to their broad variation also within the same climatic zone. However, MCFs cannot reflect the variety of possible solutions for manure treatment and are grouped in generic categories poorly defined. In the case of direct N₂O emissions, the IPCC equation reflects the amount of N excreted by the animal category corrected for an emission factor (named EF₃). The EF₃ is equal to zero for uncovered anaerobic lagoons, but our data do not support this assumption.

Experimental results underline the relevance of the manure removal strategy in the determination of emissions and highlight the need of more precise and flexible emission factors in order to make estimations closer to the actual level of emissions.

Table 1. Farm characteristics and key parameters used in the IPCC equations.

Farm	Cows	Manure removal strategy	Manure management system selected from IPCC	MCF (% of B ₀)	EF ₃ $\left(\frac{kg\ N_2O - N}{kg\ N_{excreted}}\right)$
1	110	Concrete floor (covered with rubber) with scraper	Liquid slurry with natural crust	27	0.005
2	300	Flushing system	Uncovered anaerobic lagoon	74	0.000
3	450	Perforated floor with pit storage below animal confinement	Pit storage below animal confinement	3 (< 1 month) 27 (> 1 month)	0.002

MCF is the Methane Conversion Factor for the selected manure management system.

EF₃ is the emission factor for direct N₂O emissions from the selected manure management system.

Table 2. Comparison between housing emission factors (measured) and overall emission factors (estimated with IPCC Tier 2 method). Data are expressed as kg of gas·head⁻¹·year⁻¹.

Farm	CH ₄			N ₂ O		
	M (min-max)	E	%	M (min-max)	E	%
1	5.84 (0.31-13.11)	33.60	17.38	0.187 (0.026-0.329)	1.131	16.53
2	1.11 (0.14-3.93)	153.36	0.72	0.024 (0.004-0.091)	0.000	-
3	2.17 (0.59-5.17)	* 5.43 ** 48.88	* 39.92 ** 4.44	0.051 (0.015-0.107)	0.709	7.18

M: median of measured emission factors from the building.

E: IPCC Tier 2 estimations, including housing and storage emissions. They refers to the CH₄ originating from the manure management and to the direct emissions of N₂O.

%. Contribution of housing to the overall emissions (expressed as % of IPCC estimation).

* calculated considering storage duration < 1 month.

** calculated considering storage duration > 1 month.

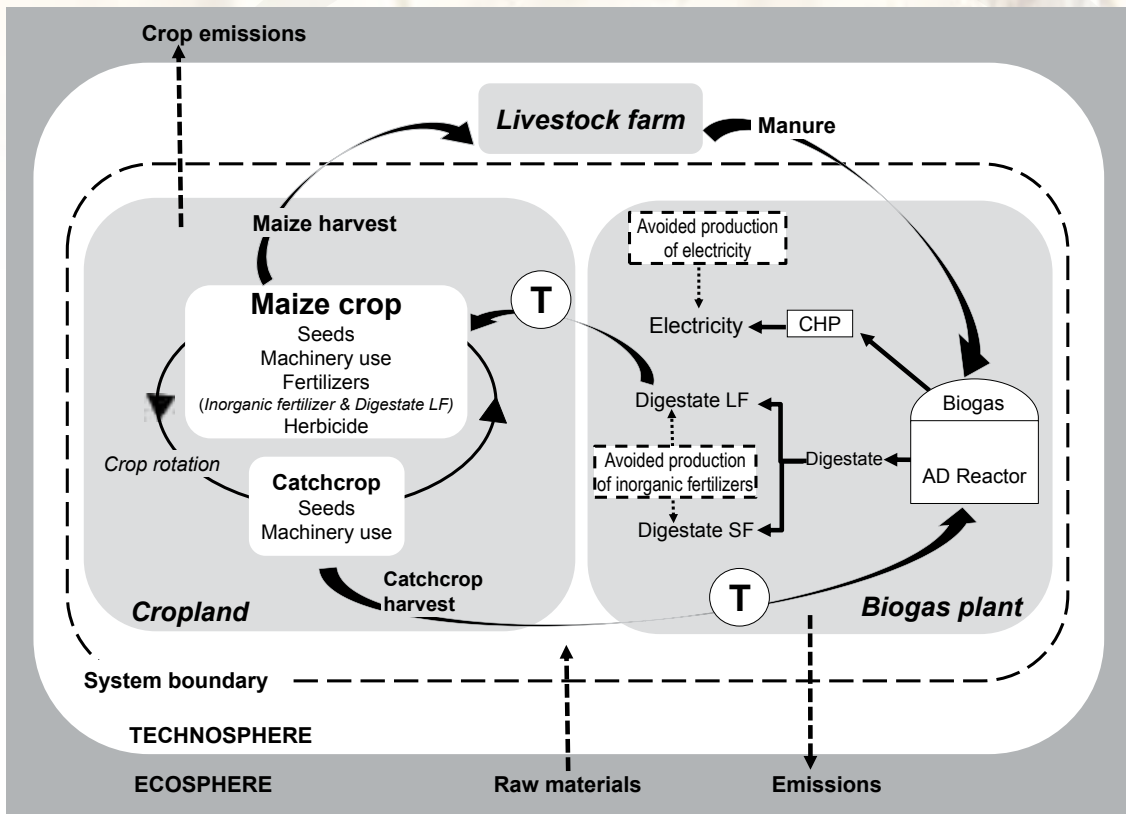


Fig 1. System flow diagram for the production of 1 ton of maize.

Crop emissions: emissions due to organic and inorganic fertilizer application in maize crop; *Seeds*: seed production and transport; *Machinery use*: machinery and diesel production, and emissions of machinery use. T: transport

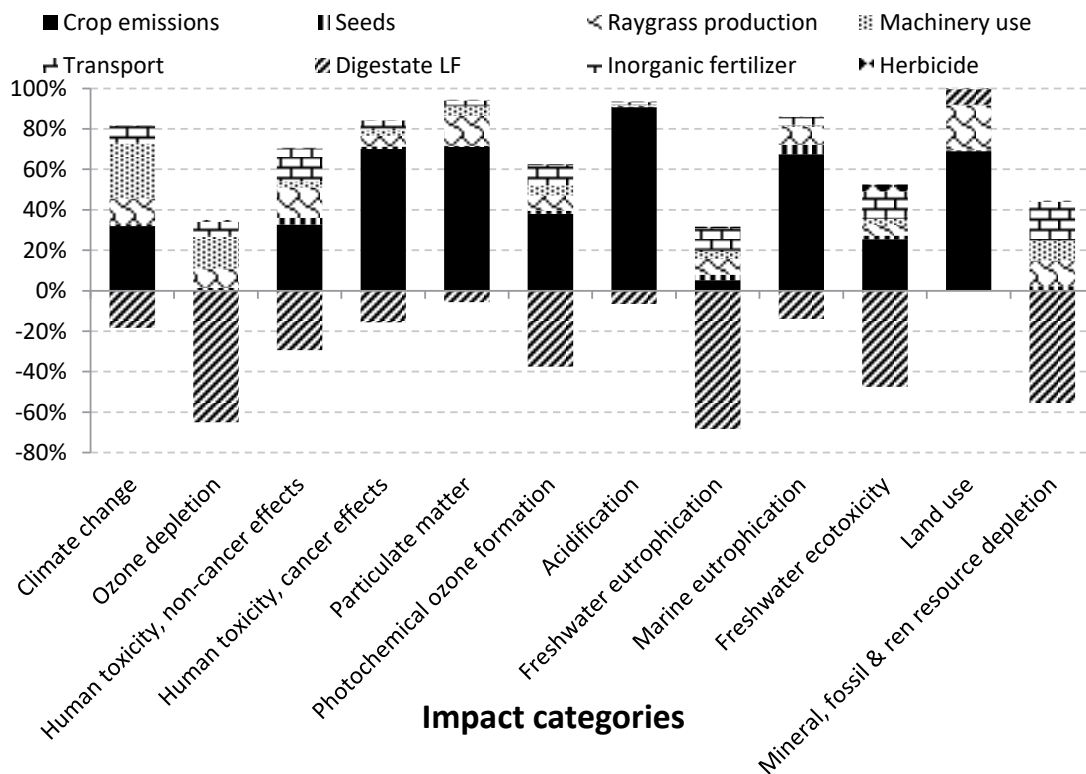


Fig 2. Processes contribution to impact categories for 1 ton of maize production (%)

P98. Environmental impact generation in the cradle-to-farm gate link: analysis of the importance of the on-farm versus off-farm stage

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Most LCAs carried out in the agricultural field focus on the entire cradle-to-farm gate link of the agri-food chain, without specifically distinguishing between on- and off-farm environmental impacts. Although this allows for a comprehensive assessment of the overall environmental impacts, a precise knowledge on their distribution between the on-farm and off-farm (upstream) stages is necessary to better understand the role of these two stages in the agricultural environmental impact generation. The aim of our work is to determine for Swiss dairy farming in the alpine area the relative importance of these two stages in the overall environmental impact.

Our work relies on the dataset made of a pooled sample of 56 Swiss dairy farms of the alpine area (see Table 1), described in detail in Jan et al. (2012). For each farm, the environmental impacts generated from the cradle to the farm gate have been estimated using the SALCA (Swiss Agricultural Life Cycle Assessment) approach. We decomposed these impacts into their off-farm (upstream) and on-farm parts. In a subsequent step, the proportion of impacts generated on- versus off-farm is analysed specifically for each environmental impact category.

The average and median share of the impacts generated on- versus off-farm, as well as the variability of the on-farm proportion, vary according to the impact category considered (Fig. 1). More than half of the overall average impact generation in the cradle-to-farm gate link takes place on-farm for the environmental impact categories terrestrial eutrophication, acidification, ozone formation, land competition, global warming potential, eutrophication aquatic P and eutrophication aquatic N. For the impact categories water deprivation, aquatic ecotoxicity, terrestrial ecotoxicity and human toxicity, the off-farm share predominates on average and varies between 50% and 80% depending on the impact category considered.

Our work shows that the importance of the on- versus off-farm stage varies according to the environmental impact category considered. In that sense, it highlights for each category where (i.e. off-/on-farm) the focus should be primarily placed if we wish to reduce the environmental impact generation. This information will be especially precious when analysing the compliance of a farm with the carrying capacity of its local ecosystem.

Reference

JAN, P., DUX, D., LIPS, M., ALIG, M. & DUMONDEL, M., 2012. On the link between economic and environmental performance of Swiss dairy farms of the alpine area. *The International Journal of Life Cycle Assessment*, 17, 706-719.

Table 1: Range and mean value of farm characteristics of the 56 sample farms, based on farm accountancy data

Farm intensity and scale	Minimum	Maximum	Average
Farm livestock units (LU)	9.81	50.50	24.97
Total milk production in kg	30'265	243'587	111'624
Milk yield per cow in kg	2'858	12'167	6'027
Farm usable agricultural area (UAA) in ha	7.98	40.60	22.49
Stocking rate (LU/UAA)	0.45	2.00	1.18

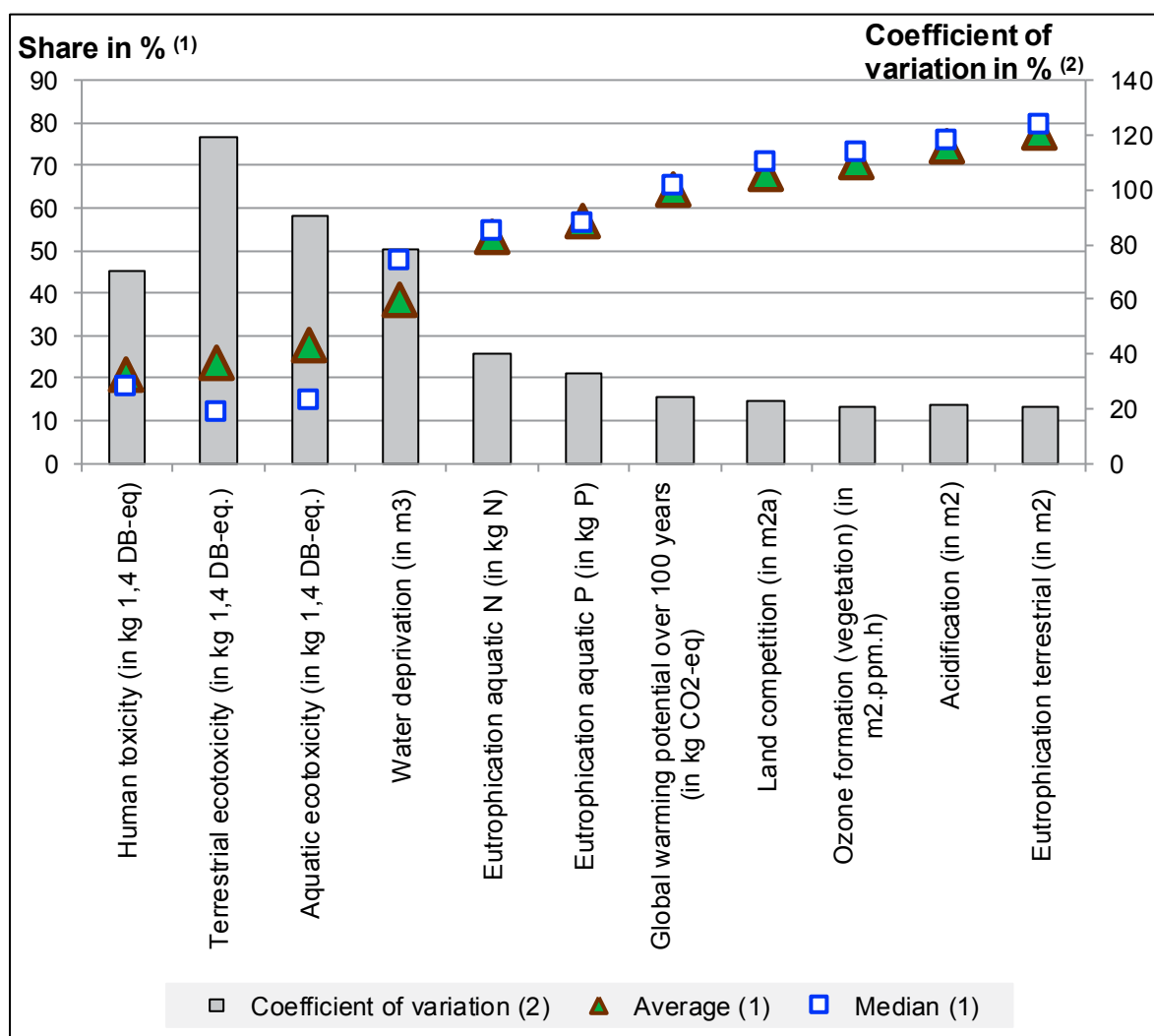


Figure 1: On-farm impact share for different impact categories listed from left to right in ascending order of average on-farm share

P99. Unhealthy diet – unhealthy climate

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Food consumption strongly influences the climatic impact of the food system. Therefore, individual food choices have the potential to substantially influence the quantity of greenhouse gas emissions (GHGE) released to the atmosphere. The objective of this study was to evaluate GHGE associated with the food consumption of Irish adults. It aimed to investigate if there are distinct and different patterns in how dietary emissions are generated and if adherence to dietary guidelines and motivations regarding food choice varied between such patterns. The National Adult Nutrition Survey (NANS) provided a representative database on habitual food and beverage consumption for Irish adults in addition to socio-demographic, food motivation and lifestyle measures. Emission factors that included emissions associated with food production, packing, distribution, storage, transportation, food preparation, and consumer waste were drawn from the literature and assigned to food groups. Individuals who adhered to specific World Health Organization dietary recommendation were considered to achieve dietary guidelines. K-means cluster analysis categorised individuals together based on similarities in how GHGE were generated. For data analyses energy misreporters were omitted and each gender assessed separately. Mean GHGE derived from daily dietary intakes was estimated as 6.5 kgCO₂eq per person. Cluster analysis identified three meaningful gender specific segments labelled 'Traditional', 'Western' and 'Prudent' (Table 1). The Western segment was younger and had the highest carbon footprint of the three groups; attaining significantly higher GHGE from alcohol and processed meat. Traditional and Prudent clusters did not differ significantly in overall emissions but diverged in how they generated dietary emissions. Traditional clusters gained significantly higher GHGE from red meat. Conversely, Prudent clusters attained significantly lower emissions from red meat but higher GHGE from fruit, vegetables, and fish. Red meat consumption was therefore not found to be a decisive factor in determining overall GHGE. Nevertheless, Prudent clusters adhered to more dietary guidelines than the other groups and were motivated primarily by health and nutrition. Food neophobia was high in Western clusters and they were motivated by price and taste. Hence, negative healthy eating motivations may not only lead to undesirable food choices but also to higher dietary emissions. Results suggest that

policy instruments adopted for health and sustainability should be holistic in nature rather than focusing on one food group alone.

Table 1. Mean daily GHGE (gCO₂eq) and percentage contribution of each food group towards total daily GHGE (kg CO₂eq) of each of the gender specific clusters.

	Traditional (51%)			Western (24%)			Prudent (25%)		
	\bar{x}	SD	%	\bar{x}	SD	%	\bar{x}	SD	%
Male (n=470)									
Starchy staples	764 ^a	222	11	657 ^b	177	8	817 ^a	274	11
Dairy	877 ^a	572	12	607 ^b	411	7	925 ^a	503	12
Vegetables	56 ^a	56	1	36 ^b	57	1	111 ^c	127	2
Fruit	54 ^a	57	1	28 ^b	50	0	155 ^c	108	2
Legumes, pulses, nuts	40 ^a	45	1	40 ^a	49	1	82 ^b	75	1
Red meat	2263 ^a	1774	28	2131 ^{ab}	1793	21	1603 ^b	1460	19
Eggs, poultry, pork	680	383	10	707	390	8	735	473	10
Fish	212 ^a	305	3	96 ^b	234	1	486 ^c	529	7
Processed meat	327 ^a	369	5	685 ^b	534	8	248 ^a	304	3
Savoury snacks	142 ^a	215	2	606 ^b	584	7	188 ^a	246	3
High sugarsnacks	306 ^a	216	5	168 ^b	182	2	385 ^c	226	5
Fats, oils	405 ^a	419	6	240 ^b	238	3	316 ^{ab}	329	4
Carbonated beverages	165 ^a	288	2	489 ^b	499	6	180 ^a	285	2
Other beverages	306 ^a	250	4	500 ^b	352	6	642 ^c	461	9
Alcohol	605 ^a	827	8	1749 ^b	1667	18	328 ^c	501	4
Miscellaneous	174 ^a	173	3	300 ^b	250	3	464 ^c	403	6
Total (kg CO₂eq)	7.4^a	2.1	100	9.0^b	2.7	100	7.7^a	2.0	100
	Traditional (45%)			Western (27%)			Prudent (28%)		
	\bar{x}	SD	%	\bar{x}	SD	%	\bar{x}	SD	%
Female (n=490)									
Starchy staples	572 ^a	166	12	501 ^b	165	9	547 ^{ab}	186	11
Dairy	716 ^a	415	15	461 ^b	281	9	706 ^a	403	14
Vegetables	73 ^a	60	2	44 ^b	49	1	116 ^c	84	2
Fruit	85 ^a	78	2	33 ^b	42	1	106 ^a	90	2
Legumes, pulses, nuts	32 ^a	32	1	35 ^a	38	1	87 ^b	83	2
Red meat	1432 ^a	1259	25	1336 ^a	1263	20	829 ^b	931	14
Eggs, poultry, pork	540	335	11	459	327	8	373	256	7
Fish	143 ^a	174	3	108 ^a	176	2	549 ^b	359	11
Processed meat	162 ^a	206	3	443 ^b	393	8	84 ^c	158	2
Savoury snacks	108 ^a	144	2	369 ^b	337	7	108 ^a	149	2
High sugarsnacks	247 ^a	144	5	183 ^b	140	3	296 ^c	204	6
Fats, oils	300 ^a	300	6	151 ^b	180	3	156 ^b	177	3
Carbonated beverages	68 ^a	163	1	494 ^b	540	8	89 ^a	175	2
Other beverages	284 ^a	206	6	402 ^b	300	7	484 ^b	303	10
Alcohol	120 ^a	198	2	560 ^b	646	9	216 ^c	287	4
Miscellaneous	179 ^a	177	4	226 ^a	242	4	351 ^b	320	7
Total (kg CO₂eq)	5.1^a	1.3	100	5.8^b	1.7	100	5.1^a	1.3	100

^{abc} indicates significant differences between clusters (p<0.05) after appropriate post-hoc tests
SD represents standard deviation

POSTER PAPERS

MEAT PRODUCTION 2

18. Meat Production (2)

P100. Regional Evaluation of Environmental Impacts of Swine Production in the USA


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Objective: To evaluate regional variation in swine production in the USA. Differences in management practices, particularly manure management, and climate as well as a population influence the resource consumption and environmental impacts associated with livestock production. We evaluate the name swine production areas in the US focusing primarily on greenhouse gas emissions, energy consumption and water use.

Method: We have coupled the Pig Production Environmental Footprint Calculator with historical weather files (10 year simulation) from the Modern-Era Retrospective Analysis for Research and Applications(MERRA) archive published by the NASA Global Modeling and Assimilation Office. This data set covers the modern era from 1979 through the present with hourly weather data in a 1/2 x 1/3° spatial resolution. Hourly resolution weather files are used in the environmental calculator to estimate dire heating and cooling requirements in order to maintain appropriate temperature for comfort and optimal performance. We used publicly available data from the National Agricultural Statistics Service and the US EPA to estimate the number of animals on each manure management system at county level. The MEERA weather files were chosen at the approximate centroid of agricultural production for each county, and assumed to be representative of weather for the county. The output of the environmental calculator includes an estimate of nitrogen content in land applied manure, and this was used as input to the DNDC model to simulate field emissions based on the same weather files, NASS data on major crops produced in the county and SSURGO soil types prominent in the region. Manure application rates were based on crop demand.

Results and Impact: Post-processing with SAS® and Power Bi® enabled presentation of the data in an interactive format such that users can drill down and evaluate the underline contributions to differences in emissions at various locations. Figures 1 and 2 present screenshots of the user



interface available for exploring the data. The ability to view, at large scale, and interactive manner the cradle-to-gate life cycle impacts of an important livestock sector should lead to an improved ability for informed policymaking in the agricultural sector.



Figure 1. Screenshot of user interface for exploring regional LCA results



Figure 2. Screenshot showing detail of annual and monthly contributions to cumulative county-level greenhouse emissions

P101. Low Emission Food Production through intelligent combination of best practices in highly integrated broiler and pig production

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ABSTRACT

Animal raising and livestock production are major players in global environmental issues. Different players along the value chain must cooperate to lever existing knowledge and move towards more sustainability- based tools to measure the overall progress. The supplementation of feed with amino acids reduces feed consumption and the nitrogen content in feed, waste treatment in a biogas plant brings methane emissions to energy production, purification of methane offers new alternatives for improved energy provision and finally, specific treatment of digested residues provides new fertilizer applications. This LCA case study combines for the first time the different aspects of nutrient management, waste management, emissions management and finally fertilizer treatment enabling new ecological and economical measures to improve the nutrient cycles especially for nitrogen compounds.

Keywords:

Amino acids, methane, life cycle analysis, nitrogen, eutrophication, acidification, fertilizer

1. Introduction

Livestock is the major player in global environmental issues. The huge demand for feed crop production shapes entire landscapes and can reduce natural habitats, causing degradation in some areas, technological improvement, but it is also a key driver of global livestock production. Growing productivity has been achieved through advanced breeding and feeding technology, and through irrigation and fertilizer technology in crop production, leading to higher yields per hectare. Intensification, the vertical integration and up scaling of production also lead to larger units and larger livestock operations.

According to FAO (Gerber 2013), it is important to set up advanced technologies such like feed strategies, manure management practices and energy use efficiency to further reduce livestock production related emissions especially based on nitrogen based compounds like NO_3 , N_2O , NO_x or NH_4 . Modern livestock production is characterized by efficient nutrient management to reduce feed consumption, waste management to reduce waste volumes and finally emission management to reduce environmental impacts. All three are followed by efficient energy use and recycling. This shall be assessed in the present LCA case study. The modelling of the scenario covering the nutrient management is based on the studies of Kebreab (Kebreab 2016). The calculations for the emissions management and waste management follow theoretical assumption of the future LEF-concept.

2. Methods

Life Cycle Assessments (LCA)

Life Cycle Assessments can be used to display and monitor the specific mitigation option of these measures, but can also help to identify hotspots and further options for improvement. A couple of studies are already in place to show the different scenarios to manage feed, waste or energy, but never before concepts have been developed to bring all the different options together to one holistic solution of a low emission livestock production. In general, life cycle assessments describe the complete fate of a product by compiling and evaluating all ecological input and the consequences for the environment during each phase in the life cycle of the product based on international standards (DIN EN ISO 14040/44:2006). For livestock production, the impact categories Global Warming Potential (GWP, expressed in kg CO₂-equivalents/kg Functional Unit), Eutrophication Potential (EP, expressed in kg PO₄-equivalents/kg Functional Unit), and Acidification Potential (AP, expressed in kg SO₂-equivalents/kg Functional Unit) are the most relevant ones.

The Low Emission Farming Concept (LEF)

The concept of the “Low Emission Farming” (LEF) as a solution from the chemical industry for the feed to food value chain offers the best practice to reduce livestock related emissions to the lowest possible level. The supplementation of feed with amino acids helps to overcome nutrient gaps and reduces feed consumption. Thus leads to a lower nitrogen content in the feed. Waste treatment in a biogas plant brings methane emissions to energy production, purification of methane offers new alternatives for improved and independent renewable energy provision by further reduction of nitrogen-based emissions.

In addition, a further specific treatment of biogas fermentation residues provides new fertilizer applications. LEF combines the different options for nutrient management, waste management, emissions management and finally fertilizer treatment (Figure 1) to show the ecological and economical improvement potentials individually and in combination, evaluated through LCA methodologies monitoring the nitrogen flows.

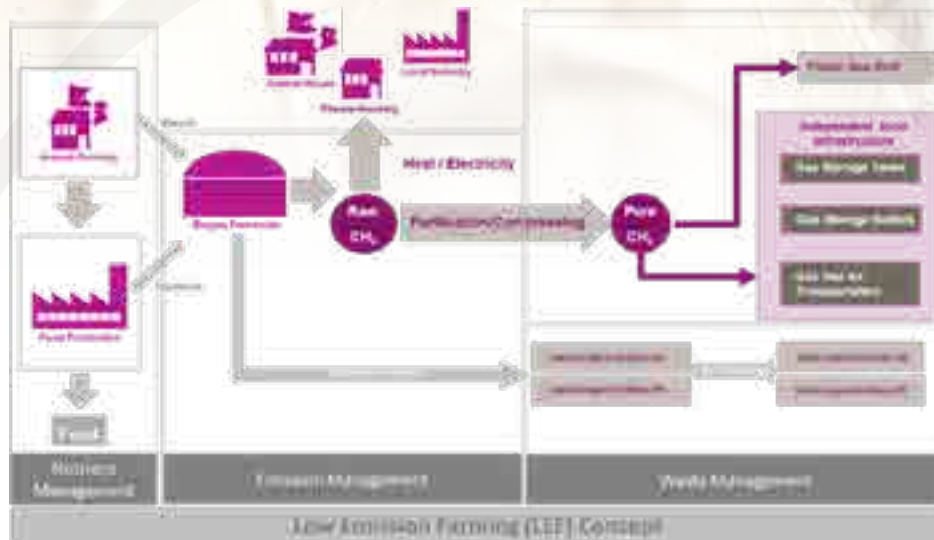


Figure 1: The concept of the Low Emission Farming, which combines Nutrient management, emission management and waste management at farm level

Nutrient management

A first step towards a more sustainable livestock production is the increase of productivity through modern feeding technologies. Improving feed efficiency and reducing the nutrient excretion enables mitigation of the overall impact of livestock production. A simple life cycle assessment (LCA) for a typical pig and broiler production scenario can demonstrate this. In a recently finalized and certified LCA study of Evonik (Haasken 2015) the very positive environmental benefit of supplementing amino acids such like MetAMINO®, Biolys®, ThreAMINO®, TrypAMINO® and ValAMINO® to pig and broiler feed could be demonstrated. By supplementing deficient diets with these amino acids, soybean meal and corn were replaced and thus, the ecological footprint was significantly improved. Kebreab (Kebreab 2016) described comparable effects for different regions and feeding regimes around the world.

Waste and Emission Management

Another technology following the efficient nutrient management is the emission or waste management, realized in the approach of the “Low Emission Farming” concept (LEF) (Binder 2015). As a solution from the chemical industry for the food production, it offers the best practice to reduce livestock related emissions to the lowest possible level. The supplementation of feed with amino acids as a first measure reduces feed consumption and the nitrogen content in feed. Further waste treatment by managing manure in a biogas plant brings methane emissions to energy production, and thus, additional improvement of emissions normally related to manure disposal. Additional purification of methane offers new alternatives for improved energy provision (own on farm use or external applications). Finally, specific further physical and chemical treatment of biogas fermentation residues provides new fertilizer opportunities allowing more nutrient specific applications in crop production. This helps to reduce the environmental impact and to comply with the more and more strict limitations for nitrogen and phosphorus fertilization of grass- and cropland.

3. Results

The results shown exemplarily in the figures 2 and 3 illustrate in terms of LCA terminology for the eutrophication mitigation potential (EP) in the European broiler and pig production at 55% and respectively 44% through implementation of the LEF concept. Due to selective addition of required amino acids to the feed mix lowers the crude protein content in the feed. Thus, less nitrogen is excreted. Integration of a biogas plant enables control and separation of emerging NH₃ and leads to smaller contributions to EP than conventional storage. Credits for energy, natural gas or diesel show only a small impact on EP as their production and incineration is more relevant for airborne emissions.

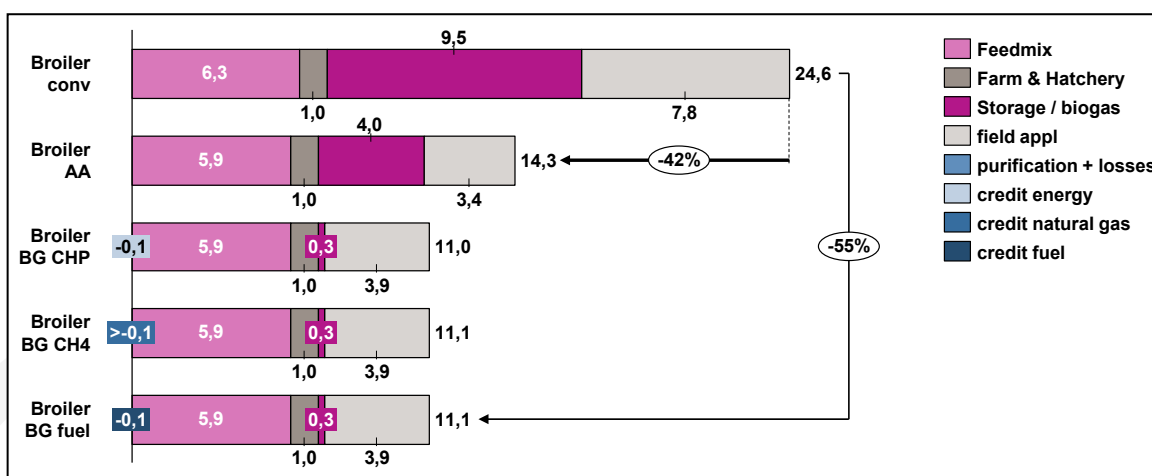


Figure 2: EP [kg PO₄e] of broiler production and stepwise integration of the LEF concept per 1,000 kg LW broiler in Europe

(Broiler cov=broiler feed without amino acids; Broiler AA= broiler feed with amino acids; Broiler BG CHP= broiler feed with amino acids and biogas production; Broiler BG CH₄= broiler feed with amino acids and biogas production and purification; Broiler BG CH₄= broiler feed with amino acids and biogas production and purification inclusively credits for fuel replacements)

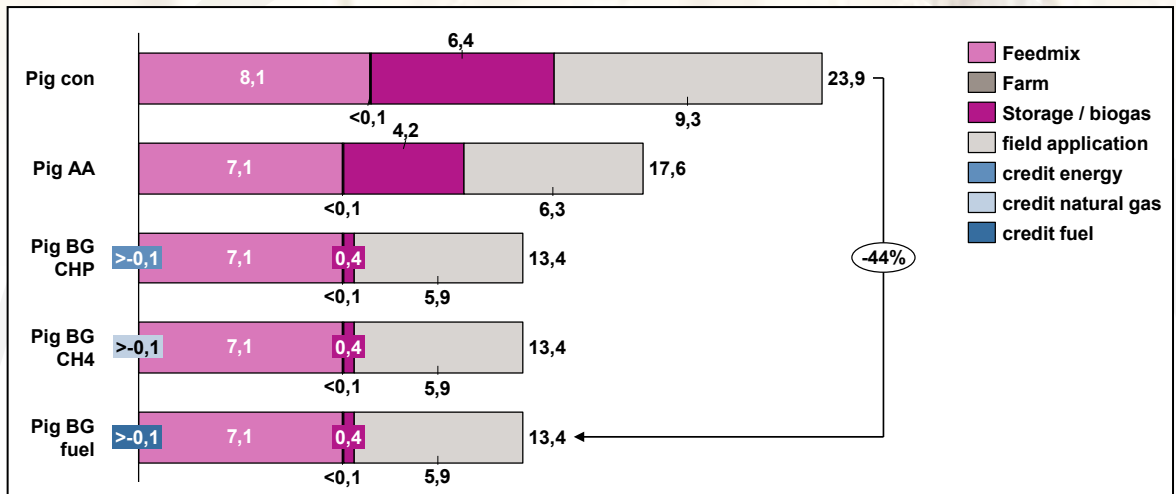


Figure 3: EP [kg PO_{4e}] of pig production and stepwise integration of the LEF concept per 1,000 kg LW pigs in Europe

(Pig cov=pig feed without amino acids; Pig AA=pig feed with amino acids; Pig BG CHP= pig feed with amino acids and biogas production; pig BG CH₄= pig feed with amino acids and biogas production and purification; pig BG CH₄= broiler feed with amino acids and biogas production and purification inclusively credits for fuel replacements)

4. Discussion

Low protein diets contribute to reduce the impact of livestock production especially on climate change in general. Since the reduction of the protein content in feed is closely linked to lower nitrogen amounts excreted, this technology significantly influences the mitigation options specifically for those impact categories of nitrogen based emissions to soil and air, as exemplarily shown for the eutrophication potential in European broiler and pig production. As for current feeding practices, there is still a major potential to mitigate this impact. There is still a considerable gap between the average content of crude protein in standard diets compared to scientifically proven low protein diets.

5. Conclusions

Additionally to an improved nutrient management further measures such like the LEF concept on a farm level are leading to significantly improved ecological performance of livestock production. These applications not only reduce the ecological impact, but also open new business opportunities for renewable energy production, energy self-provision or advanced organic fertilizer use adapted to specific recommendations as best practice with regard to sustainable agriculture.

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P102. Improving the environmental sustainability of UK poultry production: an overview of different strategies

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Some key results of recent systems modelling-based Life Cycle Assessment studies aiming to quantify and hence improve the environmental performance of the main UK poultry production systems, including broiler meat, chicken egg and turkey meat production are presented here. These results mainly originate from two large-scale projects that accessed detailed production data from the UK poultry industry.

Amongst different sub-processes, feed production and transport contributes ~70% to the global warming potential (GWP) of poultry systems, whereas manure management contributes ~40-60% to their eutrophication and acidification potentials, respectively. These impacts can be reduced by improving the feed efficiency, either by changing the birds through genetic selection or by making the feed more digestible (e.g. by using additives such as enzymes, Fig. 1). Although genetic selection has the potential to reduce the resources needed for broiler production (including feed consumption), the resulting higher demand for certain feed ingredients (e.g. high grade protein) may limit the benefits of this strategy. The use of alternative feed ingredients, such as locally grown protein crops and agricultural by-products, as a replacement of South American grown soya can potentially also lead to improvements in several environmental impact categories, as long as such feeding strategies have no negative effect on bird performance (Fig. 2).

Improving poultry housing and new strategies for manure management have also potential to further improve the environmental sustainability of the poultry industries. Using poultry litter as a fuel to generate electricity has considerable environmental benefits compared with its traditional use as a fertilizer. These benefits are mainly related to reduced emission of ammonia and nitrogen leaching and hence eutrophication and acidification. Other benefits include the reduction of non-renewable energy use, whilst the net effect on GWP is very small.

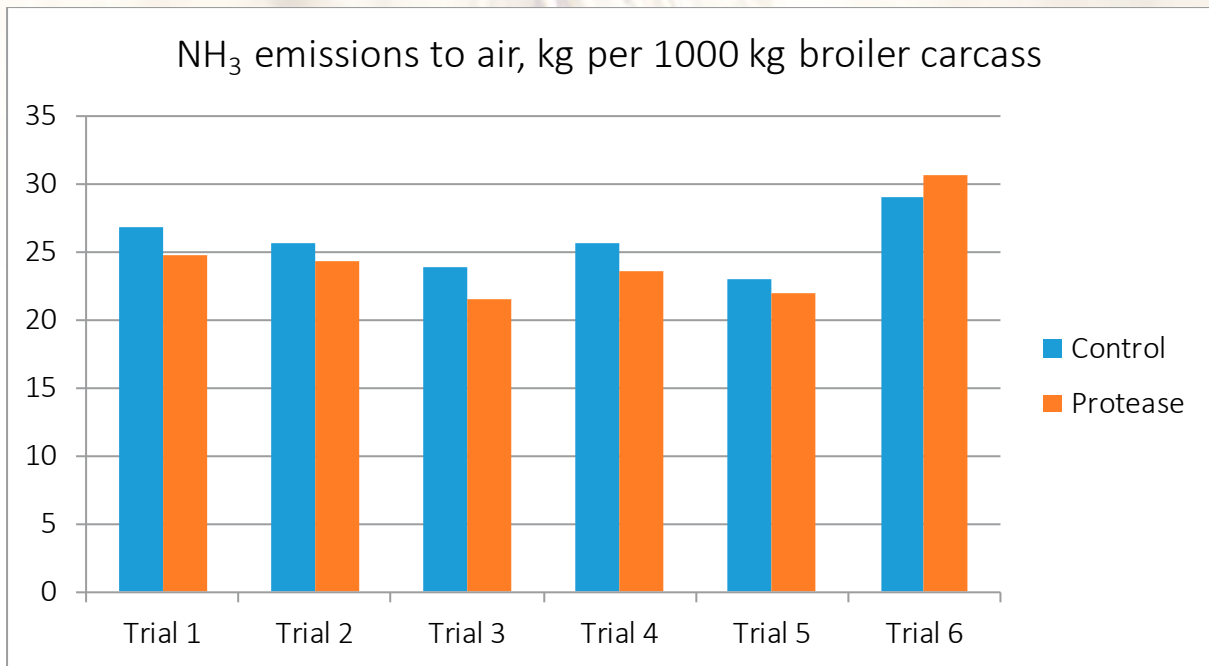
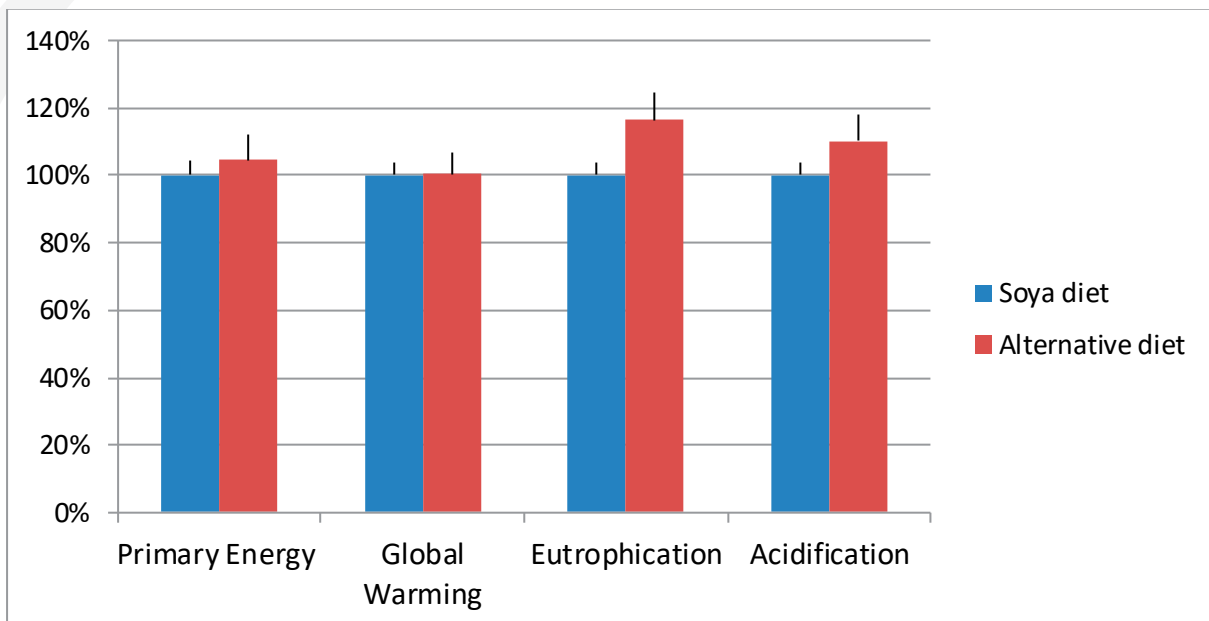


Figure 1 Emissions of ammonia (NH₃) to air with standard soya-based feed (Control) and low-protein feed with protease enzyme supplement (Protease) as modelled based on data from six different broiler feeding trials. Adapted from Leinonen and Williams (2015) *J Sci Food Agric* 95: 3041-3046.






Figure 2 The relative environmental impacts of turkey meat production when feeding the birds using an alternative diet based on European protein sources compared to the standard soya-based diet. The increase of impacts is caused by lower feed efficiency when the alternative diet was used in a feeding trial.

P103. Carbon footprint of three organic beverage products of an Italian company

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- Objective of the work

The objective of this study is to calculate the environmental impacts of the entire life cycle of 1 kg of hen eggs produced by a North Italian company considering a national scenario for the product commercialisation: in particular the research aims to quantify the carbon footprint of this product taking into account all life cycle steps, starting from the breeding of chicks to the final product distribution and consumption.

- Methodological detail

The study has been implemented in conformity to the standard ISO/TS 14067, the PCRs (Product Category Rules) 2011:15 “Hen eggs, in shell” and 2013:05 “Arable crops”. The functional unit chosen for the study was “1 kg of packaged hen eggs (category A) commercialised and consumed in Italy”. Primary data have been used and it was assumed a mean weight of 65 grams for an egg. For the analysis the Simapro software has been used with the implementation of the “IPCC 2013 GWP 100a” assessment method. Because the considered product system gives other co-product for the human consumption than “hen eggs – category A” a specific mass allocation has been considered, in line with the standard ISO 14040 requirements.

- Main Results of the study

In reference to the functional unit the result of the carbon footprint is 3.507 kgCO₂eq; this result could be split in the following way, in line with the standard requirements: 3.310 kgCO₂eq from fossil emissions, 0.183 kgCO₂eq from biogenic emissions and 0.014 kgCO₂eq from land-use change emissions. Analysing the result, the main contributions to the final result come from the primary packaging (0.953 kgCO₂eq), the breeding of chicks and pullets (0.582 kgCO₂eq), farm waste (0.453 kgCO₂eq) and the use phase (0.735 kgCO₂eq)

- Implications, meanings, conclusions

A 5% mass-based cut-off was considered for the study. Moreover, to give more consistency to the results a sensitivity analysis has been made, considering variations of the mean weight of an egg. Also, an uncertainty analysis has been implemented: these analysis confirmed the goodness of obtained result..

Figure 1. System boundaries

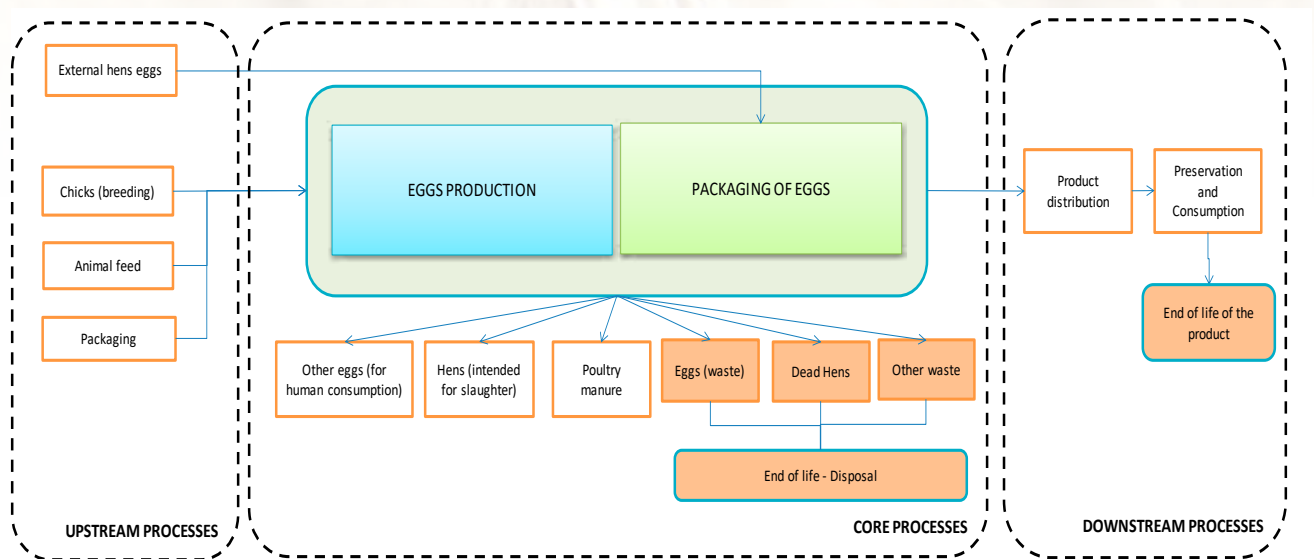


Table 1. Data for allocation

Type of product/co-product	Mass percentage
Hen eggs – category A.	72.7 %
Other eggs – category B	21.5 %
Hens for slaughter (meat)	5.8 %

Table 2 Sensitivity analysis – results

Case	Mean weight considered for an egg	Carbon footprint results
Base case	65 grams	3.507 kgCO ₂ eq
Alternative case 1	60 grams	3.691 kgCO ₂ eq
Alternative case 2	70 grams	3.348 kgCO ₂ eq

P104. Environmental Footprint Calculator for Intensive Broiler Production Processes in Greece

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An environmental footprint calculator appropriate for the intensive broiler production processes in Greece was developed. It consists of an MS excel workbook whose user is invited to provide a number of inputs concerning the animal capital grown, its nutrition, the bedding material used, transport distances and fuel and electricity consumption. As a result, ten environmental impact category indicators (EICI's) are estimated, among which the Acidification Potential (AP), the Eutrophication Potential (EP), the Cumulative Energy Demand (CED) and the Global Warming Potential (GWP₁₀₀). In this paper, the LCA methodology followed is described and the calculator is utilised for the evaluation of the environmental performance of two reference systems: a) formation of the broilers' ration within the farm (RS₁) and b) purchase of the same ration from the feed industry (RS₂). Constant ration quantity and composition throughout the broilers' growth period and constant values for the rest of the required inputs are considered between the two systems. Apart from reference provision to the user, these systems are compared regarding their environmental performance.

The calculator is based on an attributional 'cradle-to-farm-gate' LCA approach. The environmental burden is entirely assigned to the broilers' live-weight (LW). On-farm, gas emissions are estimated by using internationally accepted methods (e.g. IPCC and EMEP/EEA Tier 2 approaches). Background data are received by various databases, mostly the Agri-footprint v.1 database and economic allocation is applied in the background system's processes.

The results revealed that on-farm formation of the broilers' ration has either no or slightly positive effect on the impact categories assessed. Taking into account that compilation of Life Cycle Inventory (LCI) databases in Greece is still at an infancy stage, there is potential for future improvement of this calculator. It is believed that this tool is simple enough for the farmer to trigger the adoption of environmentally friendly practices (e.g. feed supply) in their broiler meat production processes.

Table 3 Uncertainty analysis – results

Method implemented: Montecarlo method with 1000 iterations

Data Typology	Value
Mean	3.514
Median	3.507
Standard deviation	0.165
Coefficient of variation	4.71%

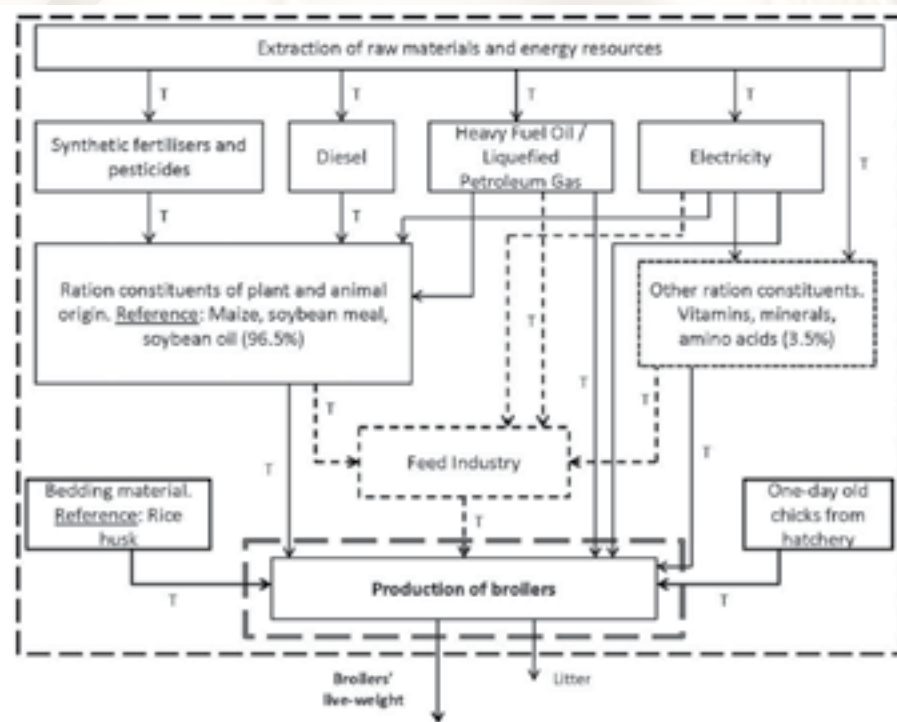


Figure 1. System boundaries considered in the calculator’s life cycle approach. Single bold long-dashed line: system boundaries, double bold long-dashed line: foreground system, single bold lines and arrows: upstream flows whose life cycle is not visible, single bold short-dashed lines and arrows: RS₂, single short-dashed line: ration constituents not considered in the calculator at the moment, T: Transport processes

Table 1. Annual LCIA results per kg broiler LW at the farm gate.

Environmental Impact Category Indicators (EICI's)	System	
	RS ₁ ^a	RS ₂ ^a
Abiotic Depletion Potential ($\cdot 10^{-6}$ kg Sb eq)	1.46	1.46
Photochemical Oxidation Potential ($\cdot 10^{-4}$ kg C ₂ H ₄ eq)	4.59	4.91
Acidification Potential (kg SO ₂ eq)	0.09	0.09
Eutrophication Potential (kg PO ₄ ³⁻ eq)	0.04	0.04
Cumulative Energy Demand (MJ)	13.69	16.02
Global Warming Potential (100 years timeframe, kg CO ₂ eq)	4.71	4.84

Human Toxicity Potential, cancer ($\cdot 10^{-8}$ CTUh)	4.63	6.67
Human Toxicity Potential, non-cancer ($\cdot 10^{-6}$ CTUh)	4.05	4.14
Freshwater Ecotoxicity Potential (CTUe)	17.13	19.64
Water Depletion Index (m^3)	0.23	0.23

^a Values were rounded to the 2nd decimal digit

POSTER PAPERS

NOVEL FOODS AND APPROACHES

19. Novel processing and approaches

P105. Hotspots in the climate impact of local food products

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¹Natural Resources Institute Finland

The aim of the paper is to present the climate impact of 23 Finnish local food products and identify hotspots of local food supply chains regarding climate impacts.

Climate impacts for 23 Finnish local food products were assessed by using carbon footprint and IPCC methods and a comparative case-study approach. Products include organic, non-organic, consumer and food service products. Primary data were used extensively.

Local products had equally often a lower and a higher impact on climate than average mainstream products (Figures 1 and 2). According to the results, any stage of the local chain can be a hotspot of climate impact. Primary production can be either a strength or a weakness for a local product. Crucial issues are soil type (organic/mineral), yield rate, and manner and efficiency of production (Figure 2). Energy use was critical in a greenhouse production and in the manufacturing phase in bakeries and in production of blackcurrant juice (storage). Utilization of seasonality was in turn critical in the case of perch, where a high catch in a limited season leads to low impact on climate during the season. These very same issues can form a solid base for climate-friendly local products if they are well managed. The fundamental challenge is caused by logistics, packaging and, in some cases, originality. The share of logistics in total climate impact can increase due to small vehicles and a low loading rate (carrot slices, rye bread), and packaging due to a need to differentiate the product (a glass bottle for blackcurrant juice, a plastic tube for honey). In addition, if “locality” is perceived as “an original product” (Karelian pasty, Savonian barley bread), ingredients can become a factor in the climate impact of local products, e.g. milk in Karelian pasty and Savonian barley bread increases the climate impact compared to “a normal bread”.

Local products are often differentiated by highlighting a closeness of the production, a distinct package, being labelled organic or being original to the area. These are also critical issues in relation to the climate impact of a local product. While other issues are universal, logistical challenges may be specific to Finland, where markets are small, leading easily to inefficient logistics. These challenges are, however, often avoided in food service by bigger

product flows, and may also be avoided in a consumer product distribution network by combining products from different suppliers.

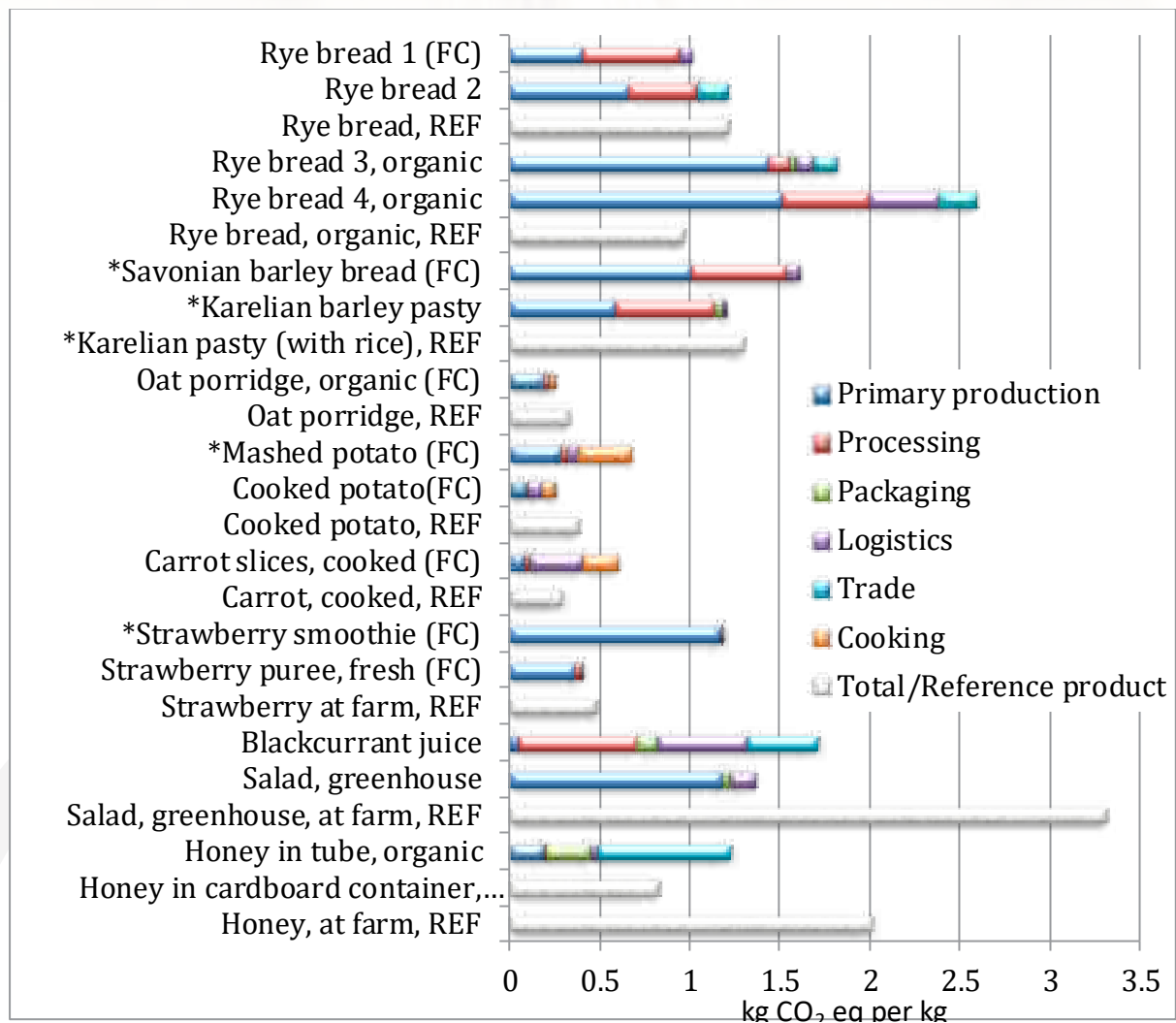


Figure 1. Climate impact for plant-based case-products and honey; some products (*) also have milk as ingredient. REF= reference product, FC=food service product.

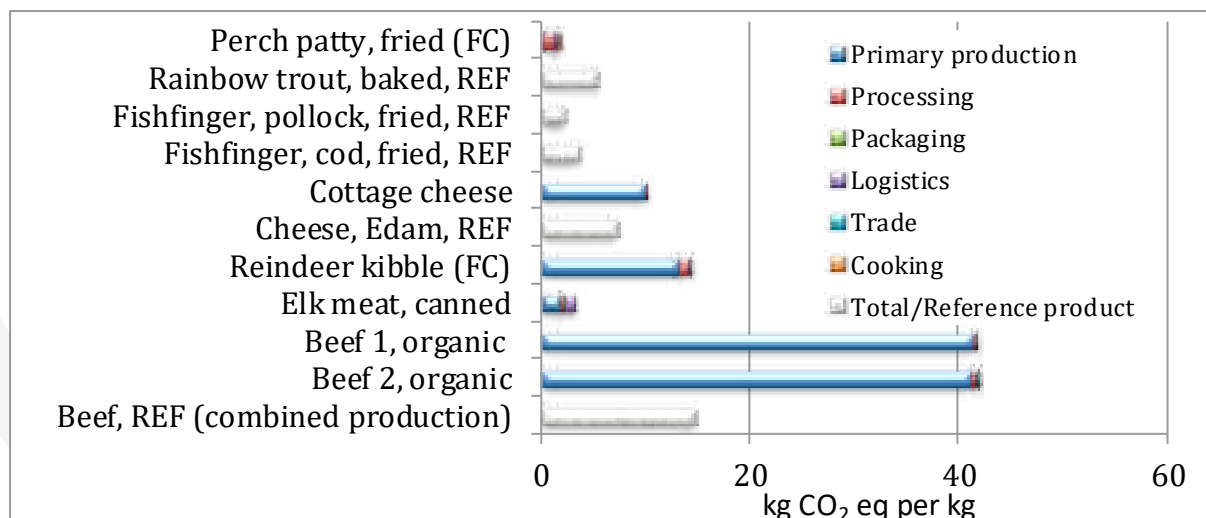


Figure 2. Climate impact for animal-based products. REF= reference product, FC=food service product.

P106. Food Redistribution in a Life-Cycle Perspective

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The 1.3 billion tons of edible food wasted annually is estimated to release 3.3 billion ton of CO₂-eqv (FAO 2013). Whereas food redistribution is traditionally seen as a way to feed the hungry, the possible environmental benefits have until recently been rather unexplored (Hanssen et al. 2015). Based on the assumption that redistributed food substitutes new food production, redistribution can be seen as an effective tool for reducing the environmental impact of the food systems. Thus far, very few studies have looked at the actual environmental benefits of redistribution activities, let alone compared this to other waste treatment measures (for notable exceptions, see Raadal, et al. 2016; Eriksson et al. 2015).

Based on a three-phase project investigating food redistribution in the Nordic countries (Hanssen et al. 2015, Gram-Hanssen et al. 2016), this study provides such initial analysis by looking at food redistribution in a life-cycle perspective. The study investigates the environmental costs of redistribution activities based on different redistribution models found in the Nordic countries and compares this data to other forms of food waste processing, such as energy recovery and biogas. A life-cycle approach guides the study and, where available, LCA data is used (SimaPro, Ecoinvent). When LCA data is unavailable, estimates are made based on in-depth knowledge of the redistribution process in the Nordic countries with the cities of Oslo and Fredrikstad, Norway as cases.

Where previous studies have shown that redistribution per ton of food waste reduces CO₂-eqv by ~1138 kg compared to biogas production (Raadal et al. 2016), the case study used indicates that savings can be more than twice as high, depending on the composition of the redistributed food. This supports the claim that redistribution can be used as a food waste reduction measure.

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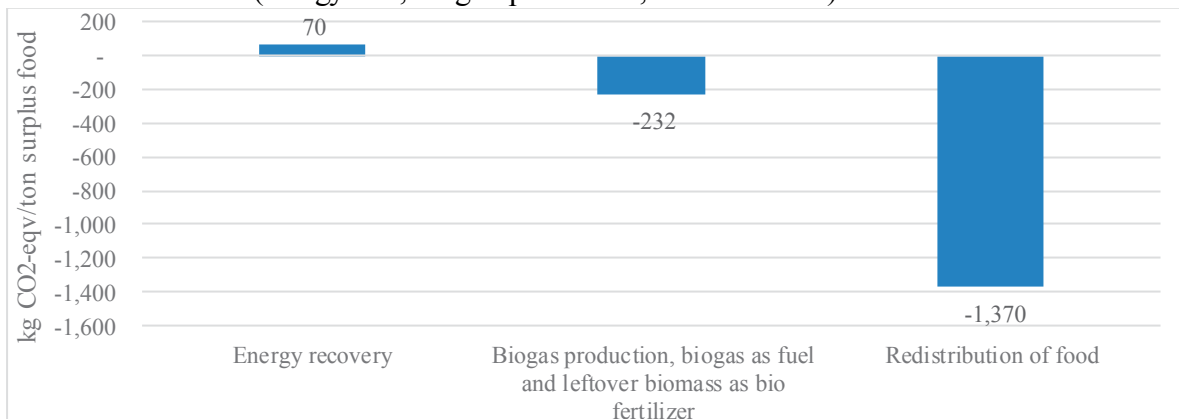


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FIGURES

Figure 1 and 2: Breakdown of the main source of surplus food for both food banks and social organisations in the Nordic countries. (Source: Gram-Hanssen et al. 2016)

Figure 3: Kg. CO₂-eqv/ton food waste released or saved during different organic waste treatment methods (energy use, biogas production, redistribution)



(Source: Raadal et al. 2016)

Figure 4: CO₂-eqv emissions connected to the various types of food redistributed in 2015 by Matsentralen in Oslo, indicating the total amount of CO₂-eqv saved through redistribution.

Food category	Food redistributed in 2015, kg	Kg CO ₂ -eqv per kg food	Kg CO ₂ -eqv , total
Meat	77 000	9,84	1065421
Dairy	169 417	5,64	954831
Grain products	96 956	1,66	161419
Vegetables	121 585	0,87	105474
Soft drinks	177 810	0,45	79125
Fruit	27 084	2,07	19424
Eggs	776	1,70	1319
Fish	15 575	1,73	25547
Candy	20 821	2,83	58923
Baby food	3 285	3,33*	10937
Cleaning products	17 363	3,33*	57817
Other	51 081	1,00	51273
Total kg of food		Average kg CO₂	Total kg CO₂
778 751		3,33**	2 591 510

* Due to the lack of data for these food types, the average kg CO₂-eqv calculated was used as substitute.

** The average is calculated based on the division of the different food types redistributed by Matsentralen in 2015 and can thus not be considered a general average for 1 kg of food.

P107. Life Cycle Assessment of Microalgae-based Antioxidants
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In recent years microalgae have been brought forward as a possibility for production of biofuels, bio-based chemicals, feed and bioactive compounds without using agricultural land. One way to achieve sustainability of algal-based products is by integration in biorefinery systems in which biofuels, bio-based chemicals and high-value chemicals are produced in parallel from one raw material, in this case microalgae. Lutein is an antioxidant that potentially promotes several health benefits and that can be used as a food supplement. This study considers future production of lutein in a biorefinery system in which algae production benefits from integration with other industries by utilizing their residual flows. These can be nutrients from waste waters, CO₂ from flue gases and excess process heat.

The main objective of this work is to assess the environmental performance of using microalgae for production of antioxidants such as lutein and make comparisons to conventional production of lutein from marigolds. The functional unit is 1 kg lutein and the algae biorefinery is assumed to be placed in Sweden.

A consequential approach which uses system expansion to account for by-products will be applied. The major by-products from algae production which is biomass after lutein extraction that in the first scenario is used as feed and in the second scenario is used for biogas production. Environmental impact categories of relevance for this study are GWP, EP and, energy and resource consumption. One key issue for this study is the data inventory.

Data on algae production in northern climates is scarce, Previous studies have made attempts to assess integration of microalgae production with industrial waste flows for more sustainable production. A review by Mayers et al. (2016)¹ showed that there were several inconsistencies in the data used, for example are previous LCA often based in production targets and not on what is technically feasible. A throughout inventory of data was performed in the review and that will provide input for this assessment and give guidance where assumptions are unavoidable. Integration with industrial waste flows will most likely provide benefits for the algae industry, however if the integration will be beneficial also for the other industrial part will depend on scale and conditions for integration.

¹Mayers J., Ekman Nilsson A., Svensson E. and Albers E. (2016) *Integrating microalgal production with industrial outputs –reducing process inputs and quantifying the benefits*, Manuscript submitted to *Industrial Biotechnology*

The assessment is still in preparation and which input data that will be used is not yet decided.

P108. Environmental life cycle assessment of insect mass-rearing facilities: current challenges and future prospects

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2irstea, Umr Itap, 3ynsect, 1inra, Umr Sas

Objectives: Insects are seen as a promising alternative source of protein for animal feed. Despite a growing literature in the field, very few studies have been conducted to assess environmental impacts associated with mass-rearing systems. The aim of this study was to provide references on a yellow mealworm (*Tenebrio molitor*) large-scale facility and to identify the principal hotspots.

Methods: A life cycle assessment from cradle to farm gate was performed using inventory data from a pilot building. Chemical composition of larvae and manure were obtained by laboratory analyses. Ammonia, methane, and nitrous oxide emissions were recorded by in situ measurements. LCIA was conducted using CML-IA baseline method and TCED v1.8 implemented in Simapro Software v8.1.0.60. We used the ECO-ALIM database for French feed ingredients and the ecoinvent V3.1 database for background data.

Results: Environmental impacts associated with one kg fresh live weight of mealworm larvae harvested after 13 weeks were significantly lower than those found in literature which can be explained by differences in electricity consumption and electricity country mix, and a lower feed conversion ratio. Feed contribution ranges from 32 to 90% depending on the impact category. The results for gas measurement show very low NH₃, CH₄ and N₂O emissions. The feed conversion ratio (kg of ingested feed per kg larvae harvested, calculated on fresh matter basis) was 1.78.

Discussion and conclusion: The feed production has the higher contribution to total impact over all categories. The feed conversion ratio is thus the main driver as in many animal production systems. This study raises several questions concerning the development and the assessment of future large-scale production systems: How to overpass the attributional LCA as the insect production may compete with traditional sources of ingredients for livestock feed? Is it possible to eco-design an insect biorefinery on the basis of pilot scale farms as the environmental impacts rely on the availability of large amounts of wastes and co-products? Is there a contradiction between a goal of efficiency for the production of protein for livestock feed and the idea of insect biorefinery implying the use of waste and co-products? All these points will be commented in the presentation on the basis of actual results.

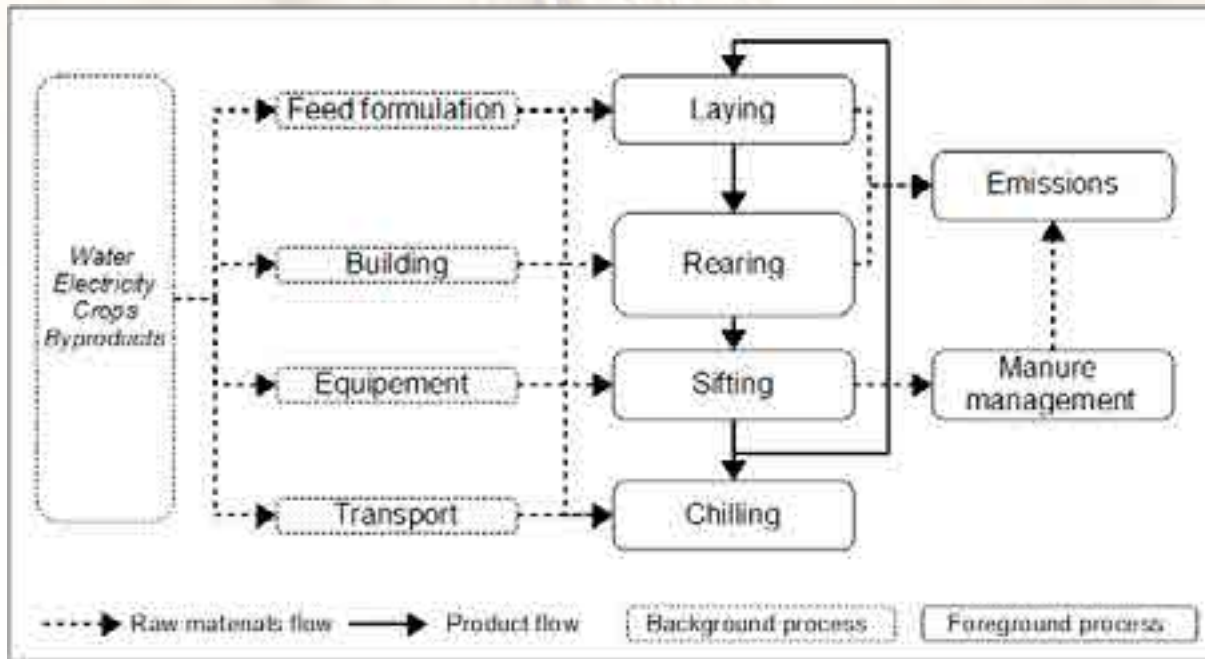


Figure 1: System boundaries for the cradle-to-farm gate production of one kg fresh live weight of mealworm larvae.

Table 1: Values and contributions of the main stages to total environmental impacts.

Impact categories	Production of feed	Electricity	Building and equipment	Water	Total
Acidification (g SO ₂ eq)	6,01	0,66	0,59	0,02	7,28
Eutrophication (g PO ₄ ²⁻ eq)	3,77	0,65	0,25	0,02	4,69
Land competition (m ² a)	1,26	0,01	0,13	0,00	1,40
Global warming (kg CO ₂ eq)	0,64	0,13	0,14	0,00	0,91
Cumulative energy demand (MJ eq)	7,62	14,19	1,75	0,06	23,62

P109. Environmental cost of Irish dairy farming

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Due to the great variety of environmental impacts associated with production chain. Weighting methods are need to aggregate all impacts into a single unit, which can provide results to stakeholders in a straightforward manner and more convenient for decision makers to make decision. Monetization could be an option for weighting and bridge the gap between environment and economy. The environmental burden of dairy farm takes a significant share in total environmental impacts from supply chain of milk production. In order to evaluate the sustainability of the Irish dairy industry, we need to understand the real cost of Irish milk production to society and thus the environmental cost arising throughout the life cycle of milk production has to be investigated. In addressing this issue, we modeled the environmental cost of a typical grass-based Irish dairy farm as a case study. Life cycle assessment was used to quantify the environmental cost of dairy farm by first classifying environmental impact flows from farm system to different midpoint impact categories or life cycle inventory results and then by aggregating these results into a single score using conversion factors in the Stepwise 2006, Eco-cost 2012 and EPS 2015, respectively. The results from these weighting methods will be compared and test the sensitivity of the weighting methods. All activity data are from farm survey and background data are from Eco-invent and the Gabi database. The results showed that LCA compatible monetization methods could be used for evaluation of total environmental impacts on Irish dairy farm. The total environmental impacts of Irish dairy farm could be reduced by more efficient management, with high herbage utilisation. Due to the difference in methodology, each method may lead to different environmental cost, in order to investigate the effect of management on dairy farming system, it is crucial to keep the method consistent.

Key words: milk production, environmental cost, life cycle assessment, sustainability

