colloque.inra.fr/lcafood2012

Proceedings

8th International Conference on

Life Cycle Assessment in the Agri-Food Sector

LCA FOOD 2012

October 1-4
2012
Saint-Malo
France

INRA

colloque.inra.fr/lcafood2012

Proceedings

8TH International Conference on

Life Cycle Assessment in the Agri-Food Sector

October 1-4, 2012 • Saint-Malo, France

EDITORS
Michael S. Corson,
Hayo M.G. van der Werf

INRA

Please cite this publication as:

Corson, M.S., van der Werf, H.M.G. (Eds.), 2012. Proceedings of the 8th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2012), 1-4 October 2012, Saint Malo, France. INRA, Rennes, France.

Please cite a paper or abstract in this publication as:

Proust, V.L.G.E.M., 2012. In search of lost time, in: Corson, M.S., van der Werf, H.M.G. (Eds.), Proceedings of the 8th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2012), 1-4 October 2012, Saint Malo, France. INRA, Rennes, France, p. 20-25.

ISBN 978-2-7466-5740-3

9 782746 657403 >

Table of Contents

Welcome

Soyez les bienvenus à LCA Food 2012 à Saint-Malo, France !
Welcome to LCA Food 2012 in Saint Malo, France!

"Towards Sustainable Food Systems"

The LCA FOOD conference series is the world's premier scientific and technical forum on Life Cycle Assessment in the agri-food sector. We hope that you will find the conference interesting and enjoyable and that you will "harvest" new ideas and contacts. Your input to the conference will contribute to its success.

The previous conferences in this series took place in Brussels (1996, 1998), Gothenburg (2001, 2007), Horsens (2003), Zürich (2008) and Bari (2010). This year, for the first time, the conference takes place in France. It has been organised by INRA, the French National Institute for Agricultural Research, with the support of ADEME, the French Environment and Energy Management Agency.

Objectives of the conference

The production, transformation, distribution and consumption of food and drink contribute strongly to human prosperity and health. However, the food and agriculture sector also contributes a large part of the environmental impacts caused by human activities. Because these impacts, in particular climate change and biodiversity loss, need to be reduced urgently, a shift towards sustainable food systems is essential.
Over the last two decades the Life Cycle Assessment (LCA) methodology has been developed and applied in the agriculture and food sectors to quantify environmental impacts and assist decision making. In recent years, LCA in the agri-food sector has developed rapidly, in particular for sustainability assessments of agricultural systems and their products, and for guiding consumers toward sustainable food-consumption patterns (e.g., via eco-labelling).
LCA Food 2012 will serve as a global forum in which to share recent developments in LCA methodology, databases and tools, as well as applications of LCA to food-production systems and food-consumption patterns. All of this will contribute, we hope, to achieving the 2012 conference motto: "Towards Sustainable Food Systems".

From the 362 abstracts submitted, the conference is scheduled to have 121 oral presentations and 183 posters, and at the time of writing, we expect more than 420 participants from at least 42 countries. In addition to this book of abstracts, which contains 2-page abstracts for most oral presentations and posters, you will find 6-page papers for most oral presentations, along with the poster abstracts, in the conference proceedings, provided as a PDF file on the memory stick in your conference beach bag.

We want to thank the authors for their presentations and posters. We are very grateful to the 23 members of our scientific committee for their efforts in reviewing the abstracts and selecting the papers for oral presentations. We warmly thank our sponsors for supporting the conference. Last but not least, we want to thank our indefatigable INRA colleagues of the organising committee for their essential contribution to the success of the conference.

We hope you will appreciate the scientific and technical content of the conference, contacts with participants, the French and Breton cuisine during the lunches and Gala Dinner, and the city of Saint Malo and its seaside. We are delighted to welcome you to this beautiful region to join the rapidly growing LCA Food community and hope you will meet old friends and make new ones.

Michael Corson
LCA Food 2012 co-chair

Hayo van der Werf
LCA Food 2012 co-chair

LCA Food 2012 Committees

Scientific Committee	
Hayo van der Werf	INRA, Rennes, France (co-chair)
Michael Corson	INRA, Rennes, France (co-chair)
Assumpció Anton	IRTA, Cabril, Spain
Joël Aubin	INRA, Rennes, France
Claudine Basset-Mens	CIRAD, Montpellier, France
Cécile Bessou	CIRAD, Montpellier, France
Christel Cederberg	SIK, Göteborg, Sweden
Leda Coltro	Institute of Food Technology, Campinas, Brazil
Imke de Boer	WUR, Wageningen, Netherlands
Ulrike Eberle	Corsus, Hamburg, Germany
Benoît Gabrielle	AgroParisTech, Paris, France
Geneviève Gesan-Guiziou	INRA, Rennes, France
Jeroen Guinée	CML, University of Leiden, Netherlands
Kiyotada Hayashi	National Agriculture and Food Research Organisation, Tsukuba, Japan
Niels Jungbluth	ESU services Ltd., Uster, Switzerland
Llorenç Milà i Canals	Unilever, Sharnbrook, United Kingdom
Rattanawan Mungkung	Kasetsart University, Bangkok, Thailand
Thomas Nemecek	ART, Zürich, Switzerland
Bruno Notarnicola	University of Bari Aldo Moro, Bari, Italy
Brad Ridoutt	CSIRO, Melbourne, Australia
Rita Schenk	Institute for Environmental Research and Education, Vashon, WA, USA
Ulf Sonesson	SIK, Göteborg, Sweden
Marlies Zonderland-Thomassen	AgResearch, Hamilton, New Zealand

Organising Committee	
Aurélie Wilfart	Thi Tuyet Hanh Nguyen
Eric Beaumont	Maryvonne Pertué
Sylvaine Bitteur	Barbara Redlingshöfer
Xiaobo Chen	Thibault Salou
Michael Corson	Thierry Trochet
Karine Derrien	Hayo van der Werf
Emmanuelle Garrigues	

LCA FOOD 2012

1-4 October 2012 • Saint Malo, France

PROGRAMME

Programme Overview

Monday, October 1st	
16:00 - 18:30	Participant arrival and registration
18:30 - 20:30	*Welcome reception : cocktail and buffet*

Tuesday, October 2nd			
08:00 - 08:50	Participant arrival		
08:50 - 09:10	Opening session		
09:10 - 11:10	Keynote session		
11:10 - 11:40	*Break*		
11:40 - 13:00	Parallel session 1a	Parallel session 1b	Parallel session 1c
13:00 - 14:30	*Lunch*		
14:30 - 15:50	Plenary session 1		
15:50 - 16:20	*Break*		
16:20 - 16:50	Poster session A		
16:50 - 18:30	Parallel session 2a	Parallel session 2b	Parallel session 2c

Wednesday, October 3rd			
08:30 - 08:50	Opening of the Conference Centre		Quantis Workshop *Amphithéâtre Maupertuis*
08:50 - 10:30	Plenary session 2		
10:30 - 11:00	*Break*		
11:00 - 13:00	Parallel session 3a	Parallel session 3b	Parallel session 3c
13:00 - 14:30	*Lunch*		
14:30 - 15:50	Plenary session 3: Food		
15:50 - 16:20	*Break*		
16:20 - 16:50	Poster session B		
16:50 - 18:30	Parallel session 4a	Parallel session 4b	Parallel session 4c
19:30 - 24:00	*Congress Gala dinner*		

Thursday, October 4th			
08:30 - 08:50	Opening of the Conference Centre		GaBi Workshop *Amphithéâtre Maupertuis*
08:50 - 10:30	Parallel session 5a	Parallel session 5b	
10:30 - 11:00	*Break*		
11:00 - 13:00	Parallel session 6a	Parallel session 6b	Parallel session 6c
13:00 - 14:30	*Lunch*		
14:30 - 16:10	Parallel session 7a	Parallel session 7b	Parallel session 7c
16:10 - 16:40	Conference Closure		

Friday, October 5th	
08:30 - 14:00	Optional tour of Mont Saint-Michel or Dinan/Dinard
09:00 - 16:30	3rd International ecoinvent Meeting

Room color code:

- Auditorium Châteaubriand
- Rotonde Surcouf
- Amphithéâtre Maupertuis

GENERAL INFORMATION

General Information

Conference venue
The conference takes place in the Palais du Grand Large, 1 Quai Duguay Trouin, 35400 Saint Malo, France. Tel. +33 2 99 20 60 20. Smoking is prohibited in the conference centre.

Registration
The registration fees include:
- Admission to all conference sessions, poster sessions and the exhibition area
- A conference beach bag, including this book of abstracts, your badge, an electronic version of the proceedings on a memory stick, the conference programme, a list of participants, a conference mug, and an INRA pen.
- Welcome reception: cocktail and buffet on 1 October, 18.30 hours
- Lunches: 2, 3 and 4 October, 13.00-14.30 hours
- Gala Dinner: 3 October, 19.30-24.00 hours
- Refreshments during session breaks
- Access to the Quantis and PE International workshops

Upon registration you will receive a badge to be worn during the conference.

The Young Researcher Wall
LCA Food 2012 is happy to offer Ph.D. students and other young researchers an opportunity to connect with research institutes and private companies by publishing their CVs on the conference web site as well as during the conference on the Young Researcher Wall, located at the entrance of the Salle du Grand Large (level 1). Research institutes and private companies can also publish their job offers on our web site and on the Young Researcher Wall during the conference.

Oral presentations
Plenary-session presentations will occur in the Auditorium Chateaubriand (level 0). Parallel-session presentations will occur in the Auditorium Chateaubriand and two rooms on level 2: Rotonde Surcouf and Amphithéâtre Maupertuis.
Ideally, presentations should not exceed 15 minutes in length; timekeepers will sound a bell to indicate when 2 minutes remain. Any time remaining in the presentation slot will be available for questions. Timekeeping will be strict to allow participants to switch between sessions.
For those who have not sent in their presentation before the conference, please give it to Thierry Trochet in the Preview Room in Salle Charcot (level 1) the day before the presentation is scheduled.

Poster sessions
Posters should be put up in the Salle du Grand Large (level 1) on October 2 between 7.00 and 8.50 hours and stay up for the entire conference. Posters have been grouped according to topic. Two poster sessions have been scheduled: session A on 2 October, 16.20-16.50 hours, and session B on 3 October, 16.20-16.50 hours. Poster authors should stand next to their poster during the session in which their poster has been scheduled.

Exhibition booths and Breaks
Sponsors have exhibition booths available to present their products and services in the Rotonde Jacques Cartier (level 1), where refreshments will be served during morning and afternoon session breaks.

Lunches and Gala Dinner
Lunches will be served from 13.00-14.30 hours in the Espace Lammenais (level 3). Special food requirements (vegetarian, fish) expressed during online registration have been taken into account. Those who registered their desire for a special meal should be sure to wear their badge during the meal so they can be identified by the wait staff. Persons having other requirements (e.g. vegan, allergies) should inform wait staff.
The Gala Dinner will be served on 3 October from 19.30-24.00 hours in the Espace Lammenais.

Internet access
Six computers with Internet access are available in the Salle Bouvet (level 1). Internet access via wifi is available on levels 1 and 2. A password is not required.

Cloakroom
On Thursday, 4 October, a staffed cloakroom will be available on level 0 to keep your luggage.

Questions?
The members of our local organising committee will be happy to answer any questions you may have. They wear pistachio-green T-shirts with the LCA Food 2012 logo on the back. In the Salle Charcot (level 1) you will find conference secretaries Karine Derrien and Maryvonne Pertué for administrative matters (e.g., attendance certificate).

Conference Centre Map

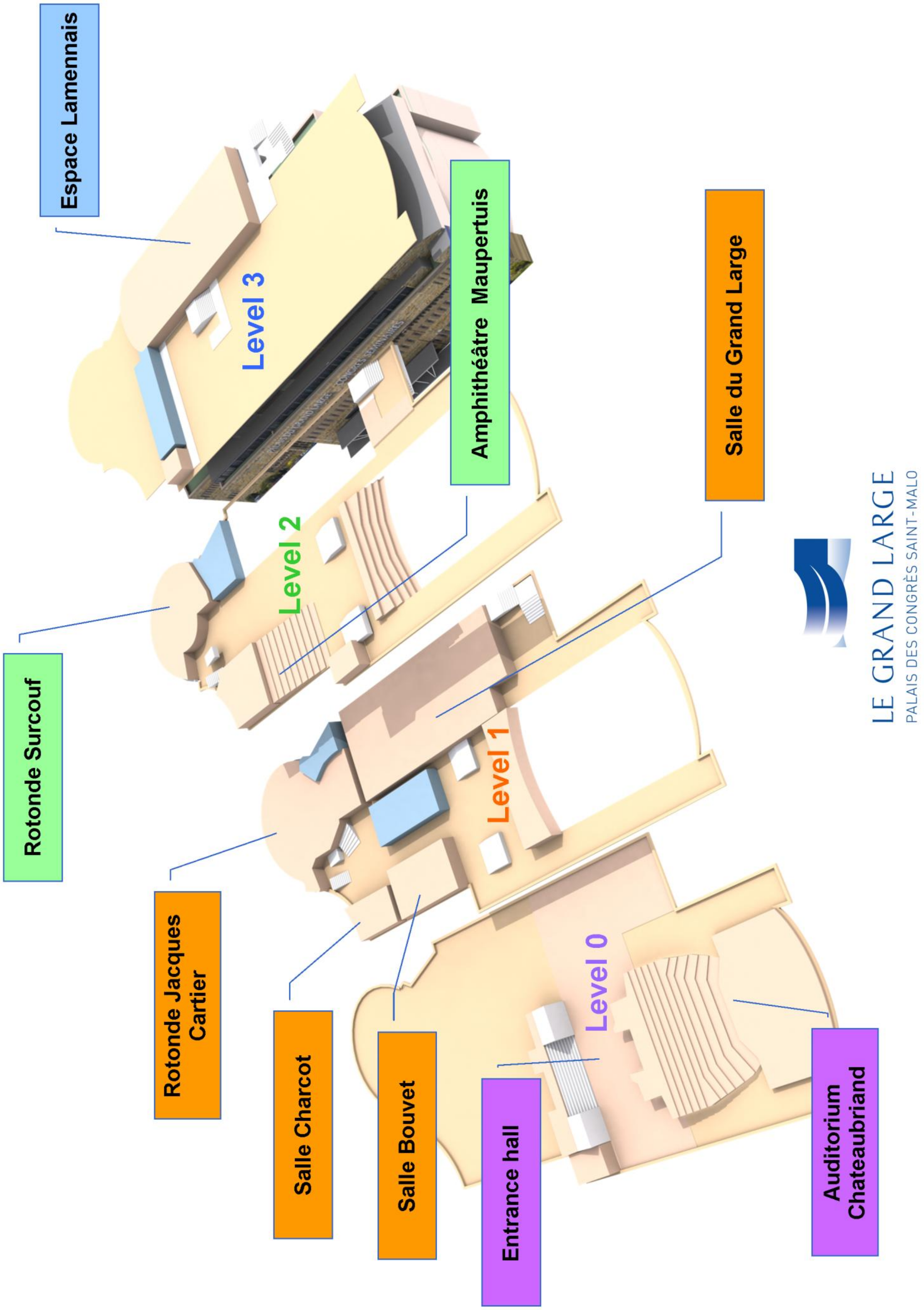


Table of Contents for Sessions

TUESDAY, 2 OCTOBER

POSTER SESSION A - TUESDAY, 2 OCTOBER, 16.20-16.50

GROUP 1, SESSION A: ANIMAL PRODUCTION SYSTEMS

GROUP 3, SESSION A: LABELLING, CONSUMERS, DIET

POSTER SESSION B - WEDNESDAY, 3 OCTOBER, 16.20-16.50

GROUP 4, SESSION B: CROP PRODUCTION SYSTEMS

GROUP 5, SESSION B: FOOD PRODUCTS

ORAL SESSIONS

Sustainable food, a component of the green economy

Dominique Dron

Interministerial Delegate and General Commissioner for Sustainable Development, Ministry of Ecology, Sustainable Development and Energy, Paris, France.

ABSTRACT

From the French perspective, the green economy has both economic and environmental but also social objectives. Agriculture is at the crossroad of various environmental challenges and involves living organisms. Sustainable agriculture calls for new public policies which mobilise both consumers and producers. The proposal for environmental labelling of products aims at launching signals to these different stakeholders to redirect their practices towards a low-carbon economy with low use of natural resources and pollution.

Keywords: green economy, sustainable food, environmental labelling

1. The challenges of the green economy

1.1 Roads towards a green economy after the "Rio+20" conference

Firstly, during the "Rio+20" summit, France defended a certain view of what a green economy should be. It cannot be limited to a pure environmental approach, and even less to the question of climate change. This economy has to be friendly to natural balances (climate, ecosystems, etc.), sparing resources and be also low-carbon. Its objectives are environmental and economic, but also societal: it has to change consumption patterns in parallel with production methods. In fact, the green economy requires a new pattern of development with its cultural changes.

It should be based on new technologies which spare energy, preserve natural resources and reduce greenhouse gases. Although new sectors have to emerge to modify the way our societies work, all existing sectors, and agriculture in particular, must also change their production models.

Refocusing economic models requires different types of tools:

- Wider knowledge and better information available to stakeholders. Research is required for this, as is specific work to inform producers and consumers. In particular this requires working on indicators downstream of science as these are means of conveying or translating scientific knowledge to users, practitioners and citizens.
- Regulatory instruments and economic instruments. Complying with the constitutional principle of polluter-pays in particular requires taxes and subsidies to be settled. Yet immediate economic interests of some representatives of farming often oppose the former and do not enable the latter to be ideally targeted. Here again, scientific work upstream is essential to build a given instrument, prove its legitimacy and assess its influence and effects.

The second element highlighted by Rio+20 is the necessity of making progress on an international level. The conference underlined the gap between industrialised countries, emerging countries and developing countries with regard to the concept of green economy, whose global nature, particularly on the social level, is often desired but has not been recognised yet. We therefore welcome the interest of international work leading to general awareness raising, prior to action, such as the LCA-FOOD 2012 conference. Better identifying the impact of production and consumption methods in different regions of the planet is essential in the hope of establishing fruitful and effective compromises in future. For this we need new, culturally shared tools on a scientific basis.

Finally, the third idea to be taken from Rio+20 is the ever increasing role of society, often called “civil society”, in negotiations. This mobilisation, and we could say this questioning, not only target States and public decision makers. They also address the research community. They should lead us to increase integration of disciplines and activities and to involve society in specialists' work far upstream.

1.2 The agricultural nexus

As to sustainable agriculture, the starting point is the following point shared by the research community and international futurists: sufficient and varied food for 9 to 10 billion people in 2050 can be reached, but only if food production almost doubles.

This scenario would imply a strong mobilisation of natural resources. Over the same period, energy demand will thus strongly increase, as will pressure on water resources. The scale and success of the 6th World Water Forum (March 2012 in Marseille) enabled these analyses to be shared. A nexus is now commonknowledged, illustrating the intricate questions which are bundling the food/water/energy challenges, themselves linked to ecosystem functions as underlined during the preparatory work for Rio+20. In addition, 43% of terrestrial area is already urbanised or cultivated, consequently reducing the regulatory capacity of ecosystems by almost half.

Agriculture is at the heart of this nexus. And yet the question of large-scale agricultural production is still too often used as a simplistic watchword, masking the complexity of the actual situation. In reality, as the FAO says, "climate-smart farming systems that make efficient use of resources like water, land, and energy must become the basis of tomorrow's green economy". We must add the dependence of conventional systems on non-renewable materials like phosphate deposits, and essential soil conservation.

Our preoccupation is therefore to commit to change towards sustainable systems. This was highlighted by the Grenelle programming law, proposing to generalise sustainable and productive farming practices.

1.3 Widening the concept of nexus: climate change and biodiversity

The idea of a nexus leads to mobilising multiple resources; it underlines the links between economy and environment. It is useful but remains incomplete, notably because, as we have seen, it does not really take into account biodiversity or climate change. Thus it only addresses environmental challenges in so far as they directly contribute to production. But as we look deeper inside natural processes, for reasons of sustainable development, and considering the precautionary principle and the notion of strong sustainability, preservation of natural environment must be added to it.

The work of INRA has highlighted the depressant effect of climate change on wheat yields. This effect is seen in European countries that have sought to free themselves as far as possible from natural production conditions to obtain the highest yields in the world. They are also observed in wheat and other crops in other countries in the world (Lobell et al., 2011). After a long period of wheat yields growth, their stagnation since the 1990s or even the 1980s can be related to the hydric and thermal stress caused by climate change. As far as agriculture is concerned, climate must be considered as a natural resource in itself, a challenge distinct from the energy subject.

A lot remains to be understood about the functions of living organisms and the relationships that agricultural production forges with its environment. Biologists are highlighting complex equilibria in which every living component plays a role. Dupraz and Capillon (2005) and Dupraz, Liagre and Borrell (2005) recognised the roles played by trees in promoting and protecting crops. We therefore have to better identify and characterise ecosystems, focusing on the ecosystem services provided. This is the purpose of numerous studies worldwide, to which your research institutions are contributing.

We know that ecosystems and their different components do not provide interchangeable services; there is no principle of general equivalence between the various elements of the natural environment. The services provided by ecosystems must be quantified and valued for economies and societies to measure to what extent and for which reasons they are depending on them. This absolutely does not mean that ecosystem exchange markets have an environmental science-based footing, nor that money could replace a natural service. Nonetheless, the value of biodiversity - to concentrate on this - is on the agenda for public decision makers and research. The purpose of the assessment is above all to underline the challenges. It must be equally qualitative and quantitative, and be expressed in physical volumes and monetary value to progress towards economic instruments for preservation.

As a conclusion to these initial remarks, a few points can be highlighted at this stage:

- The link between production and consumption, and the need to change simultaneously both parts of the equation in order to move towards a green economy which could be viable for a population of 9 to 10 billion sharing the same planet.
- The role of society and of interaction with all stakeholders, to build viable models for the future.
- The strong linkages between natural resources and environment in general on one side, and economy and production on the other.
- The multi-criteria nature of the environmental challenge, which in reality covers several fields, all the more so when considering the aim of sustainable development. Therefore, we must consider that there is no absolute indicator and that a set of indicators is required in spite of legitimate demands for simplification.
- The need for research in numerous fields: identification and characterisation of ecosystem services, environmental impacts, their measurement, and the assessment of their value and their functional links with a view to sustainable production.

2. Tools for sustainable food

Sustainable food is one component of a green economy. It is also an emerging theme that needs to be characterised in order to establish suitable areas to focus on and identify the tools needed to achieve it.

2.1 State incentives

Food is a major sustainability challenge. According to the EIPRO (Tukker et al., 2006*)* study, it represents 20 to 30% of all consumption impacts. Sustainable food sets the challenge of correctly feeding, in quality and quantity, 9 to 10 billion people in 2050, whilst preserving the vital functions of our planet. The importance of health in the social pillar of sustainable development is a characteristic of sustainable food. The sustainability of food systems is therefore a complex question with no single solution.

Research on agricultural production methods that have proven to be sustainable both today and historically is one part of the answer, as the former example of trees and crops shows. One of these production methods is organic farming. It proved its environmental quality: scientific evidence attests the positive impact of organic farming on soil fertility, water resources, ecosystemic functions, landscape and biodiversity. This led to the ambitious objectives being set for supply and consumption of organic products by the "Grenelle de l'Environnement" in France: 6% of the national total cultivated area devoted to organic farming in 2012 and 20% in 2020. Organic farming has also demonstrated its capacity to produce in large volumes insofar as it is not limited in nitrogen (Seufert et al., 2012).

In addition, public authorities are encouraging the diversity of agri-food marketing channels, thus responding to a social demand which is itself diversified. They are doing this through supporting the development of alternative marketing channels, in addition to the dominant marketing model of supermarket distribution. For example, short distribution circuits and fair trade enable socio-economic factors to be better integrated into commercial exchanges. Both are examples of food policies implemented in France.

However, the challenge of sustainability in the whole agri-food sector requires more global approaches. The Government is contributing to this in particular through the development of "public green markets" which consist in including environmental criteria in public markets and thus increasing the purchase of ecological products by the public sector. The correct environmental criteria remain to be identified, which points at the former discussion about LCAs.

2.2 Industrialists' approaches to environmental management

Tools have also been implemented to enable business to move towards integrated sustainable approaches. The agri-food sector understood that sustainability could be a source of innovation and competitiveness. Notably, for several years it has been undertaking work on Corporate Social Responsibility (CSR). According to standard ISO 26000, this is *"the responsibility of an organization for the impacts of its decisions and activities on society and the environment"*. This is expressed in particular *"through transparent and ethical behaviour that contributes to sustainable development, including health and the welfare of society [and] takes into account the expectations of stakeholders"*. CSR leads professionals to ask to themselves several questions on the impacts of the organisation and its stakeholders, as well as on possible areas for improvement. It illustrates the possible role of industrialists in sustainability, considering all the stakeholders involved.

The ANIA (French National Association of the Food Industries), Coop de France (French federation of cooperatives) and AFNOR (French Standardisation Agency) have worked together to develop a guide to using standard ISO 26000 for the agri-food sector. This tool can help industrialists who wish to materialise their action to promote sustainability. In particular, it identifies the specific challenges of CSR for the sector. We particularly note its proposals of challenges and actions to take into consideration the agricultural production level and encourage its participation when applying the CSR approach.

This work represents an important step towards sustainability to be taken into account by the agri-food business. Both organic farming and CSR are mobilising certification and environmental management tools. These start from voluntary approaches of stakeholders and not from regulatory instruments. The State's intervention consists in ensuring that the regulatory framework necessary to these approaches is in order and in economic support of organic farming and, to a far lesser extent, short distribution circuits.

2.3 Action on consumption patterns

We could not feature talking about sustainable food systems without discussing the role of consumers. Indeed, the progress realised by sectors could not deliver if consumers did not adapt their behaviour. The Collective Scientific Expertise by INRA (Etiévant et al., 2010) on the determining factors on dietary behaviour of French people underlined the difficulty of changing individual practices. Here again, a combination of several tools appears to be necessary.

In this field, the agri-food sector can be considered as a pioneer with regard to the nutritional aspects: generic consumer information such as "5 fruits or vegetables a day", community schemes to distribute milk and fruit in schools, nutritional rules in French school canteens and the labelling of nutritional characteristics on products are all tools created to educate consumers throughout their daily life with a view to eating a varied and balanced diet. Confronted with the new questions of sustainability, the agri-food sector is now widening its consumer-focused action to meet other challenges. The profession is highly involved in the development of environmental labelling, particularly in France and Europe.

A large amount of research remains to be conducted in the food sector, as highlighted by the Agrimonde (Paillard, Treyer and Dorin, 2010) and duALIne (Esnouf et al., 2011) foresight studies by INRA and CIRAD. These two studies converge in underlining the importance of diet in sustainable food. In particular, with regard to the environmental impacts of eating habits, the essential role of both quantities consumed and the consumption of animal products in the diet has been stressed.

DuALIne also shows that an integrated approach to sustainability in food research must urgently be developed. Therefore the ANR (National Research Agency in France) launched the ALID (sustainable food systems) research program. After two years of calls for proposals, we can draw the conclusion that inducing communities of researchers to update their working methods and jointly address the three challenges of sustainable, economic, environmental and social development (including the significance of health in relation to food) is not easy. In 2012, more projects were corresponding to the spirit of the calls and this progress is to be praised. Two projects in particular are planning to work on the assessment of agri-food products according to the three dimensions of sustainable development and in particular to reduce their impacts. Nonetheless, this type of project remains all too rare. This demonstrates the scale of the challenge to be met by research teams in terms of sustainable development.

2.4 Action on agri-food processes

Food sustainability is also to be achieved through significant improvements in processes and channels. We would just mention here two directions for deliberation and research.

The first one is inspired by the physiology of living organisms: observing that to feed their chicks, adult Emperor penguins could regurgitate fish that had remained fresh in their stomach after several days, Hubert Curien Institute in Strasbourg (France) understood the process and isolated a molecule which stops the deterioration of tissues. This discovery was at the root of patented applications that replace the use of refrigerators. The second one arises from observing the increasing vulnerability of crops to various hazards, and the significant robustness of combined or mixed crops, to question the capacity of agro-industries to increase the security of their supply by improving their plasticity. Distributors and consumers will also be concerned!

Following conclusions can be drawn from these considerations on the sustainability challenges for the agri-food sector:

- the necessity for the State and private stakeholders to combine a wide range of instruments for intervention, as the challenges of sustainable food are themselves multiple and there cannot be a single solution;
- the importance of actions targeting consumers, with a result which remains unforeseeable; here again, a combination of tools is essential to be effective;
- strengthening actions targeting consumers for themes other than nutrition is one key of the sustainability of the agri-food sector; the need for more objective environmental information in particular has been identified, giving guarantees to the consumer;
- finally, the need for research to progress in the assessment of environmental and social impacts of both consumption and production, impacts that remain complex if we want to take account of all the challenges; the question of diets and their sustainability is at the heart of the research questions: many foresight models show that a change in diet of rich populations through the world towards fewer animal proteins is essential to preparing a planet viable for 9 to 10 billion.

3. The environmental labelling project

This last part puts an emphasis on labelling products with regard to environmental impacts.

3.1 The operational scheme in France

The French environmental labelling project corresponds to actions targeting consumers, to help them orientating their choices in favour of less impacting products. This project arose from the "Grenelle de l'environnement": article 54 of law No. 2009-967 of 3 August 2009, known as "Grenelle 1", establishes the consumer right to "be given access to sincere, objective and complete environmental information concerning the overall characteristics of the product and packaging".

This strong Grenelle measure should enable consumers to include an environmental component when purchasing, whilst providing the entire production and distribution chain with new, specific management indicators. It thus encourages the eco-design of products.

The deployment of this project requires harmonised environmental assessment methodologies, the availability of the data necessary to the LCA calculation and access to the needed calculations for all.

To achieve this, the ADEME-AFNOR (2011) platform has produced a general best practices guide for the assessment of products' environmental impacts, BPX-30-323-0. Sector-specific working groups have been put into place within the platform to develop sector-specific environmental assessment reference documents.

The platform proves the efficiency of an exemplary governance: progress by consensus, free enrolment and participation, and an open process of establishing technical reference documents. This enables it to bring together various stakeholders: LCA scientific experts, experts from various businesses and sectors, associations, federations, environmental NGOs, consumer associations, government ministries, ADEME and AFNOR. 1,200 participants are involved and 700 bodies are represented within the platform.

The platform's No. 1 working group, dedicated to the agri-food sector, has established the environmental impact assessment reference document for food products, which has now been finalised and validated. The main characteristics of this reference document are as follows:

- functional unit: 100g/100ml, or the portion when this can be defined;
- life cycle phases: production (including of packaging), transport, manufacturing process, marketing and distribution, use and end of life of the product and packaging;
- the environmental impacts adopted for the products in the sector and the calculation methods are:

Environmental impact category	Indicator	Unit	Impact calculation method
Climate Change	Greenhouse gases emissions	g CO_2-equiv.	IPCC 2007
Water consumption	Water consumption	Litres	Net consumption; Water released in another watershed from which it was withdrawn is not included; sea water or stable groundwater (over 3 years) is not included
Water quality	Marine eutrophication	g N-equiv.	Recipe 2008
Water quality	Aquatic ecotoxicity	CTUe	USE Tox
Biodiversity loss	To be defined	To be defined	To be defined

Agri-food sub-sectors are currently establishing product category rules (PCRs) specific to their product family: oil, milk, wine and spirits, mineral water, coffee and pet food.

In parallel, the national trial defined by article 228 of French law no. 2010-788, known as "Grenelle 2", is being used to verify the feasibility of the approach on a practical level.

3.2 Work at a European scale

Significant work on methodology is also being conducted at European level and, overall, is highly consistent with the French work. Both levels share a multi-criteria approach to assessing the environmental impact of products which is not limited to the carbon footprint. In the agri-food sector, this approach enables the inclusion of the fight against climate change, the preservation of biodiversity, water quality, etc.

The European Commission is relying on its Joint Research Centre (JRC), to develop a European standard for assessing products' environmental footprints (PEF-guide). This document is currently being finalised. European directives may later draw on the PEF-guide to implement larger scale policies on environmental information to consumers. The JRC and European agri-food sectors are also working together on the assessment of products' environmental impacts through the Food SCP (Sustainable Consumption and Production) Round Table. This Round Table should finalise the ENVIFOOD Protocol, a harmonised methodological guide to assessing the environmental impacts of agri-food products, in autumn 2012.

France supports harmonisation in this field and is ready to adopt European methodological standards as soon as they have been established via multi-stakeholder consultation processes, as is currently the case.

3.3 Which research for which project?

There still remains a significant need for research to achieve a complete environmental assessment of agri-food products. Scientific advances, which concern all the participants to the LCA-FOOD 2012 conference, are particularly anticipated for certain environmental challenges, namely:

- a "water footprint" approached with more detail and relevance to agriculture, in other words, taking account of the vulnerability of the environments from which water is abstracted,
- a "biodiversity footprint"; on this point, LCA approaches that put too much emphasis on the effect of land occupation are poorly understood by either agricultural producers or consumers. The MEDDE (Ministry of Ecology, Sustainable Development and Energy) believes important progress must be made to better take into account agriculture's impact on biodiversity and, at this stage, proposes putting the emphasis rather on areas of biodiversity on farms.

More generally, tools are needed that can better take into account the environmental interest of extensive farming's low use of inputs. The choice to use the functional unit of the kg or litre is liable to give a bonus to the most intensive use of inputs, which makes sense for the overall environmental impacts but risks relegating local impacts to the background. We are waiting for research to bring an analytical view to this point in order to inform the public decision.

Another sensitive subject is the measurement of greenhouse gas emissions taking into account carbon storage (Rabl et al., 2007 ; Benoist et al., 2012). Agricultural areas effectively present the characteristic of not only being carbon emitters but also potential carbon sinks. LCAs, which originally stem from an industrial environment, are still struggling to take account of this dimension. And yet it is essential, both from the scientific point of view and to ensure LCA approach to be acceptable to agricultural producers.

These points lead to a more general remark on the relationship between research and society's stakeholders. As mentioned above, sustainable development requires an increased dialogue between all parts of society. Notably, research must be conducted in interaction with the needs and questions raised by businesses and NGOs, including by taking into account their points of view on what the research should be and how it should be conducted.

Public and private stakeholders are called to work with research to democratise LCA and provide the necessary methods, tools and data. Research must lead to operational methods and robust data. In the agri-food sector, the short-term provision of data that responds to the diversity of the methods of production, processing, distribution and use by consumers is essential. The in-depth work already conducted by several sectors should enable the calculation of new LCAs to be simplified.

The operational involvement of the scientific community is essential to ensuring the rigour of the methods and tools as well as the quality of the data. Researchers are already involved, particularly in French and European methodological work, but their involvement must increase further. This will enable all stakeholders, including the consumers, to appropriate the subject of the environment and act rapidly faced by the environmental crisis, in cooperation and by sharing scientific evidence culturally appropriated.

4. Conclusion

- environmental labelling is an appropriate tool for promoting sustainable development in the agri-food sector;
- the large-scale development of this tool requires methodologies, tools and data to be rapidly developed to calculate products' environmental impacts;
- to do this, the involvement of and dialogue between all stakeholders, including scientific communities, are essential; this is the price at which we will be able to confront the environmental crisis with the support of robust evidence properly appropriated;
- it may be useful to adopt temporary rules or provide rough data to begin with, in order to be able to shed some light over the short term and enable business and consumers to act immediately;

- there is a significant need for research; the assessment of products' impacts on the functions of living organisms in particular is of major importance to the agri-food sector and this subject should receive more attention in the international research agenda.

5. References

ADEME-AFNOR, 2011. General principles for an environmental communication on mass market products: BPX-30-323-0, 38 p.

Benoist, A., Dron, D., Zoughaib, A., 2012. Origins of the debate on the life-cycle greenhouse gas emissions and energy consumption of first generation - A sensitivity analysis approach. Biomass and Bioenergy, février 2012.

Dupraz, C., Capillon, A., 2005, L'agroforesterie: une voie de diversification écologique de l'agriculture européenne ?, Cahier d'étude DEMETER.

Dupraz C., Liagre F., Borrell T., 2005, "Economics of silvoarable systems using the Land Equivalent Ratio concept", in Silvoarable Agroforestry For Europe) final Report, available at: http://www.agroof.net/agroof_dev/documents/safe/Economics_silvoarable_systems_LER_approach.pdf

Esnouf, C., Russel, M. et Bricas, N., (Editors), 2011. duALIne – durabilité de l'alimentation face à de nouveaux enjeux. Questions à la recherche, Rapport Inra-Cirad (France), 254 p. Chapitre 3 consacré à la présentation de l'étude de cas ADEME-INRA sur l'impact carbone de régimes alimentaires différenciés selon leur qualité nutritionnelle.

Etiévant, P., Bellisle, F., Dallongeville, J., Etilé, F., Guichard, E., Padilla, M., Romon-Rousseaux, M. (editors), 2010. Les comportements alimentaires. Quels en sont les déterminants ? Quelles actions, pour quels effets ? Expertise scientifique collective, rapport, INRA (France), 277 p.

Lobell, D. B., Schlenker, W. and Costa-Roberts J., 2011. Climate Trends and Global Crop Production Since 1980. Science, Vol. 333, n°. 6042, pp. 616-620.

Rabl, A., Benoist, A., Dron, D., Peuportier, B., Spadaro, J., Zoughaib, A., 2007.How to account for CO_2 emissions from biomass in an LCA. Int J LCA 12 (5) 281, June 2007.

Paillard, S., Treyer, S., Dorin, B., 2010. Agrimonde Scénarios et défis pour nourrir le monde en 2050. Ed. Quae, 295 p.

Seufert, V., Ramankutty, N., Foley, J., 2012. Comparing the yields of organic and conventional agriculture. Nature 485, 229-232.

Tukker, A., Huppes, G., Guinée, J., Heijungs, R., de Koning, A., van Oers, L., Suh, S., Geerken, T., Van Holderbeke, M., Jansen, B., P. Nielsen, 2006. Environmental Impact of Products (EIPRO). Analysis of the life cycle environmental impacts related to the final consumption of the EU-25. Main report IPTS/ESTO project.

Research priorities for sustainable agri-food systems and LCA

Jean-François Soussana

INRA, Paris, France
E-mail: *jfsoussana@paris.inra.fr*

ABSTRACT

Recognizing that research for sustainable agri-food systems will be essential to meet global and European challenges in the coming decades, European countries participate in two Joint Programming Initiatives (JPIs): Agriculture, Food Security and Climate Change (FACCE) and Healthy Diet for Healthy Life (HDHL). Mission oriented research agendas have been developed and are focused on delivering key outputs. For FACCE: i) to sustainably intensify European agriculture, ii) to operate agriculture within greenhouse gas, energy, biodiversity and contaminant limits and iii) to build resilience to climatic change in agricultural and food systems. For HDHL: i) determinants of diet and physical activity, ii) developing healthy, high-quality, safe and sustainable foods, iii) diet-related chronic diseases. The role of life cycle assessment (LCA) in the context of these research priorities is discussed. Bridging nature capital, on the one hand, and health issues, on the other, with the assessment of the life cycle may lead to breakthroughs in the sustainability assessment of food systems.

Keywords: food security, agriculture, climate change, health, diets

1. Introduction

Today's agriculture and the food systems that it underpins are at crossroads. Food security – the availability of and access to sufficient and healthy foods and good nutrition - is central for the well being of people and nations. Until recently, it was expected that despite climate change and increasing world population, there would be several decades with food surplus - and low prices - ahead (IPCC, 2007). Contrary to this expectation, the volatility of world food prices has increased and two out of the last three years have been characterised by large spikes in international grain prices with some grains more than doubling in cost (von Braun 2008).

1.1 Agriculture, food security and climate change

A key challenge is to increase the global food supply to accommodate a world growing to 9 billion or more people by 2050 while preserving a safe operating space for humanity by avoiding dangerous environmental change (Rockström et al., 2009). Climate change is already negatively impacting food production (Lobell et al., 2011, Coumou and Rahmstorf, 2012), while the agriculture, land use and forestry sectors contribute almost one third of total greenhouse gas emissions and have a high potential for mitigation (IPCC, 2007). Yet, the Earth still provides enough. While a billion people go hungry, half a billion are obese. The global calories deficit reaches only a few percents of global supply by agriculture and could be compensated by reducing food over-consumption. Moreover, 40% of the grains are used to feed livestock and an additional 6.5% for biofuels. Furthermore, 40% of the totality of global food products is either lost after harvest or wasted (Beddington et al., 2012).

A number of recent studies (Beddington et al., 2012; Paillard et al., 2011; IPPC, 2007) have indicated the need for increasing research efforts in the area of agriculture, food security and climate change. International research programs (e.g. the CGIAR research programme on Climate Change, Agriculture and Food Security, CCAFS) have been initiated to address this for the developing world. A Global Research Alliance on agricultural greenhouse gases has also been launched (Shafer et al., 2011). The Joint Programming of research Initiative on Agriculture, Food Security and Climate Change (FACCE JPI) has been working over the past two years to define the critical research elements of a European response to food security under climate change.

Agriculture is a significant contributor to land degradation and anthropogenic global greenhouse gas emissions, being responsible for 25% of carbon (largely from deforestation), 50% of methane, and more than 75% of N_2O emitted annually by human activities (Tubiello et al., 2007). An estimated one third of the world's cropland is losing topsoil faster than new soil is forming and many of the poor live on degraded land (Nkonya et al., 2011). Agricultural expansion in the tropics is mainly based on deforestation, since 80% of new tropical croplands are replacing forests (Foley et al., 2011), which affects biodiversity and key ecosystem services. Species-rich regions of the world are under pressure from agriculture conversion, putting at risk animal and plant species, including hundreds of medicinal plants that are the basis for global health care (TEEB, 2010). Land use change resulting from expansion of agricultural land significantly contributes to CO_2 emission (IPCC, 2007). Unprecedented water shortages are also increasingly apparent in many parts of the world, including southern Europe (Fereres et al., 2011) and an increased frequency of heatwaves and precipitation extremes has caused widespread agricultural production losses in the last decade (Coumou and Rahmstorf, 2012).

In many European countries, the growth trends of the yields of major crops, especially wheat, have declined over the past two decades (Olesen et al., 2011). Moreover, the variability of crop yields has increased as a consequence of extreme climatic events, such as the summer heat of 2003 which led to 36 billion Euros economic losses for the agriculture sector in the EU (IPCC, 2007) and to large carbon losses from ecosystems (Ciais et al., 2005), the summer drought and heat in 2010 destroying vast areas of crop stands in Russia, and the 2011 spring drought in France. Future climate change impacts on the European agricultural ecosystems are likely to contrast increases in yield and expansion of climatically suitable areas in northern Europe, and more frequent water shortages and extreme weather events (heat, drought, storms) in southern Europe (Bindi and Olesen, 2011).

1.2. Diets and health

The food system in its entirety (including pre-chain inputs, agricultural production, food processing and retailing) is by far the largest industrial sector in Europe (Eurostat, 2008). European governments are struggling with the growing social and economic consequences of the alarming increase in obesity and diet related diseases, including malnutrition (Stratton, 2007) and micronutrient deficiencies and diet-related chronic diseases. Increased affluence and urbanisation tend to reduce daily physical activity and ready to eat foods with high energy densities tend to replace the traditional diets. In 2008, across the 27 countries of the European Union, 59% of adult men and 48% of adult women were either overweight or obese. There is growing evidence that obesity continues throughout the life cycle with associated health-related problems, such as type 2 diabetes, cardiovascular diseases, hypertension, and a range of cancers. Such lifestyle-related diseases have a negative impact on life expectancy, reduce the quality of life and lead to increased health costs (HDHL JPI, 2012).

2. Priorities in research

Recognizing that research will be essential to meet global and European challenges in the coming decades, European countries participate in Joint Programming Initiatives (JPIs). The mission of a JPI is to achieve, support and promote integration, alignment and joint implementation of national resources under a common research and innovation strategy for facing key societal challenges. Two JPIs are concerned with agri-food systems: Agriculture, Food Security and Climate Change (FACCE) and Healthy Diet for Healthy Life (HDHL). Both JPIs are developing strategic research agendas in consultation with stakeholders. A new European Era-Net, Susfood, on Sustainable Food Production and Consumption has also been launched. Its strategic goal is to maximize the contribution of research to the development of more sustainable food systems from farm to fork.

2.1 Agriculture, Food Security and Climate Change (FACCE JPI)

Twenty-one European countries contribute to the Agriculture, Food Security and Climate Change (FACCE) JPI (Soussana et al., 2012) and develop a common research and innovation strategy for facing the challenges at the intersection of agriculture, food security and climate change.

An integrated scientific research agenda has been designed and is focused on delivering key outputs: i) to sustainably intensify European agriculture to avoid increasing the demand on food production in other (e.g. developing) world regions, ii) to operate agriculture within greenhouse gas, energy, biodiversity and contaminant limits and iii) to build resilience to climatic change in agricultural and food systems. Crop and animal production systems of increased productivity with reduced environmental footprint per unit product should be developed. This will require accurate benchmarking (genotype x environment x management) of the main drivers, including socio-economics, of current agro-ecosystems, and the design, experimentation, and modelling of alternative systems. To substantiate this, a Scientific Research Agenda (FACCE - JPI, 2011) has been agreed, including five evidence based interdisciplinary core research themes, which will yield high returns with the prospect to reinforce the contribution of Europe to global public goods.

1. Sustainable food security under climate change, based on an integrated food systems perspective: modelling, benchmarking and policy research,
2. Environmentally sustainable growth and intensification of agricultural systems under current and future resource availability,
3. Assessing and reducing trade-offs between food production, biodiversity and ecosystem services,

4. Adaptation to climate change throughout the whole food chain, including market repercussions,
5. Greenhouse gas mitigation: N_2O and CH_4 mitigation in the agriculture and forestry sector, soil carbon sequestration, fossil fuel substitution and mitigating GHG emissions induced by indirect land use change.

To reach these goals a systemic understanding should be gained, by developing and integrating a large range of disciplines, that must be strongly connected to a foundation of agro-ecological and socio-economic modelling. Key European infrastructures need to be assembled and developed in order to integrate scenarios, observations, experiments and models, develop and inter-compare agro-ecological and socio-economic projections while assessing their uncertainties. This should result in possible pathways for innovative developments of European food systems and of a bioeconomy with a decadal time frame reaching 2030 or 2050. For instance, given the generation times involved in breeding major food crops, to have adapted commercial lines for 2030 will require at least 15 years of development.

The development of this research will require increased training and capacity building in a number of disciplines which have been neglected over the past decades (e.g. agronomy and animal husbandry, farming systems) and that need to move toward more integrated systems approaches, by better integrating developments from a range of other disciplines such as ecology, earth sciences, social sciences, applied mathematics and computing. A pilot action has been launched by the FACCE JPI concerning the modelling of climate change risks in the form of a large collaborative network (see www.macsur.eu).

2.2 Healthy Diets for Healthy Lives (HDHL JPI)

The scientific agenda of the HDHL initiative stresses that better diets and increased physical activity will contribute to preventing or reducing the risk of illness and to reducing the high costs of health services. Attempts to increase public awareness of the best way to eat more healthily have not led to major changes in patterns of food purchase and consumption. More attention must be given to finding ways to increase people's motivation, abilities and opportunities to make healthy choices (Brug, 2006). Research has shown that knowledge is often not a direct determinant of eating behaviour: some nutrition knowledge appears to be necessary but an insufficient prerequisite for health behaviour changes. The following three key interacting research areas were identified by the HDHL JPI:

1. Determinants of diet and physical activity: ensuring the healthy choice is the easy choice for all consumers. The challenge is to understand the most effective ways for improving public health through interventions targeting motivation, ability and opportunity to adopt and maintain healthy dietary and physical activity behaviours.
2. Diet and food production: developing healthy, high-quality, safe and sustainable foods. The challenge is to stimulate the European consumers to select foods that fit into a healthy diet and to stimulate the food industry to produce healthier, high-quality foods in a safe, sustainable and affordable way.
3. Diet-related chronic diseases: preventing diet-related, chronic diseases and increasing the quality of life. The challenge is to prevent or delay the onset of diet-related chronic diseases by gaining a better understanding of the impact of nutrition and lifestyle across Europe on human health and diseases (HDHL JPI, 2012).

3. Some recent trends in environmental assessment methods

The FACCE and HDHL joint programming initiatives have set mission oriented scientific agendas, underlining the interdependency between climate change, environment, agriculture and health issues and stressing the pivotal role of food systems. Given their complexity, such systemic issues are intrinsically difficult to boil down to changes in agricultural systems, in food products designs and in consumers choices. Nevertheless, both research agendas require improved methodologies for assessing the sustainability of agricultural supply chains and of food systems within Europe. In addition to environmental sustainability, economical and social dimensions, including foremost food security and health are also identified as key challenges for sustainability.

3.1 Towards more holistic assessments of environmental sustainability

Increasingly, environmental evaluation methods are moving towards integrated assessments by: i) incorporating several environmental dimensions, ii) connecting local and global issues and iii) assessing long term effects, as well as non linearities and thresholds.

First, there are many examples of recent assessments which have attempted to unify environmental dimensions which were previously seen as being distinct. For instance, the European Nitrogen Assessment (Sutton et al., 2011) considers the fate of anthropogenic nitrogen as a cascade of reactive N forms and effects. This cascade highlights how policy responses to different reactive N forms and issues are inter-related, and shows that a holistic approach is needed, maximizing the abatement synergies and minimizing the trade-offs. Another example is the Millenium Ecosystem Assessment (2005) which has provided a unified perspective on biodiversity and ecosystems, showing their role for a number of basic services to mankind: i) provisioning services (providing resources that are directly exploited by humans, such as food, fibres, water, raw materials and medicines); ii) supporting services (processes that indirectly allow exploitation of natural resources, such as primary production and pollination); iii) regulating services (natural mechanisms responsible for climate regulation, nutrient and water circulation, pest regulation, flood prevention, etc…); iv) cultural services (benefits people gain from the natural environment for recreational, cultural and spiritual purposes).

Second, planetary boundaries that must not be transgressed to prevent unacceptable environmental changes have been identified and to some extent quantified (Rockström et al., 2009). The increasingly perceived finite nature of world natural resources has provided support for connecting local and global issues. For instance, indirect land use changes arising from biofuel expansion on arable lands were identified as likely causes of tropical deforestation leading to large indirect emissions of carbon dioxide (Searchinger et al., 2008).

Third, more attention is now being paid to some of the long term consequences of current decisions and policies. The gradual integration of socio-economic and emission (SRES) scenarios, of climate models and of impacts, adaptation and mitigation studies (IPCC, 2007) has pioneered novel integrated modelling studies which have also been used in other areas than climate change, such as biodiversity (Bellard et al., 2012) and food security (Paillard et al., 2011). Some of the results, point to large scale irreversible changes, such as the decline of the Amazonian forest, the collapse of the Greenland ice sheet (Lenton et al., 2008) and of ecological networks (Barnosky et al., 2012).

3.2 Consumption and lifestyle based environmental assessment

Most of these holistic approaches to environmental assessment aggregate sectors and regions and report global impacts, as well as impacts per unit production or per unit land. Being production oriented, such assessments tend to miss the global consequences of consumption patterns and lifestyles. A different picture is obtained by including imports and exports, thereby reflecting the impacts of consumption patterns rather than those of production. For instance, not only does the European Union's own production result in significant greenhouse gas emissions, but as a net importer of primary agricultural and industrial commodities it causes large greenhouse gas emissions elsewhere (e.g. in China and Latin America, Davis and Caldeira, 2010). Another example concerns food systems, with studies showing that reducing overconsumption and food wastes (Paillard et al., 2011) or, more specifically, reducing meat consumption (Stehfest et al., 2009) has large potential consequences for food security and for the environmental sustainability of the agriculture sector.

4. Life cycle assessment in the context of research priorities

At a more disaggregated level, a number of methods have been used, such as material flow analysis, net energy analysis and life cycle assessment (LCA) to assess the environmental impacts associated to man made products. Given its flexibility and widespread use, LCA plays an important role in designing products, orienting consumer's decisions and evaluating policy measures (e.g. Weiss and Leip, 2012). However, the applicability of LCA to agricultural systems may be challenged by considering the boundaries of the system studied.

4.1 Nature capital and life cycle assessment: towards a new synthesis?

The emergence of LCA started with studies aiming at reducing the consumption of energy and of raw materials in the industry. In this context, the life cycle of a product can easily be seen from 'cradle to grave', that is from resource extraction to product disposal. Such boundaries tend, however, to become fuzzy when considering agricultural and food products which have a majority of their life cycle nested into biological processes. Ecosystem goods and services support the technological activities in the life cycle of agricultural and food products (Zhang et al., 2010a). Moreover because of their organic nature, food products and their associated by-products are ultimately recycled in multiple loops within biogeochemical cycles such as the carbon and nitrogen cycles (Sutton et al., 2011). Accounting for such multiple loops would require complex 'cradle to cradle' (McDonough and Braungart, 2002) approaches that may render even more complex the attribution of the environmental burden.

Recently approaches aiming at including natural resources, such as land and water, into the LCA methodology have been proposed (e.g. Garrigues et al., 2012; Pfister et al., 2009). First, it is crucial to consider soil quality in the environmental assessment of agriculture and forestry products given its major role for plant productivity. Refining soil quality impacts in LCA requires the estimation of soil properties, functions and processes (e.g., erosion, compaction) on a regional, or even local, basis given the large spatial variability of soil properties. Developing robust impact indicators for individual soil processes before attempting to aggregate them into a single indicator has been proposed as a way to make progress (Garrigues et al., 2012). Second, Pfister et al., (2009) presented a regionalised approach for assessing water-use related environmental impacts within existing LCA methods, using the example of worldwide cotton production. Their method is based on the watershed level, at which hydrological processes are connected, and on the use of the virtual water concept, which describes the amount of water that is lost by evapotranspiration during agricultural production in a given region. Not surprisingly, impacts from water consumption in the cotton industry were found to be highly variable ranging from high damage levels (e.g. 77% in Egypt) in dry areas to virtually no impact in areas with ample water resources (Pfister et al., 2009).

Bridging the gap between LCA and natural capital assessment can be seen as a key target for future research on the environmental sustainability of food systems. Such an approach focuses on inputs from nature rather than on emissions to nature. A step in this direction was proposed by Zhang et al., (2010b) through the development of an Ecologically Based LCA (Eco-LCA) that includes a large number of provisioning, regulating, and supporting ecosystem services as inputs to a life cycle model at the process or economy scale. Including an ecosystem service like grassland soil carbon sequestration (Soussana et al., 2007, 2010) into a LCA has led to significant changes in the estimate of GHG emissions from European livestock systems (Weiss et al., 2012). However, further progress is needed in order to regionalise ecosystem services prior to their inclusion in a LCA framework. Such a regionalisation has already been attempted through the development of a first atlas of ecosystem services at the scale of Europe (Maes et al., 2011).

Finally, spatial issues like indirect land use changes and temporal issues, such as irreversible environmental changes, are not yet tackled by LCA methodologies and this may be required to match the criteria of holistic environmental assessments.

4.2 Squaring the circle: diets, health and life cycle assessment

The DUALINE study (Esnouf et al., 2011) has analysed some of the research needs for assessing the environmental sustainability of diets and their impacts on health and food security. There are several dimensions to the nutritional impacts of diets (e.g. calories, proteins, essential amino-acids, micronutrients, etc…) (Sands et al., 2009) and, moreover, diet impacts on health may also vary according to a number of other drivers, including lifestyles and physical activity (HDHL JPI, 2012).

A well-known value established by the World Health Organization in LCA models, which has been applied to water use (Pfister et al., 2009) and air pollution (Sutton et al., 2011), could be used for the health effects of diets. The disability-adjusted life year is a value that expresses the number of years a person's healthy life will be shortened as a result of disease or premature death. Provided that an increased understanding of the consequences of diets for chronic diseases and for premature deaths can be developed for a given population (e.g. according to age and gender structure and to physical activity levels) in a given region, this index could be used to standardize the impacts of diets on health. If at all possible, given the many potential interactions across nutrients within diets, the individual contribution to this index of food products could also be calculated.

Yet, assessing the environmental impacts of aggregated diets from the life cycle of individual products is also challenging. First, a consumption based approach would be required whereas most life cycle studies start from the cradle of individual products. Second, fully assessing the complex product mix along the long supply chains characterising Western diets and food systems may seem out of reach. Finally, how to reconcile health oriented and nature capital oriented LCAs is still an open question.

5. Conclusions

Judicious use of the European land resources supported by agricultural sciences could adapt production to climate change, lower emissions, and eliminate net imports thus contributing to increasing global food security. Europe is well placed to address these issues since it recognizes the significance of global climate change and could therefore provide a space for change, testing an implementation of novel strategic concepts based on new bio-physical and socio-economic research (Soussana et al., 2012). Moreover, European cultures have developed a large variety of diets (e.g. 'Mediterranean' diet) and of traditional ways of food production and food consumption that may help in developing healthy diets for a healthy life. However, this will depend ultimately on consumer's choices that need to be better informed. Life cycle assessment can play an important role in designing products, orienting consumer's decisions and framing policy options, but it needs to overcome several difficulties some of which have been addressed here. Bridging nature capital, on the one hand, and health issues, on the other, with the assessment of the life cycle may lead to breakthroughs in the sustainability assessment of food systems.

6. References

Barnosky, A.D., Hadly, E.A., Bascompte, J. et al., 2012. Approaching a state shift in Earth's biosphere. Nature 486, 52-57.

Beddington, J.R., Asaduzzaman, M., Clark, M.E. et al., 2012. What next for agriculture after Durban? Science 335, 289-290.

Bellard, C., Bertelsmeier, C., Leadley, P., et al., 2012. Impacts of climate change on the future of biodiversity. Ecol. Lett. 15, 365-377.

Bindi, M., Olesen, J.E., 2011. The responses of agriculture in Europe to climate change. Reg. Envir. Change 11, 151-158.

Brug, J., 2008. Determinants of healthy eating: motivation, abilities and environmental opportunities. Family Practice 2008;25:i50-i55.

Ciais, P., Reichstein, M., Viovy et al., 2005. Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature 437, 529-533.

Coumou, D., Rahmstorf, S., 2012. A decade of weather extremes. Nature Climate Change 452, 1-5.

Davis, S.J., Caldeira, K. (2010) Consumption-based accounting of CO_2 emissions. Proc Natl Acad Sci USA 107, 5687-5692.

Esnouf, C., Russell, M. et Bricas, N. (eds) 2011. Dualine, Durabilité de l'alimentation face à de nouveaux enjeux. Questions pour la recherche. QUAE, Paris, 288 pp.

Eurostat, 2008. Food: from farm to fork statistics. Luxembourg: Office for Official Publications of the European Communities. 199 pp. ISBN 978-92-79-08736-3.

FACCE-JPI, 2011. Scientific Research Agenda of the 'Agriculture, Food Security and Climate Change' Joint Programming Research Initiative. See www.faccejpi.com.

Fereres, E., Orgaz, F., Gonzalez-Dugo, V. 2011. Reflections on food security under water scarcity. J. Exp. Bot. 62, 4079-4086.

Foley, J.A., Ramankutty, N., Brauman, K.A. et al., 2011. Solutions for a cultivated planet. Nature 478, 337-342.

Garrigues, E., Corson, M.S., Angers, D.A. et al., 2012. Ecol. Indicators 18, 434-442.

HDHL-JPI, 2012. Strategic Research Agenda of the Joint Programming Initiative 'A healthy diet for a healthy life'. See www.healthydietforhealthylife.eu

Intergovernmental Panel on Climate Change, IPCC, 2007. Climate change: impacts, adaptation and vulnerability, Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (eds. Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE). Cambridge University Press, Cambridge, New York.

Lenton, T., Held, H., Kriegler, E. et al., 2008. Tipping elements in the Earth's climate system. Proc Natl Acad Sci USA 105, 1786-1793.

Lobell, B., Schlenker, W., Costa-Roberts, J. 2011. Climate trends and global crop production since 1980. Science 333, 616-620.

Maes, J., Paracchini, M.L., Zulian, G., 2011. A European assessment of the provision of ecosystem services. European Commission, JRC, Institute for Environment and Sustainability, 88 pp. EUR 24750 EN – 2011 ISBN 978-92-79-19663-8. Luxembourg: Publications Office of the European Union.

McDonough, W., Braungart, M. 2002. Cradle to cradle: remaking the way we make things. Vintage publisher, New-York, 208 pp.

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC., 155 pp.

Nkonya, E., Gerber, N., Baumgartner, P. et al., 2011. The Economics of Land Degradation Toward an Integrated Global Assessment. Development Economics and Policy. Volume 66. Springer, Frankfurt am Main, Berlin, Bern, Bruxelles, New York, Oxford, Wien.

Olesen, J.E., Trnka, M., Kersebaum, K.C. et al., 2011. Impacts and adaptation of European crop production systems to climate change. European J. Agron. 34, 96-112.

Paillard, S., Treyer, S., Dorin, B., eds. 2011. Agrimonde: Scenarios and Challenges for Feeding the World in 2050. Éditions Quae, Versailles.

Pfister, S., Koehler, A., Hellweg, S., et al., 2009. Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environ. Sci. Technol., 43, 4098–4104

Rockström, J., Steffen, W., Noone, K. et al., 2009. A safe operating space for humanity. Nature 461, 472-475.

Sands, D.C., Morris, C.E., Dratz, E.A., 2009. Elevating optimal human nutrition to a central goal of plant breeding and production of plant-based foods. Plant Science 177, 377-389.

Searchinger, T., Heimlich, R., Houghton, R.A., et al., 2008. Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science, 319, 1238-1240.

Shafer, S., Walthall, C.L., Fransluebbers, S.R. et al., 2011. Editorial. emergence of the global research alliance on greenhouse gas mitigation. Carbon Management 2, 1-5.

Soussana, J.F., Allard, V., Pilegaard, K. et al., 2007. Full accounting of the greenhouse gas (CO_2, N_2O, CH_4) budget of nine European grassland sites Agric. Ecosyst. Envir. 121, 121-134.

Soussana, J.-F., Fereres, E., Long, S.P., Mohren, G.M.J., Pandya-Lorch, R., Peltonen-Sainio, P., Porter, J.R., Rosswall, T., von Braun, J.; 2012. A European science plan to sustainably increase food security under climate change. Global Change Biology. DOI: 10.1111/j.1365-2486.2012.02746.x

Soussana, J.F., Tallec, T., Blanfort, V., 2010 Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. Animal 4, 334-350.

Stehfest, E., Bouwman, L., van Vuuren, D.P. et al., 2009. Climate benefits of changing diet. Climatic Change 95:83–102
Stratton, R.J., 2007. Malnutrition: another health inequality? Proc. Nutr. Soc., 66:522-529.
Sutton, M.A., Oenema, O., Erisman, J.W. et al., 2011. Too much of a good thing. Nature, 472, 159-161.
TEEB, 2009. The economics of ecosystems and biodiversity for national and international policy makers – Summary: Responding to the Value of Nature. 47 p. United Nations Environment Program, ISBN 978-3-9813410-0-3.
Tubiello, FN, Soussana, J-F, Howden S.M., 2007. Crop and pasture response to climate change. Proc Natl Acad Sci USA 104:19686–19690.
Von Braun, J., 2008. The food crisis isn't over. Nature 456, 701.
Weiss, F., Leip, A., 2012. Greenhouse gas emissions from the EU livestock sector: A life cycle assessment carried out with the CA-PRI model. Agric. Ecosyst. & Envir., 149, 124-134.
Zhang, Y.I., Singh, S.H, and Bakshi, B.R., 2010a. Accounting for ecosystem services in life cycle assessment, Part I: A Critical Review. Environ. Sci. Technol. 44, 2232–2242
Zhang, Y.I., Singh, S.H, and Bakshi, B.R., 2010b Accounting for ecosystem services in life cycle assessment, Part II: Toward an Ecologically Based LCA Environ. Sci. Technol. 44, 2624–2631

Three perspectives on sustainable food security: efficiency, demand restraint, food system transformation. What role for LCA?

Tara Garnett

Food Climate Research Network, Environmental Change Institute, University of Oxford
E-mail: *taragarnett@fcrn.org.uk*

ABSTRACT

Achieving food system sustainability is a global priority but there are different views on how it might be achieved. Broadly three perspectives are emerging, defined here as: efficiency oriented, demand restraint and food system transformation. These reflect different conceptualisations on what is practically achievable, and what is desirable, underpinned by different values and ideologies about the role of technology, our relationship with nature and fundamentally what is meant by a 'good life.' This paper describes these emerging perspectives and explores their underlying values; highlights LCA's role in shaping these perspectives; and considers how LCA could be oriented to clarify thinking and advance policy-relevant knowledge. It argues that more work is needed to understand the values underlying different approaches to the food sustainability problem. This can shed light on why stakeholders disagree, where there are genuine misunderstandings, and where common ground is possible and ways forward agreed.

Keywords: climate change; food system; stakeholder perspectives; mitigation, life cycle analysis

1. Introduction

The food 'problem' has become a global obsession. How much and what kind of food is produced, how and by whom; how it is moved, processed, packaged and sold and with what impacts; who gets what and how much to eat, and at the expense of whom - and what the future might hold for all these variables; these questions are now the subject of measurement, analysis, critique and campaigning in research journals, policy documents, newspapers and television screens worldwide.

As such, the scale of the problems we face and their relationship with the food system are now well recognised and have been exhaustively described elsewhere (Godfray et al., 2010; Beddington et al., 2011 Foresight 2011). Put briefly: our global population is rapidly growing, urbanising and becoming wealthier, one consequence being that our dietary patterns are changing and our demand for land, resource and greenhouse gas (GHG) intensive foods, such as meat and dairy products, is on the increase. But while the demands we place on the earth may be growing, its available resources– of land, water, minerals - are finite. The difficulties presented by this demand-supply imbalance are compounded by changing environmental conditions which make food production increasingly difficult or unpredictable in many regions of the world; and production systems that not only undermine aspects of the ecosystem, such as biodiversity and water quality, upon which we ultimately depend, but also exacerbate zoonotic diseases and other risks that directly affect our health. Perhaps most starkly, inequities and distortions in how both the inputs to and outputs from food production are distributed have given rise to a paradoxical situation wherein 1.4 billion people world wide is overweight or obese, while 850 million lack sufficient calories and are undernourished (Swinburn et al., 2011; FAO 2011).The challenge is therefore to refashion the food system to deliver better nutritional outcomes at less environmental cost. But while this much is clear, the proposed solutions have been less coherently articulated and are certainly more contested. Stakeholders - across and within the food industry, civil society, policy makers and the research community - have often strikingly different views on what should be done.

It is argued here that broadly three perspectives are emerging in the debate on food system sustainability today. These in turn reflect different conceptualisations as to what is practically achievable given the variables of technological innovation, the functioning of the global economy and human motivations and behaviour - as well as different visions of what a sustainable food system actually looks like. These are in turn underpinned by different values and ideologies about the role of technology, our relationship with nature and fundamentally of what is meant by a 'good life.'

The purpose of this paper is threefold: to describe these three emerging perspectives on the nutrition-environment challenge and explore their underlying values; to highlight the role that life cycle analysis (LCA) has played in shaping these perspectives; and finally, to consider how LCA could be oriented and utilised in ways that clarify thinking and help advance policy-relevant knowledge in this field. It should be emphasised that, except at the extremes, these perspectives are not rigid and mutually exclusive. There will often be overlap between them and they are perhaps better viewed as ideological 'tendencies' rather than closed belief systems. The research community is represented across all three approaches.

2. Perspectives on achieving food system sustainability

The three approaches are defined as follows: efficiency oriented; demand restraint perspective; and food system transformation.

2.1. Efficiency

This is perhaps the dominant approach. Its advocates include governments and food industry actors such as agricultural input businesses, farming unions, manufacturers and retailers. These stakeholders see technological innovations and managerial changes as key to reducing environmental impacts and enhancing nutrition (ADAS et al., 2011). Agricultural efficiencies can be achieved by, for example, optimising the timing and quantity of fertilisers applied, using drip irrigation and other precision agriculture techniques and deploying technologies, such as anaerobic digestion, that recover utility from agricultural waste (manure, crop residues). Productivity increases in livestock can reduce emissions per unit of production, with approaches including: breeding for higher yields (of meat, milk or eggs), formulating feeds to maximise yields while minimising nitrogen or methane losses, and developing housing systems that optimise conditions for growth (Garnett 2011). Post harvest, emissions can be reduced through the refrigeration, manufacturing and transport technologies that are more energy efficient or based on renewable energy source. Waste is minimised through better inventory management, by modifying packaging and portion sizes and through other approaches that either prolong the shelf life of foods or help consumers reduce food waste in other ways (WRAP 2012).

While this perspective does not explicitly argue for this approach on moral grounds, it is nevertheless underpinned by a moral framework and set of values - a sense not only of what constitutes progress for humanity but an optimism that it can be achieved. Thus, a strong component of a good life is one in which more people will achieve the material comforts enjoyed by affluent consumers in the developing world today – but with less environmental impact. Using technology, the boundaries of our environmental limits can be extended to accommodate us and, provided the right market signals are in place, the global economy will enable both the material and environmental benefits to trickle down to all sectors of society.

LCA's influence on this approach has been critical. LCAs have helped companies identify environmental hotspots in the supply chain, reassess 'common sense' assumptions that (for example) locally sourced, or organic food has a lower environmental impact, and identify foods with the most significant impact. (Williams et al., 2006, Edwards-Jones et al., 2008; Sim et al., 2007; Defra 2008; FAO 2010b; Nemecek et al., 2012) As recognition of the contribution that agriculturally induced land use change makes to food's GHG emissions has grown, (Burney et al., 2010; FAO 2006) so the need to raise productivity so as to 'spare' land from further agricultural encroachment and associated CO2 release is emphasised.

However the LCA approach has not just informed this perspective but has also been influenced by it - or rather the use of LCA has helped strengthen the efficiency mindset. For example, it generally draws upon attributional LCAs where like product is compared with like and an alternative consumption possibility is not considered. Through this lens, Spanish lettuces are compared with, and found to be environmentally preferable to British lettuces grown under glass out of season. The desirability of consuming lettuces out of season is not considered, nor are comparisons made between lettuces and a more seasonal substitute food, such as cabbage. Similarly for meat and dairy; the carbon footprint of meat or milk production needs to be reduced, but demand for meat or milk per se is not questioned (IDF 2009). Notably, the metrics used are relative – impacts are expressed as GHGs per unit of production, not per absolute quantity produced or consumed. This choice of relative metrics suggests implicit endorsement of an economic model predicated on growth, and the primacy of consumer choice.

The problem of food waste is illustrative. Wasted food represents not only a waste of embedded GHG emissions and a threat to food security but, often, a financial inefficiency (Parfitt et al., 2010; WRAP 2009; UNEP 2009) - reducing waste saves money. Triple wins are possible. Implicit in this analysis of waste is the assumption that if less food were wasted this could have a role in addressing both GHGs and food security (WRAP 2011, Gustavsson et al., 2011, United Nations 2012). However there is less recognition of the porous boundary between food and other economic sectors, nor of the non-supply related causes of hunger. The supply of sufficient food on the market per se by no means guarantees food security (Sen 1981) an observation that motivates more systemic approaches to addressing the food challenge (below). As regards the interaction between food and other sectors, businesses may diversify into non-food products, all of which carry an associated carbon footprint – and indeed this is often an explicit goal for many companies (Tesco 2012). Consumers may use their saved money to buy other products or services that have an environmental impact, at least partially offsetting the emission reductions. In short, there is a rebound effect; relative improvements in efficiency may be partly or wholly offset by increased emissions in other sectors (Druckman et al., 2011). From a demand restraint perspective (discussed below) this underlines the need to address consumption per se, of which food behaviours are just one component (Jackson 2010).

For biodiversity, the priority is to avert further land use conversion to agriculture. Intellectual support is provided by model based research showing that, for a given agricultural yield and a defined land area, more intensive agricultural practices support greater biodiversity than less intensive 'wildlife friendly' production, since a dedicated block of land can be set aside purely for wildlife (Phalan et al., 2011). While the latter system may foster greater biodiversity on farm, more land is needed to produce a given quantity of food, and so land available for wilderness is reduced. Moreover the species supported in wildlife friendly farming are of lesser conservation interest than those found on virgin land. The strength of this approach is that it underlines the importance of addressing the knock on effects of different agricultural systems on land use elsewhere – in essence a 'spatial rebound effect'. It is increasingly used to consider the impacts of biofuels (Searchinger et al., 2007) but needs to be more comprehensively applied to understanding food production systems too.

However, while this approach delivers a theoretical insight into the relative benefits for biodiversity of different farming systems, less attention may be paid to the socio-economic context within which farming is practiced. Critics suggest that the economics of high yielding production create incentives to expand production into 'spared' land to increase profits further, thereby undermining the theoretical benefits (Fischer et al., 2011) an issue returned to below. Hence while, as noted this perspective implicitly endorses a growth-based economic model in other respects, it does not consider how the workings of the market might actually affect land sparing approaches in practice.

Regarding food security and nutrition, the two are considered somewhat separately within the efficiency mindset. The food security challenges is seen as one of increasing production to meet demand, with demand projections based on assumptions about income growth and its relationship with demand for certain foods, such as meat (Conforti 2011; Tilman et al., 2011; Foley et al., 2011). Less attention is paid to other dimensions of food security (access, utilisation, stability over time) (FAO 2008) or to the nutritional quality of food.

Moreover, the efficiency approach tends to shy away from saying what people should or should not be 'demanding' either for environmental reasons (as noted) or for health. Just as GHG efficiencies can be achieved using the insights from LCA to target environmental hotspots in the supply chain, so 'health efficiencies' are to be secured through product reformulations that deliver foods similar in taste to the originals but, for example, lower in fat, sugar or salt or with enhanced nutrition (prebiotics, omega 3 fatty acids). Supported by appropriate information, the consumer is then free to choose the healthier option without fundamentally needing to change their diet.

As regards animal products – criticised both on environmental and nutritional grounds by others (below) - the efficiency perspective is more positive, arguing, for instance, that milk delivers greater value per environmental impact than many other beverages. It points to research concluding that, for a combination of nutrients delivered per unit of GHGs emitted, low fat milk represents better nutritional 'value for climate' than orange juice, milk substitutes and others (Smedman et al., 2010; ICUSD 2010). Notably, low fat milk is chosen as the subject for analysis. While the removed fat might be incorporated into another product (a cake, say) the nutrient-climate impact of that product is not considered; a system expansion LCA approach might therefore yield different results. The emphasis, again, is on relative merits and there is no further analysis of what an 'optimal' level of consumption might look like - the minimum quantity of milk needed to deliver nutritional benefits without incurring excessive GHG cost through 'unnecessary' consumption, surplus to requirements.

In low income countries, the nutritional priority is to address micronutrient deficiencies. Food fortification (post harvest) and biofortification (breeding crops higher in target nutrients) are strategies that resonate with the efficiency perspective (HarvestPlus, undated), since they offer a technical way forward. Biofortification is considered particularly promising: while initial research investment costs are high, ex ante assessments suggest their cost effectiveness in addressing deficiencies is even greater. (Meenakshi et al., 2010). The approach, however has been criticised by the systems perspective as over-simplistic (below).

2.2. Demand restraint

For the efficiency mindset, the onus is on producers to develop appropriate techniques and strategies to reduce emissions; for the demand restraint perspective, the problem lies with the consumer and with the companies who promote unsustainable consumption patterns. The end point in the supply chain – the consumer – becomes the focus of concern. Central to this perspective lies the conviction that excessive consumption is a leading cause of the environmental crisis we face. Its vision of change is therefore an overtly moral one - it explicitly criticises the status quo rather than (for reasons that may also be morally motivated but less explicit) endorsing it.

The priority is to curb consumption of high impact foods. While in the 1990s the focus was particularly on foods high in 'food miles', as the findings of LCA research filtered through to the environmental community, combined with accusations of being 'anti poor' from poverty organisations, (MacGregor and Vorley), the locus of concern since shifted to animal products. The FAO's seminal Long Shadow report (FAO 2006) and numerous LCA-inspired scientific and NGO publications have highlighted the heavy burden that livestock place on land, water, biodiversity – and their contribution to GHG emissions (Pelletier and Tyedmers 2010; EC 2006; Weber and Matthews 2008; Stehfest et al., 2009; McAlpine et al., 2009).

Thus, where the efficiency perspective uses LCA to identify opportunities where technology and management can improve production efficiency to reduce the relative 'footprint' of existing consumption patterns, the demand restraint approach targets the consumption habits that ultimately drive production (they may also argue that the producers are seeking to generate the demand in the first place). It also investigates alternatives to the status quo, should consumption patterns change. Thus it draws not only upon attributional 'snapshot' LCAs that identify the most GHG intensive foods but also on those that adopt a 'what if?' approach to considering alternative scenarios. Increasingly, there is a focus on the opportunity cost and missed carbon sequestration potential arising from livestock production – arguably, if this land were not used for livestock it could regenerate naturally, or be used for other carbon sequestering purposes (Audsley et al., 2009; Schmidinger and Stehfest 2012). In other words, while the efficiency perspective looks at the implications for land use of different production systems (extensive versus intensive) the demand restraint perspective complements this by considering different consumption patterns.

For this perspective, the climate challenge is not separate from that of biodiversity or nutrition. They are all connected (CIWF 2010; Hamerschlag 2011). Livestock are not only dominant GHG contributors but also the main driver of land use change, deforestation and associated biodiversity loss; and they are associated with the rise in obesity and associated chronic diseases too (Popkin and Gordon-Larsen 2004; Sinha et al., 2009; FOE 2010; Pan et al., 2012). Other ethical and environmental concerns are added to the mix, such as water use and pollution, animal welfare (confined livestock in industrial scale units are a particular target) and labour conditions. Unlike the efficiency perspective, where technology holds the promise of expanding or overcoming environmental limits, for this perspective technology is at times problematic, limits are absolute and humans are, essentially, damaging. Nature is not to be managed by humans – rather humans need to 'get out of nature.' This, arguably, is a darker, more misanthropic view of our relationship with the natural world, although to an extent both it and the efficiency perspective view nature as other - to be 'spared' for conservation without human influence.

Regarding nutrition and food security, this perspective draws upon an emerging body of LCAs examining the relationship between environmental and nutritional goals. Studies, which tend to focus on developed countries, whose citizens typically enjoy access to a diverse range of plant foods, generally show that plant based diets can supply an adequate balance of nutrients at lower GHG 'cost' than meat-dominated diets. (WWF 2011; Carlsson-Kanyama and González 2009; Davis et al., 2010) In contrast with the efficiency perspective it focuses less on the positive nutrients found in animal products, such as calcium, iron and zinc, that are of critical importance to people on low income in the developing world, particularly children (Dror and Allen 2011). Hence, these perspectives draw upon different metrics to assess the nutrition-GHG relationship; one to endorse the status quo and the other to challenge it, offering a different vision of how we ought to consume.

Much is made, by restraint advocates, of the point that there is enough food in the world to feed everyone, in contrast with the 'more food' emphasis in the efficiency perspective. The challenge is therefore to address inequitable and resource intensive consumption patterns (Soil Association 2010), but a sophisticated analysis of how structural inequalities might be addressed is lacking. For example, feeding grains to livestock that could be more efficiently consumed directly by humans is identified as a 'waste,' (UNEP 2009) although some models find that due to global commodity price dynamics the effect on hunger reduction would be muted. Some argue that reductions in cereal prices would be partially offset by increases in prices of other foods, while lack of demand from the livestock sector would reduce farmers' incentives to grow the crops in the first place (Rosegrant et al., 1999; Msangi and Rosegrant 2001). Just as 'efficiency' may be undermined by the rebound effect (discussed above), so efficiency, redefined here by the restraint community, may not translate into substantially greater food availability and affordability for the poor. Both approaches are based on views about the way the world ought to work to ensure environmental benefits rather than the way it actually works, given current conditions.

Similarly, the use of the word 'efficiency' is often used differently by the two perspectives. So, while the efficiency perspective concludes that grain fed livestock in fact have a lower GHG footprint then those fed on grass and byproducts inedible to humans (Pelletier et al., 2010) and that such production is thus more efficient, the demand restraint approach uses LCA to highlight the GHG inefficiency of consuming grain indirectly (by eating meat) instead of eating grain products directly.

2.3 Food system transformation

Both the efficiency and the demand restraint perspectives focus on the individual – whether the individual farmer or company who produces, processes and distributes, or the consumer or company consuming or marketing the product. From a third perspective however, the problems stem not from individuals or even individual institutions but from the dynamic interactions among natural, technological, behavioural and economic systems. Within this perspective can be found a broad spectrum of opinions, some more radical than others in their analysis of the problems and their vision of the solutions. For all, though, the central argument here is that the problems we face are socio-economic rather than simply technical or a consequence of individual decisions. Environmental sustainability can only be achieved through structural change (IAAKSTD 2009; Foresight 2011; Oxfam 2011). At the ethical heart of this perspective lies an emphasis on social justice – on the moral necessity of developing systems of production and consumption that explicitly address the needs of poor people. It shares with the demand restraint perspective a moral explicitness, but the emphasis is on the responsibility of the system to deliver the desired objectives rather than on the individual. In common with the efficiency perspective it says little about what the limits to growth might be – growth is implicitly a good thing – but it questions the ability of the market, as it stands, to deliver benefits equitably.

How does this view engage with LCA methodologies and findings? By its very nature, an analysis of the problem that sees causes and outcomes as multiple and interacting will not accept the use of simple or single metrics to assess impacts or progress, since such metrics fail to capture relationships among the different components of the food system over space and time. This means that LCA has so far had limited resonance with this perspective, and indicators against which to measure progress have not yet been developed. How far they can be is indeed a matter for debate.

To illustrate: since agricultural production and its sustainability, for this perspective, is very clearly about more than just the production of a given commodity, a simple functional unit such as kg CO2 eq/kg product will be an inadequate measure of the system's success in delivering outputs relative to environmental impact. Outputs from the system include not just products with market or food energy value (wheat, maize, rice, milk) but may, depending on context, also include micronutrients (especially important in low income settings), fibres for roofing, cooking fuels (timber, manure), animal traction, cultural identity and status and – as for livestock in developing countries - portable liquid assets that can be sold in times of need, such as sickness, or to pay for school fees. For many smallholders, the system's resilience may also be a desired 'output.' Where access to formal insurance is lacking, this may be achieved by cultivating a diverse range of crops. While sub-optimal from a CO2 eq/ kg perspective, it can be essential for farmers who cannot afford to risk investing land and resources in producing just one high yielding commodity that may be vulnerable to pests or other shocks. More diverse multi-species systems (such as agroforestry) or mixed crop-livestock systems may represent more economically sustainable approaches. Whether they are also environmentally more sustainable depends on whether consumption is bounded by what is produced locally or whether shortfalls in supply are either met by external purchases, whose production will have had environmental effects elsewhere, or else lead to other changes (a move to the city to find jobs) that have GHG implications, positive or negative. In short, from the systems transformation perspective the environmental impacts of a production system cannot be assessed without understanding the socio-economic context and the way in which environmental impacts can be transferred, as it were, from one area or sector to another.

Mainstream LCA conclusions about different livestock systems are particularly open to question here since meat or milk, while clearly a desired output, is not the only one, nor is sheer volume of production the only goal. Table 1 illustrates how the choice of a different functional unit for a given livestock production system may alter conclusions as to its sustainability even when considering GHGs alone.

Alongside the challenge to mainstream LCA conclusions on livestock, a substantial subset from this perspective argues for more localised food production, despite the weight of LCA research concluding that transport distance does not correlate well with environmental impact, at least for GHG emissions. However the systems transformation perspective considers impacts that go beyond the atemporal, often very limited purview of much LCA, with its comparisons of like with like (Royal Gala apples with Royal Gala apples) at one point in time. It adopts a more dynamic perspective, considering the impacts of food production and consumption systems over time and within a more complex spatial and socio-economic framework.

Table 1. Different metrics for assessing the GHG intensity of livestock systems

Quantity based	**Comments**
kg CO_2 eq / kg product	Favours intensive monogastric production, and feed-based over grass based ruminant systems
kg CO_2 eq / kg protein, iron, calcium, fatty acid profile and so forth	Depends on nutrient: calcium and possibly iron may favour ruminants; grass-fed ruminants may have better Omega 3-6 ratios than cereal fed animals (Aurousseau et al., 2004; Demirel et al., 2006) protein as metric will favour intensive monogastrics. All may also need to be compared with provision of these nutrients by plant based sources.
Kg CO_2 eq / per nutrient density	This is a composite measure of various key nutrients in combination. Balance here is unclear – again needs to be compared against plant based alternatives
kg CO_2 eq / kg food and non food goods provided (leather, wool, feathers, dung, traction)	Variable; on balance likely to favour ruminants in mixed systems
Area based	**Comments**
kg CO_2 eq per area of land	Emissions lower for extensive systems and for monogastrics
kg CO_2 eq per area of prime arable land required	Emissions lower for extensive systems, both ruminant and monogastric
Resources based	**Comments**
kg CO_2 eq avoided through use of byproducts or poor quality land to rear livestock; approach quantifies the GHG and land opportunity cost of needing to obtain an equivalent quantity of nutrition from elsewhere	Favours extensive systems and particularly landless household pig and poultry reliant on scraps
kg edible output per specified quantity of ecosystem services provided on farmed land	Depends on which ecosystem services are valued but may favours extensive ruminant systems
kg edible output per given area off the direct farmland eg. on land 'spared' for conservation or biomass production	Favours intensive systems, especially monogastrics
Resilience based	
Adaptability to climate and environmental change	May favour local breeds

Adapted from Garnett T (2011). Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? Food Policy 36, S23-S32.

Thus, it asks how might we consider transport's GHG impacts once the need to recoup investment in supporting infrastructure is taken into account, by increasing the throughflow of commodities? Moreover, transport is seen as inextricably linked with other energy using aspects of the food supply chain, including refrigeration, packaging, processing and information technology. The transport of most foods inherently depends on refrigeration, while refrigeration makes possible longer supply chains. Thus, the availability of one technology enables heavier use of the other, the consequence being a ratcheting up of energy dependence.

So far this reassessment of transport is still broadly within the LCA mainstream in so far as it urges the need for systems expansion. LCA methodology has also been advanced through models that consider the marginal impact of changes in consumption on production and land use within other regions, mediated by trade (Kløverpris et al., 2010). However this perspective goes further by considering the porous interface between the technical and human behavioural domains. For example: how do efficiencies in the supply chain increase the supply and affordability of certain foods which ultimately foster new behavioural norms and habits? How does wider provision of the environmentally 'efficient' option (an imported Spanish lettuce, or less GHG intensive meat) create behavioural 'lock in,' entrenching patterns of consumption that are dependent on this nexus of interdependent, energy using technologies - refrigeration-transport and IT nexus? A sense of movement in time and space is implicit in the analysis, while attention to the effects that technological developments have on human habits, assumptions and practices prompt questions about how the sustainability of different technical approaches might be assessed.

There is perhaps a deeper challenge to LCA within this perspective, which brings in the concept of human agency and moral responsibility. Systems of production, distribution and consumption are viewed in terms of the power relationships between individuals and between countries, of cultural identity and ultimately about what constitutes progress. Instances of this approach can be found in local food initiatives such as the Fife Diet in the UK (Fife Diet, undated) in overtly political 'peasant' movements such as La Via Campesina who call for 'food sovereignty' and who oppose large scale corporations (La Via Campesina, 2011) and among many within the organic movement. While such analyses cast light on the inequities associated with current systems of production and consumption, and their damaging consequences for health and human wellbeing, the corollary assumption – that smallscale, localised production systems are necessarily more sustainable – is nevertheless a value judgment. For example, smallholder adoption of agroforestry practices may or may not halt deforestation, depending on the prevailing socio-economic conditions. These conditions may include the presence or absence of land use rights, labour or forest protection legislation (Schroth et al., 2004). In both systems –large commercial and small scale subsistence - the governance framework which shapes production and consumption will influence the extent to which undesirable direct and indirect spatial (land use change) and consumption rebound effects ensue. Thus, while emphasis on improving rural livelihoods at one level reflects pragmatic recognition of how millions of people live today, for many within this perspective agrarianism is perhaps synonymous with the good life. Both wellbeing and sustainability are achieved through the harmonious integration of humans with nature through rural living– unlike the perspectives of demand restraint with its emphasis on 'humans out!' or of efficiency with its emphasis on technology to expand limits while saving space for a separate wilderness.

As regards nutrition, the system transformation perspective, as for demand restraint, sees the nutritional, and environmental challenges as interconnected and to be addressed holistically. 'Food security' is defined to include not just the 'technical' supply of nutrients but also the other key dimensions identified by the Food and Agriculture Organisation, which include accessibility (incorporating affordability), utilisation and stability over time (FAO 2008). Often an argument is made for local, diverse agricultural systems producing indigenous crops and animal breeds. These are seen as better able to provide the full range of micronutrients needed for good health than global supply chains which produce and distribute a simplified range of processed, energy- and fat-dense commodities (FAO 2010a; Toledo and Burlingame 2006). Nutritional and agricultural diversity are thus seen as connected, and essential. Fortification and biofortification strategies represent a second best strategy in that they merely 'top up' inherently inadequate diets and food systems. While they may have a part to play, these techniques must be situated within a broader food-based approach that emphasises greater nutritional and agricultural diversity within the production system (Johns and Eyzaguirre 2007).

There is clearly a need for studies that consider the implications for GHG emissions, land use, biodiversity and nutrition of different agricultural systems involving various combinations of crops, livestock and innovations such as biofortification. Such an approach would need to go beyond a simple consideration of the GHG emissions associated with different consumption patterns (such as WWF 2011, Carlsson-Kanyama and González 2009; Davis et al., 2010) since it explicitly views health and environmental sustainability as outcomes of a linked system of production-consumption rather than just of consumption. But even these approaches will be limited since they may not be able to capture the economic value of different production systems and their translation into health outcomes. For example, the nutritional contribution that livestock provide for people in low-income countries is not necessarily a simple relationship along the lines of "more production equals better nutrition." The outcomes are mediated through impacts of livestock production on household incomes and the knock on effects of income generation on health generally – for example on people's ability to pay for health care or education, both of which have independent positive effects on health. In other words, the system transformation approach recognises that a more complex understanding of health-sustainability linkages is needed (Hawkes and Ruel 2006). Whether LCA or LCA-type analyses are able to capture and quantify these dynamic interactions, however, is open to question.

3. Discussion

What do we mean by good nutrition? By biodiversity? By limiting climate change? What are our ethical boundaries - livelihoods, labour standards, animal welfare, other species? These questions go far beyond LCA, but LCA researchers need to be mindful that this is the context within which they frame their research.

This paper has broadly characterised three emerging perspectives in the discourse on food system sustainability. The vision underlying the efficiency perspective is to take current development goals – greater incomes for all, more material consumption, more food – and to use technology to deliver these goals with less environmental impact. At one level it is profoundly pragmatic: it is 'human nature' to want more; the way the world and the market operates cannot or should not radically be changed (past experiments, such as socialism, have failed); the challenge, therefore, is to improve the status quo. This perspective helps drive the development of technologies and practices that achieve greater efficiencies in production and enhance the nutritional qualities of foods that are currently marketed and becoming more prevalent. Its strength lies in careful measurement, in identifying where reductions can be achieved, and in highlighting the effects of different production approaches on land use elsewhere. It has also challenged 'common sense' assumptions about the impacts of particular stages in the supply chain, or certain production practices. Fundamentally, however, it fails to engage with the problem of absolute limits. It implicitly assumes that technological developments and the market as it operates today will ultimately be successful in decoupling production (and GDP growth) from negative environmental and health impacts and it accepts, sometimes even endorses, current trends in consumption. These assumptions are open to challenge (Jackson 2010). It also pays insufficient attention to the porous interface between the technical and the socio-economic domains and the complex relationship between technological developments and behavioural change.

In order to strengthen this perspective, some questions it needs to address include: how can LCA get to grips with assessing sustainability over different temporal and spatial scales? How can it better understand and quantify the rebound effect, including 'leakage' from the food system into other economic areas? If land sparing approaches makes theoretical sense, then what governance framework is needed so that profit considerations do not undermine the land sparing effect?

The demand restraint approach positions consumption as the cause of our environmental crisis. Environmental limits are absolute: rather than 'tinkering at the edges' we need to shrink our footprint by consuming less or reproducing less. While this perspective includes a strong social justice element (contraction and convergence) essentially its vision of the good life is an ascetic one – living better by consuming less. As such it has resonances with much religious thinking, or rather, environmentalism fills the gap that for many, can no longer be filled by religion (Dunlap 2006).

The value of this approach lies in its questioning of the sufficiency of relative, rather than absolute limits; in highlighting the critical influence of consumption on the overall burden of impacts; and by providing a framework for seeing the connections between problems and addressing them together. Livestock are seen as a convergence issue for a range of interconnected sustainability concerns, to be addressed together (through changing consumption) rather than as stand alone issues. However this perspective can suffer from a lack of nuance around livestock and their positive dietary as well as environmental contributions, perhaps reflecting this perspective's developed world origins and focus. Its overtly moral vision can be offputting to some who do not share it. Moreover, a robust account of how behaviour might be changed is lacking. This constitutes a priority research challenge. Other critical research questions for this perspective include: is it possible to define a minimum level of meat and dairy consumption such that the micronutrient value of the nutrition package are not outweighed by GHGs resulting from delivery of 'wasted' nutrients (that is, those that are surplus to requirements)? There is also a need for more LCA based assessments of what constitutes a culturally acceptable, healthy sustainable diet in different low income and emerging economies.

The food system transformation approach is perhaps the most political in so far as it sees human behaviours as the outcomes of social structures, rather than just conscious individual decisions. It is the structure that needs to be changed rather than the individual, and this requires understanding of the dynamic interactions among its social, economic and environmental components over time and space. Its rejection of clear demarcations between the environmental, technical and economic domains represent an important challenge to much LCA thinking. Its vision of a good life shares some of the redistributive morality of the demand restraint perspective but it is more optimistic about the role of humans in the natural world – integration between humans and nature is possible and can be achieved, among other things, through a greater focus on social justice. However, some within this perspective may romanticise the small scale and local, failing to subject these systems to critical scrutiny as they do in the case of commercial systems.

Perhaps a central problem with this approach is that, while it is good at identifying the complex nature of the food sustainability challenge, this very complexity presents an obstacle to the development of specific recommendations as to the way forward. To add rigour to this perspective it is worth exploring whether methodologies and metrics can be developed that capture not just environmental impacts over time and space but also the socio-economic consequences of different production approaches, that in turn give rise to environmental impacts - and vice versa. Assessments need also to consider ways of measuring outputs that are not only multiple but not sometimes intangible. There is a need too for approaches that consider the interactions between different components of the system at different scales, and across scales (that is, the relationship between local and global food systems). A few of these questions may be addressed by further developments in social and environmental LCA methodologies but most will require interdisciplinary research, linking LCA and other disciplines.

4. Conclusion

While the three perspectives: efficiency, demand restraint, and food system transformation, have been presented here as separate world views held by different stakeholders, clearly they are not. They are 'tendencies' rather than stand alone ideologies (at least for most people) and individual people or institutions may adopt any one, or all three of these approaches at different times and to different degrees. Each perspective has its strengths as well as its weaknesses and inconsistencies and, perhaps predictably, the reality is that a composite approach to tackling the food sustainability problem, drawing upon all three perspectives, will be needed. However integrating them into a workable way forward requires greater understanding of the values that underlie the individual perspectives and that give rise to differences of opinion among stakeholders.

Values matter, and they cannot be ignored if progress is to be made. Everybody wants 'sustainability' and an end to hunger – but not everyone has the same vision of what the solution – the good life – might look like. The ethical perspectives people bring to the food-sustainability problem influence both their use of the evidence and the solutions they propose – and these often lead to stakeholders arguing at cross-purposes, the result being conflict, or inaction. Greater understanding of what underlies the different approaches to the food sustainability problem can help shed light on why stakeholders disagree, where there are genuine misunderstandings, and where common ground among them may be possible and ways forward agreed. (Hulme 2009; Garnett and Godfray 2012).

5. References

ADAS et al., 2011. Meeting the Challenge: Agriculture Industry GHG Action Plan Delivery of Phase I: 2010 - 2012 04 April 2011, ADAS, AEA Agricultural Engineering Association, AHDB Agriculture and Horticulture Development Board, AIC Agriculture Industries Confederation, CLA Country Land and Business Association, Farming Futures, FWAG Farm Wildlife Advisory Group, LEAF Linking Environment And Farming, NFU National Farmers Union, NIAB/TAG National Institute of Agricultural Botany/The Arable Group, ORC Elm Farm Organic Research Centre, RASE Royal Agricultural Society of England

IAAKSTD, Agriculture at a Crossroads, 2009. The Global Report. International Assessment of Agricultural Knowledge, Science and Technology for Development, Island Press

Audsley, E., Chatterton, J., Graves, A., Morris, J., Murphy-Bokern, D., Pearn, K., Sandars, D., Williams, A., 2009. Food, land and greenhouse gases. The effect of changes in UK food consumption on land requirements and greenhouse gas emissions. The Committee on Climate Change.

Aurousseau, B., Bauchart, D., Calichon, E., Micol, D., Priolo, A., 2004. Effect of grass or concentrate feeding systems and rate of growth on triglyceride and phospholipid and their fatty acids in the M. longissimus thoracis of lambs. Meat Sci. 66, 531–541.

Beddington, J., Asaduzzaman, M., Fernandez, A., Clark, M., Guillou, M., Jahn, M., Erda, L., Mamo, T., Van Bo, N., Nobre, C.A., Scholes, R., Sharma, R., Wakhungu, J., 2011. Achieving food security in the face of climate change: Summary for policy makers from the Commission on Sustainable Agriculture and Climate Change. CGIAR Research Program on Climate Change, Agriculture and Food Security CCAFS. Copenhagen, Denmark.

Burney, J., Davis, S.J., Lobell, D.B., 2010. Greenhouse gas mitigation by agricultural intensification, PNAS, 107, 12052–12057

Carlsson-Kanyama, A., González, A.D., 2009. Potential contributions of food consumption patterns to climate change. Am. J. Clin. Nutr. 89 suppl:1704S–9S

CIWF, 2009. Beyond Factory Farming: Sustainable Solutions for Animals,People and the Planet, Compassion in World Farming, Godalming, Surrey, UK

Conforti, P., 2011. Looking Ahead in World Food and Agriculture: Perspectives to 2050, Rome: Food and Agriculture Organization, Rome.

Davis, J., Sonesson, U., Baumgartner, D.U., Nemecek, T., 2010. Environmental impact of four meals with different protein sources: Case studies in Spain and Sweden. Food Res. Int. 43, 1874–1884.

Defra, 2008. Comparative Life Cycle Assessment of Food Commodities Procured for UK Consumption through a Diversity of Supply Chains, research undertaken for Defra by AEA, ADAS, Ed Moorhouse, Paul Watkiss Associates, AHDBM, Marintek, Defra project FO0103

Demirel, G., Ozpinar, H., Nazli, B., Keser, O., 2006. Fatty acids of lamb meat from two breeds fed different forage: concentrate ratio. Meat Sci. 72, 229–235.

Dror, D.K., Allen, L.H., 2011. The importance of milk and other animal-source foods for children in low-income countries. Food Nutr. Bull. 32, 227-24317

Druckman, A., Chitnis, M., Sorrell, S., Jackson, T., 2011. Missing carbon reductions? Exploring rebound and backfire effects in UK households. Energ. Policy 39, 3572–3581

Dunlap, T.R., 2006. Environmentalism: A secular faith. Environ. Valu. 15, 321–30

Edwards-Jones, G., Milà L., Canals, I., Hounsome, N., Truninger, M., Koerber, G., Hounsome, B., Cross, P., York, E.H., Hospido, A., Plassmann, K., Harris, I.M., Edwards, R.T., Day, G.A.S., Tomos, A.D., Cowell, S.J., Jones, D.L., 2008. Testing the assertion that 'local food is best': the challenges of an evidence-based approach, Trends in Food Science & Technology 19: 265–274

EC, 2006. Environmental impact of products EIPRO: Analysis of the life cycle environmental impacts related to the total final consumption of the EU25, European Commission Technical Report EUR 22284 EN

FAO, 2006. Livestock's Long Shadow, Food and Agriculture Organisation, Rome.

FAO, 2008. An introduction to the basic concepts of food security, FAO, 2008 EC – FAO Food Security Programme

FAO, 2010a. Final document: International Scientific Symposium Biodiversity and Sustainable Diets: United against Hunger. 3-5 November 2010, Food and Agriculture Organisation, Rome

FAO, 2010b. Greenhouse Gas Emissions from the Dairy Sector: A Life Cycle Assessment, Food and Agriculture Organisation, Rome, 2010.

FAO, 2011. The State of Food Insecurity in the World, Food and Agriculture Organisation, Rome Fife Diet, undated http://www.fifediet.co.uk/about-us/

Fischer, J., Batáry, P., Bawa, K.S., Brussaard, L., Chappell, M.J., Clough, Y., Daily, G.C., Dorrough, J., Hartel, T., Jackson, L.E., Klein, A.M., Kremen, C., Kuemmerle, T., Lindenmayer, D.B., Mooney, H.A., Perfecto, I., Philpott, S.M., Tscharntke, T., Vandermeer, J., Wanger, T.C., Von Wehrden H., 2011. Conservation: Limits of Land Sparing. Science, 334, 594

FOE, 2010. Healthy Planet Eating: How lower meat diets can save lives and the planet, Friends of the Earth, UK.

Foley, J.A., Rarmankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnstone, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Seibert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342

Foresight, 2011. The Future of Food and Farming. Final Project Report. The Government Office for Science, London.

Garnett, T. 2011. Where are the best opportunities for reducing greenhouse gas emissions in the food system including the food chain? Food Policy 36, S23–S32

Garnett, T., Godfray, C., 2012. Sustainable intensification in agriculture. Navigating a course through competing food system priorities. Food Climate Research Network and the Oxford Martin Programme on the Future of Food, University of Oxford, UK

Godfray, H.C. J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M., Toulmin, C., 2010. Food Security: The Challenge of Feeding 9 Billion People. Science, 327

Gustavsson, J., Cederberg, C., Sonesson, U., 2011. Global food losses and food waste. Rome: Food and Agriculture. Organization of the United Nations.

Hamerschlag K., 2011. Meat Eater's Guide to Climate Change and Health, Environmental Working Group, Washington DC, United States

Hawkes, C., Ruel, M., 2006. Understanding the links between agriculture and health, IFPRI, Washington DC. Harvest Plus http://www.harvestplus.org/, ndated

Hulme, M., 2009. Why We Disagree About Climate Change: Understanding Controversy, Inaction and Opportunity. Cambridge University Press

ICUSD, 2010. U.S. Dairy Sustainability Commitment Progress Report, Innovation Center for US Dairy

IDF, 2009. A global dairy agenda for action – climate change, International Dairy Federation, World Dairy Summit, Berlin

Jackson, T., 2010. Prosperity without Growth: Economics for a finite planet. Earthscan.

Johns, T., Eyzaguirre, P.B., 2007. Biofortification, biodiversity and diet: A search for complementary applications against poverty and malnutrition, Food Policy 32 1–24J H Kløverpris,K Baltzer and P H Nielsen 2010. Life cycle inventory modelling of land use induced by crop consumption Part 2: Example of wheat consumption in Brazil, China, Denmark and the USA. Int. J. Life Cycle Assess., 15, 90–103

La Via Campesina, 2011. http://viacampesina.org/en/index.php?option=com_content&view=category&layout=blog&id=27&Itemid=44

MacGregor, J., Vorley, B., 2006. "Fair miles"? The concept of "food miles" through a sustainable development lens. International Institute for Environment and Development, London.

McAlpine, C.A., Etter, A., Fearnside, P.M., Seabrook, L., Laurance, W.F., 2009. Increasing world consumption of beef as a driver of regional and global change: A call for policy action based on evidence from Queensland Australia, Colombia and Brazil. Global Environ. Chang. 19, 21–33

Meenakshi, J.V., Johnson, N.L., Manyong, V.M., Degroote, H., Javelosa, J., Yanggen, D., Naher, F., Gonzalez, C., Garcia, J., Meng, E., 2010. How Cost-Effective is Biofortification in Combating Micronutrient Malnutrition? An Ex ante Assessment. World Dev., 38, 64-75

Msangi, S., Rosegrant, M.W., 2011. Feeding the Future's Changing Diets Implications for Agriculture Markets, Nutrition, and Policy, IFPRI 2020 Conference: Leveraging Agriculture for Improving Nutrition and Health February 10-12, New Delhi, India

Nemecek, T., Weiler, K., Plassmann, K., Schnetzer, J., Gaillard, G., Jefferies, D., García–Suárez, T., King, H., Milà i Canals, L., 2012. Estimation of the variability in global warming potential of global crop production using a modular extrapolation approach MEXALCA. J. Cleaner Prod. 31, 106-117

Oxfam 2011. Growing a Better Future: Food justice in a resource-constrained world, Oxfam International

Pan, A., Sun, Q., Bernstein, A.M., Schulze, M.B., Manson, J., Stampfer, M.J., Willett, W.C., Hu, F.B., 2012. Red Meat Consumption and Mortality Results From 2 Prospective Cohort Studies. Arch. Intern. Med. doi:10.1001/archinternmed.2011.2287

Parfitt, J., Barthel, M., Macnaughton, S., 2010. Food waste within food supply chains: quantification and potential for change to 2050. Philos. T. R. Soc. B. 365, 3065–3081.

Pelletier, N., Tyedmers, P., 2010. Forecasting potential global environmental costs of livestock production 2000–2050. PNAS published ahead of print

Pelletier, N., Pirog, R., Rasmussen, R., 2010. "Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States". Agric. Syst. 103, 380–389

Phalan, P., Onial, M., Balmford, A., Green, R.E., 2011. Reconciling food production and biodiversity conservation: Land sharing and land sparing compared. Science 333, 1289

Popkin, B.M., Gordon-Larsen, P., 2004. The nutrition transition: worldwide obesity dynamics and their determinants International. J. Obesity. 28, S2–S9

Rosegrant, M.W., Leach, N., Gerpacio, R.V., 1999. Plenary Lecture: Alternative futures for world cereal and meat consumption in Meat or wheat for the next millennium?' P. Nutr. Soc. 58, 219–234

Schmidinger, K, Stehfest, E., 2012. Including CO2 implications of land occupation in LCAs—method and example for livestock products, Int. J. Life Cycle Ass., DOI: 10.1007/s11367-012-0434-7

Schroth, G., da Fonseca, G.A.B., Harvey, C.A., Gascon, C., Vasconcelos, H.L., Izack, A.M., 2004 Agroforestry and Biodiversity Conservation in Tropical Landscapes, Island Press, Washington DC.

Searchinger, T., Heimlich, R, Houghton, R.A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.H., 2008. Greenhouse Gases Through Emissions from Use of U.S. Croplands for Biofuels Increases Land use Change. Science. 319, 1238.

Sen, A., 1981. Poverty and Famines Oxford University Press, Oxford.

Sim, S, Barry, M., Clift, R., Cowell, S.J., 2007. The Relative Importance of Transport in Determining an Appropriate Sustainability Strategy for Food Sourcing. Int. J. Life Cycle Ass. 12, 422–431

Sinha, R ., Cross, A.J., Graubard, B.I., Leitzmann, M.F., Schatzkin, A., 2009. Meat Intake and Mortality: A Prospective Study of Over Half a Million People. Arch. Intern. Med. 1696, 562-571

Smedman, A., Lindmark-Månsson, H., Drewnoski, A., Modin Edman, A.K., 2010. Nutrient density of beverages in relation to climate impact, Food Nutr. Res. 54, 5170

Soil Association 2010, Telling porkies: The big fat lie about doubling food production. Soil Association, Bristol.

Stehfest, E., Bouwman, L., van Vuuren, D.P., den Elzen, M.G.J., Eickhout, B., Kabat, P., 2009. Climate benefits of changing diet. Climatic Change. 95, 1-2

Swinburn, B A, Sacks, G, Hall, K.D., McPherson, K., Finegood, D.T., Moodie, M.L., Gortmaker, S.L., 2011. The global obesity pandemic: shaped by global drivers and local environments. The Lancet, 378, 804 – 814.

Tesco, 2012. Annual Report 2012, Tesco.

Tilman, D., Balzer, C., Hill, J., Befort, B.L., 2011. Global food demand and the sustainable intensification of agriculture. Proceedings of the National Academy of Sciences of the United States of America. 108, 20260-20264

Toledo, A., Burlingame, B., 2006. Biodiversity and nutrition: A common path toward global food security and sustainable development. J. Food Comp. Anal. 19, 477–483

UNEP, 2009. The environmental food crisis: The environment's role in averting future food crises, Nairobi.

United Nations, 2012. FAO, partners, urge greater push to reduce food losses and waste, 13 June 2012, Food and Agriculture Organisation, Rome http://un-foodsecurity.org/node/1345

Weber, C., Matthews, H., 2008. Food-Miles and the Relative Climate Impacts of Food Choices in the United States. Environ. Sci. Technol. 4210, 3508-3513.

Williams, A.G., Audsley, E., Sandars, D.L. ,2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Bedford: Cranfield University and Defra

WRAP, 2011. Environmental Audit Committee: Written evidence submitted by WRAP, March 2011http://www.publications.parliament.uk/pa/cm201012/cmselect/cmenvaud/879/879vw20.htm

WRAP, 2012. Courtauld Commitment 2 Voluntary agreement 2010 - 2012 Signatory case studies and quotes, Waste Resources Action Programme, 2012

WRAP, 2009. Household Food and Drink Waste in the UK. Report prepared by WRAP. Banbury, UK.

WWF, 2011. Relative contribution of different food groups to diet related GHG emissions. Livewell report, WWF, Godalming, UK.

Challenges for LCA in the agri-food sector, perspectives from Thailand and Southeast Asia

Shabbir H. Gheewala[1,2]

[1] The Joint Graduate School of Energy and Environment, King Mongkut's University of Technology Thonburi, Bangkok, Thailand
[2] Center for Energy Technology and Environment, Ministry of Education, Thailand
E-mail: *shabbir_g@jgsee.kmutt.ac.th*

ABSTRACT

Agriculture and agri-food industries are very important sectors in many Asian countries, especially in Southeast Asia. Thailand, an emerging economy, is a large producer and exporter of agri-food products. Life cycle assessment (LCA) has been used increasingly in the agriculture sector though much still remains to be done. This paper outlines the brief historical development of LCA in Thailand focusing particularly on the agri-food sector. The development process and status are described pointing out also the areas that need further development. Challenges for database development as well as methodology development are identified. The status of the development of LCA in the agri-food sector in some other countries in the region is also summarised along with the development of a regional initiative in this area.

Keywords: Agri-food, Database, LCA, Southeast Asia, Thailand

1. Introduction

Life cycle assessment (LCA) has been introduced to Thailand about 15 years ago (Lohsomboon, 2002); though efforts towards its popularisation have gathered momentum in the last 7-8 years. In the interim period, LCA has been taught in several universities in Thailand and numerous papers and research reports have been published based on LCA studies in various sectors, particularly agriculture and energy (e.g. Mungkung et al., 2006; Lebel et al., 2010, Nguyen and Gheewala, 2008a,b; Phumpradab et al., 2009; Suwanit and Gheewala, 2011; Gheewala, 2011). Considering the large agricultural base of Thailand, the focus on studies based on agriculture in not surprising. However, the data-intensive nature of LCAs and the lack of industry's understanding on the utility of such studies has been a barrier in large-scale implementation. The recent popular appeal of product carbon footprinting has initiated a spate of studies and has curiously even encouraged industries to take a fresh look at full LCAs (Mungkung et al., 2012).

The idea of life cycle concept was actually incorporated first in the Thai Green Label (Figure 1a) initiated by the Thailand Business Council for Sustainable Development (TBCSD) in 1993 and formally launched in 1994 by the Thailand Environment Institute (TEI) in association with the Ministry of Industry (http://www.tei.or.th/greenlabel/index.html). Though this is a Type I label which does not require conducting a full LCA, the criteria are based on the life cycle idea. The first life cycle inventory of electricity was conducted in 1997 by TEI and a graduate level course in LCA initiated in 1999. The Thai LCA Network linking LCA practitioners was initiated in 2001. This paper briefly describes the activities in Thailand and the region vis-à-vis LCA in the agri-food sector

2. LCA in the agri-food sector in Thailand

Activities on LCA in Thailand during the last decade have largely been dominated by individual research projects and graduate studies. One of the first sectors that received attention was shrimp which was studied first as a doctoral thesis and then as a national level project (Mungkung et al., 2006; Lebel et al., 2010). Several doctoral theses were also conducted on biofuels that included agriculture as a very important part (e.g. Nguyen, 2007; Yutthitham, 2009; Silalertruksa, 2010). Some of the studies also looked particularly at food versus fuel issues because of the competition for feedstock and resources (Silalertruksa and Gheewala, 2010, 2011, 2012). Research projects were also conducted involving industries, particularly with the popularisation of product carbon footprint (Mungkung et al., 2010; Mungkung et al., 2012). More recently, studies have been initiated on water footprint which is a very important issue for agriculture (Nilsalab et al., 2012). Once again these studies cover both work on the adaptation of methodology to the Thai context as well as research projects with industrial partnership.

National level activities took an upward turn particularly with the initiation of the national life cycle inventory (LCI) database in 2005 (http://www.thailcidatabase.net/). National database development was broadly focused on six sectors: (1) Energy, utilities and transportation, (2) Industrial materials, (3) Agriculture, (4) Commodity chemicals, (5) Building and construction materials and (6) Recycling and waste management. This activity is being led and coordinated by the National Metal and Materials Technology Center (MTEC) under the National Science and Technology Development Agency (NSTDA), Ministry of Science and Technology in collaboration with the Federation of Thai Industries, Department of Industrial Works (Ministry of Industry), TEI and Thailand Research Fund. Many universities and research institutes also participated in the effort for building up the database. A special working group dedicated to agriculture and agro-products is developing about 175 datasets based on collating information from the existing research studies mentioned earlier (and information from the Department of Agriculture and Office of Agricultural Economics) as well as direct interviews with companies.

Another national level effort has been the development of product carbon footprinting and labelling (Figure 1b) in Thailand which was initiated in 2009 (http://www.tgo.or.th/english/). As part of the guidelines development for product carbon footprinting, 24 pilot projects were conducted. Many of the products were agriculture-based, for example, rice, chicken, pineapple and tuna as well as their products. To date (as of May 2012), 487 products (from 120 companies) have been labelled out of which almost 300 are from the agri-food sector (Figure 2). The agri-food CFP labelled products cover a wide range such as chicken meat, chicken seasoning, jasmine rice, canned food, fruit juice, beverage, animal feed, instant rice vermicelli, food and beverage packaging, etc.

(a) Green label (b) Product carbon footprint label

Figure 1. Environmental labels in Thailand based on life cycle concept

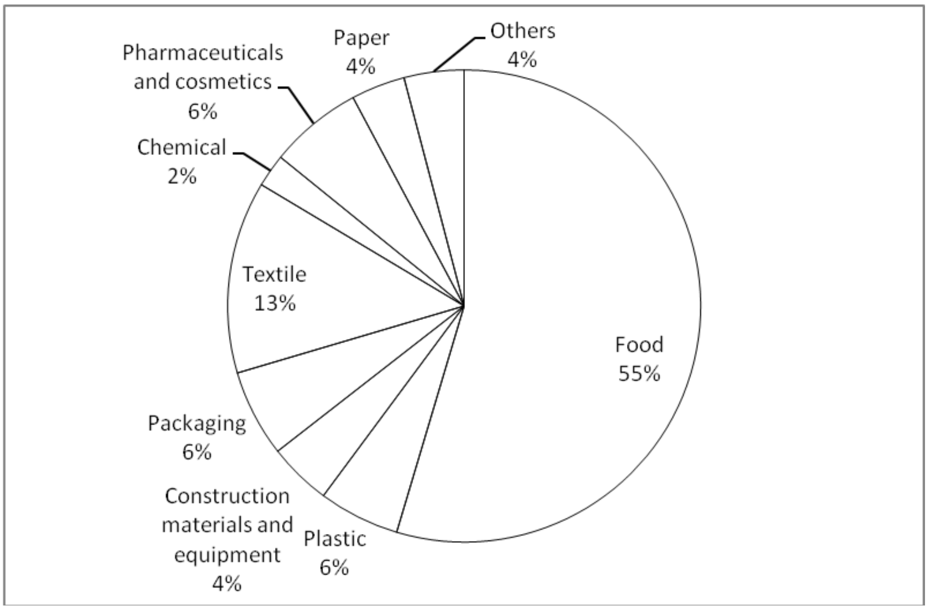


Figure 2. Percentages of carbon labelled products in Thailand based on industry type

The demand for carbon labelling of products has been increasing ever since the launch in 2009. Many small and medium enterprises have expressed their fervent interest to participate in this scheme though they face barriers due to lack of technical expertise as well as finances to hire consultants. To facilitate the wider application of carbon footprinting by the industry, especially to support business decisions rather than just for labelling, a carbon footprint calculation tool is being developed for the Thai agri-food industry, so called "FOODprint". This tool will help to coach the industry on data collection, sources of secondary data (such as from national database) and to assist in the calculation of carbon footprint (Gheewala and Mungkung, 2012).

National level product category rules (PCRs)[1] are also being developed for the agriculture sector including fruit and vegetables, livestock, fisheries and aquaculture (Mungkung and Gheewala, 2012).

Along with the LCI database development and product carbon footprinting activities, several training programs on LCA and carbon footprinting are being conducted throughout the country to develop human capacity – researchers, consultants as well as companies are being trained. NSTDA has recently also commissioned the author to develop a research capacity building program in LCA for a period of three years, the focus being on human capacity development as well as producing papers in peer-reviewed international journals. The program is also focused mainly on agriculture for food and fuel, especially related, though not limited, to effects on climate change.

One of the major challenges being faced is the representativity of the agricultural databases as most crops are produced by small-holders sometimes running into several thousands; cultivation practices, agricultural inputs and yields can vary quite widely even between adjacent farms. Processors often source crops from open markets with limited access to information on the actual cultivation sites. Another major issue is the need for development of information on impact assessment especially vis-à-vis land use change and biodiversity. At present, greenhouse gas emissions from land use change are usually estimated using default factors from IPCC's tier 1 method which are rather coarse; this can have a significant impact on the results (Siangjaeo et al., 2011). Biodiversity is usually not even being assessed in most studies due to lack of reliable information and assessment methods. These issues will be crucial particularly for sustainable production of food and (bio)fuel with the likely increase of demands for both in the coming years.

3. LCA in the agri-food sector in the region

Apart from Thailand, several countries in the region have had some experience with LCA, some countries more than others. Malaysia has recently done a national level study on life cycle greenhouse gas emissions in the palm oil sector (Choo et al., 2011), though there have been many other research studies (e.g. Yee et al., 2009; Hassan et al., 2011; Hansen et al., 2012). An LCI database, MY-LCID, has also been developed at the national level including the basic categories – energy carriers, materials (including agricultural production means), systems and transport services. The MY-LCID includes 150 regionalised datasets (based on modelled data) and more than 50 datasets based on actual production activities in the country (further information can be obtained by writing to mylcid@sirim.my). LCA is being introduced in Indonesia too at the national level particularly to support the government's plan to reduce GHG emissions; though there have been individual studies carried out by researchers (e.g. Kamahara et al., 2010; Harsono et al., 2012). Vietnam has yet to start applying LCA at the national level, though there are some individual research studies reported in literature (e.g. Phong et al., 2011).

With the objective to promote LCA in the agri-food sector at the regional level, developing regional collaboration and developing an "Asian Food Database", the LCA Agri-food Network has recently been initiated (lca-agrifood-asia.org). One of the first activities was the organisation of the first regional workshop on "LCA Agri-food Asia" in February 2012. The workshop was organised by Kasetsart University, the Joint Graduate School of Energy and Environment, the Asian Institute of Technology and NSTDA of Thailand along with the National Agriculture and Food Organisation (Agriculture Research Centre) and TCO2 company of Japan. About 160 participants from 10 different countries attended the event. Experiences on LCA in agri-food were shared from Indonesia, Japan, Korea, Malaysia and Thailand. Experiences from EU were shared by invited experts from the University of Surrey, UK and the French Agricultural Research for Development Centre (CIRAD). The event led to the development of the LCA Agri-food Asia Forum. The next workshop is planned for 2013 in Indonesia.

4. Outlook for the future

By their very nature, LCAs stretch across the entire life cycle of products which in this age of globalisation includes not only countries, but regions, continents or even the whole world. Data requirements are thus not restricted to national boundaries. Developing national LCI databases is of course imperative, but not an end in itself. As outlined in the previous section, many countries in Asia are taking the first baby steps towards cooperation. The industrialised countries with more experience in developing databases as well as in sharing data will need to be actively involved in capacity building efforts for both these aspects. A useful effort is being coordinated by the United Nations Environment Programme (UNEP, 2011); the current popularity of carbon and water footprinting as well as environmental labelling in general, also provides a driving

[1] PCRs are sets of specific rules, requirements and guidelines for developing Type III environmental declarations for one or more product categories (BSI, 2008)

force. Development of standardised data formats (e.g. ILCD) is a major step in the right direction. Capacity building of researchers, policy makers, and industry is another relevant step. Development of the national LCI database requires much effort in engagement of industrial associations for participating in the data collection efforts as well as to allay concerns of compromising confidentiality. Attempts at international collaboration will exacerbate such concerns, especially vis-à-vis perceived impacts on international trade. These issues will need to be urgently addressed to facilitate cooperation. It would be useful to learn from the experience of national and international database efforts such as Agri-BALYSE (France), LCA food database (Denmark), Ecoinvent, etc. on how they dealt with such issues. This would provide further directions for the future.

5. Concluding remarks

LCA in the agri-food sector has been recently established very strongly in Thailand through various national level initiatives such as the development of the LCI database and product carbon footprinting and labelling. Of course much work still needs to be done on improving the quality, representativity and reliability of the data; this will support the further application of LCA especially by the industries. LCA is also on the rise in other countries in the region, though some countries are still far from realising its importance. Methodological issues particularly pertaining to land use change and biodiversity need urgent attention. Efforts are ongoing to disseminate knowledge and experience from Thailand to other countries which are still in the initial stages of LCA development through the recently initiated LCA Agri-food network. Japan has played an important role in partnering with Thailand in this effort as have other countries such as Indonesia, Malaysia and Vietnam. European countries can also be interesting partners as they have much experience in developing national and regional-level agriculture and food databases as well as assessment tools. These experiences would be very useful for the emerging countries in Asia so that they can leapfrog the initial problems that the developed countries must have faced while initiating their LCA activities in the agri-food sector. The developed countries on the other hand can also learn about agriculture and food industries in Asia which will be useful especially for the many environmental labelling efforts that are ongoing and where many of the upstream process of agriculture and food processing are based in Asia. More engagement can also come through joint efforts at networking; the LCA Agri-food Asia could also link up with existing networks worldwide, such as *inter alia* UNEP/SETAC Life Cycle Initiative, ENVIFOOD protocol and the Sustainability Consortium, that have similar goals.

6. Acknowledgements

Dr. Thumrongrut Mungcharoen and Ms. Patcharaporn Pongpat of NSTDA, Thailand; Dr. Rattanawan Mungkung of Kasetsart University, Thailand and Dr. Chen Sau Soon of SIRIM Berhad, Malaysia are acknowledged for providing supporting information. Special thanks to Dr. Hayo van der Werf, French National Institute for Agricultural Research (INRA) for making useful suggestions on the paper. The opinions expressed in this article and any errors are the sole responsibility of the author.

7. References

BSI, 2008. PAS 2050:2008. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution.

Choo, Y.M., Muhamad, H., Hashim, Z., Subramanian, V., Puah, C.W., Tan, Y.A., 2011. Determination of GHG contributions by subsystems in the oil palm supply chain using LCA approach. Int J Life Cycle Assess. 16, 669-681.

Gheewala, S.H., 2011. Life Cycle Assessment (LCA) to evaluate environmental impacts of bioenergy projects. J Sust Energy Environ. Special Issue, 35-38.

Gheewala, S.H., Mungkung, R., 2012. FOODprint as a tool to support GHG reduction in the Thai agri-food sector. Proceedings of the 8th International Conference on LCA in the Agri-Food Sector, Rennes, France, 2-4 October 2012.

Hansen, S.B., Olsen, S.I., Ujang, Z. 2012. Greenhouse gas reductions through enhanced use of residues in the life cycle of Malaysian palm oil derived biodiesel. Bioresource Technol. 104, 358-366.

Harsono, S.S., Prochnow, A., Grundmann, P., Hansen, A., Hallmann, C., 2012. Energy balances and greenhouse gas emissions of palm oil biodiesel in Indonesia. Glob Change Biol. 4, 213-228.

Hassan, M.N.A., Jaramillo, P., Griffin, W.M., 2011. Life cycle GHG emissions from Malaysian oil palm bioenergy development. Energ Policy. 39, 2615-2625.

Kamahara, H., Hasanudin, U., Widiyanto, A., TaDaimon, H., Tachibana, R., Atsuta, Y., Goto, N., Fujie, K., 2010. Improvement potential for net energy balance of biodiesel derived from palm oil: A case study from Indonesian practice. Biomass Bioenerg. 34, 1818-1824.

Lebel, L., Mungkung, R., Gheewala, S.H., Lebel, P., 2010. Innovation cycles, niches and sustainability in the shrimp aquaculture industry in Thailand. Environ Sci Policy. 13, 291-302.

Lohsomboon, P., 2002. LCA Activities in Thailand. Int J Life Cycle Assess. 7, 181.

Phong, L.T., de Boer, I.J.M., Udo, H.M.J., 2011. Life cycle assessment of food production in integrated agriculture-aquaculture systems of the Mekong Delta. Livest Sci. 139, 80-90.

Mungkung, R., Gheewala, S.H., 2012. Carbon footprinting and labelling of agri-food products: Practical issues for the development of Product Category Rules (PCR). Proceedings of the 8th International Conference on LCA in the Agri-Food Sector, Rennes, France, 2-4 October 2012.

Mungkung, R., Gheewala, S.H., Kanyarushoki, C., Hospido, A., van der Werf, H., Poovarodom, N., Bonnet, S., Aubin, J., Moreira, M.T., Feijoo, G., 2012. Product carbon footprinting in Thailand: A step towards sustainable consumption and production? Environmental Development 3, 100-108.

Mungkung, R., Gheewala, S.H., Towprayoon, S., Poovarodom, N., Tanaparisutti, P., Chupairote, T., 2010. Carbon Footprinting of Thai Rice Products: Practical Issues for a Harmonised Methodology, 9th International Conference on Ecobalance, 09-12 November 2010, Tokyo, Japan

Mungkung, R., Udo de Haes, H.A., Clift, R., 2006. Potentials and Limitations of Life Cycle Assessment in Setting Ecolabelling Criteria: A case study of Thai shrimp aquaculture product. Int J Life Cycle Assess. 11, 55-59.

Nilsalab, P., Gheewala, S.H., Mungkung, R., 2012. Water assessment of agrofuels feedstock cultivation: Methodology approaches, Environment and Natural Resources Journal. (in press)

Nguyen, T.L.T., 2007. Life-cycle assessment of bio-ethanol as an alternative transportation fuel in Thailand. Ph.D. dissertation, Joint Graduate School of Energy and Environment, King Mongkut's University of Technology Thonburi, Bangkok, Thailand.

Nguyen, T.L.T., Gheewala, S.H., 2008a. Life cycle assessment of fuel ethanol from cassava in Thailand, Int J Life Cycle Assess. 13, 147-154.

Nguyen, T.L.T., Gheewala, S.H., 2008b. Life cycle assessment of fuel ethanol from cane molasses in Thailand. Int J Life Cycle Assess. 13, 301-311.

Phumpradab, K., Gheewala, S.H., Sagisaka, M., 2009. Life cycle assessment of natural gas power plants in Thailand. Int J Life Cycle Assess. 14, 354-363.

Siangjaeo, S., Gheewala, S.H., Unnanon, K., Chidthaisong, A., 2011. Implications of land use change on the life cycle greenhouse gas emissions from palm biodiesel production in Thailand. Energy for Sustainable Development. 15, 1-7.

Silalertruksa T., 2010. Sustainability assessment of biofuels for transport in Thailand. Ph.D. dissertation, Joint Graduate School of Energy and Environment, King Mongkut's University of Technology Thonburi, Bangkok, Thailand.

Silalertruksa T., Gheewala, S.H., 2010. Security of feedstocks supply for future bio-ethanol production in Thailand, Energ Policy. 38, 7476-7486.

Silalertruksa T., Gheewala, S.H., 2011. Long-term bio-ethanol system and its implications on GHG emissions: A case study of Thailand, Environ Sci Technol. 39, 834-843.

Silalertruksa, T., Gheewala, S.H., 2012. Food, fuel and climate change: Is palm-based biodiesel a sustainable option for Thailand? J Ind Ecol. (in press)

Suwanit W., Gheewala, S.H., 2011. Life cycle assessment of mini-hydropower plants in Thailand. Int J Life Cycle Assess. 16, 849-858.

UNEP, 2011. Global guidance principles for life cycle assessment databases – A basis for greener processes and products, UNEP/SETAC Life Cycle Initiative, United Nations Environment Programme.

Yee, K.F., Tan, K.T., Abdullah, A.Z., Lee, K.T., 2009. Life cycle assessment of palm biodiesel: Revealing facts and benefits for sustainability. Appl Energ. 86(Supplement 1), S189-S196.

Yuttitham, M., 2009. Full carbon accounting in bioenergy production from sugarcane. Ph.D. dissertation, Joint Graduate School of Energy and Environment, King Mongkut's University of Technology Thonburi, Bangkok, Thailand.

The yield performance of organic agriculture

Verena Seufert[1,*], Navin Ramankutty[1], Jonathan A. Foley[2]

[1] Department of Geography and Global Environmental and Climate Change Center, McGill University, Quebec H2T 3A3, Canada
[2] Institute on the Environment (IonE), University of Minnesota, 1954 Buford Aevnue, St. Paul, Minnesota 55108, USA
[*] Corresponding author. E-mail: *verena.seufert@mail.mcgill.ca*

ABSTRACT

Organic agriculture is often proposed as a solution to the challenge of producing sufficient food in a sustainable way. However, organic agriculture is also critizised for its purported lower productivity compared to conventional agriculture. Here we use a comprehensive meta-analysis to examine the relative yield performance of organic and conventional farming systems globally. Our analysis of available data shows that, overall, organic yields are typically lower than conventional. The yield difference varies, however, depending on site and system characteristics. Under certain conditions – i.e., with good management practices, particular crop types and growing conditions – organic systems can nearly match conventional yields, while under others it currently cannot. To establish organic agriculture as an important tool in sustainable food production, the factors limiting organic yields need to be more fully understood, alongside assessments of the many social, environmental and economic benefits of organic farming systems.

Keywords: organic agriculture, yields, meta-analysis, sustainability

1. Introduction

Agriculture is a major source of global environmental degradation (Foley et al., 2005). Numerous recent reports have emphasized the need for drastic changes in the food system in order to meet the double challenge of feeding a growing population with a rising demand for high-quality diets while minimizing the environmental impacts of food production (Foley et al., 2011). 'Alternative' management practices that try to mimic ecological processes while minimizing external inputs are often suggested as important tools in the solution to this problem (Schutter 2010). Organic agriculture, which currently covers 0.9% of global agricultural land (Willer and Kilcher 2011), is the most prominent of these alternative farming systems. It is a farming system aimed at producing food with minimal harm to ecosystems, animals or humans.

Driven by consumer concerns about food safety and environmental issues the market for organic products has grown rapidly and more than tripled in the last decade (Willer and Kilcher 2011). Notwithstanding its increasing popularity amongst consumers, organic agriculture has many ardent opponents (Trewavas 2001). One of the main objections against it is its purported lower productivity – critiques argue that organic agriculture would need considerable more land to produce the same amount of food, resulting in more widespread deforestation and biodiversity loss.

A recent study attempted to address this criticism by analysing data from the literature on 293 organic-to-conventional yield comparisons (Badgley et al., 2007). They concluded that organic agriculture could, overall, provide sufficient food to feed the current population for the same amount of land used. This conclusion was, however, highly contested by several critiques who claimed that serious methodological flaws had led to an overestimation of organic yields (Avery 2007; Connor 2008).

2. Methods

Here we have performed a comprehensive synthesis of the current scientific literature on organic-to-conventional yield comparisons using formal meta-analysis techniques. We compiled our own dataset of scientific studies comparing organic and conventional yields. To address the criticisms of the Badgley et al., (2007) study we used several selection criteria: (1) we restricted our analysis to studies on "truly" organic systems, defined as those with certified organic management or non-certified organic management, following the standards of organic certification bodies; (2) only included studies with comparable spatial and temporal scales for both organic and conventional systems; and (3) only included studies reporting (or we could estimate) sample size and error. Conventional systems were either high- or low-input commercial systems, or subsistence agriculture. 66 studies met these criteria, representing 62 study sites, and reporting 316 organic-to-conventional yield comparisons on 34 different crop species.

We used the natural logarithm of the response ratio, which is the ratio between organic and conventional yields, as effect size and calculated a weighted average by weighting each observation by the inverse of the mixed-model variance (Hedges et al., 1999). An effect size is considered significant if its confidence interval (CI) does not overlap one in the backtransformed response ratio. In addition to yields, we collected information on study characteristics like crop rotations, fertiliser type and experimental study design as well as information on the biophysical characteristics of the study site, and analysed their influence as well. To test for the influence of categorical variables on yield effect sizes we examined the between-group heterogeneity

(Q_B). A significant Q_B indicates that there are differences in the effect sizes between different classes of a categorical variable (Rosenberg et al., 2000). All statistical analyses were carried out in MetaWin 2.0 (Rosenberg et al., 2000). For representation in graphs effect sizes were backtransformed to response ratios.

3. Results

The overall organic-to-conventional yield ratio is 0.75 (with a 95% CI of 0.71 to 0.79), meaning that across the 316 yield comparisons organic yields are 25% lower than conventional yields (Fig. 1a). This result only changes slightly (yield ratio of 0.74) if the analysis is limited to studies following high scientific quality standards (Fig. 2).

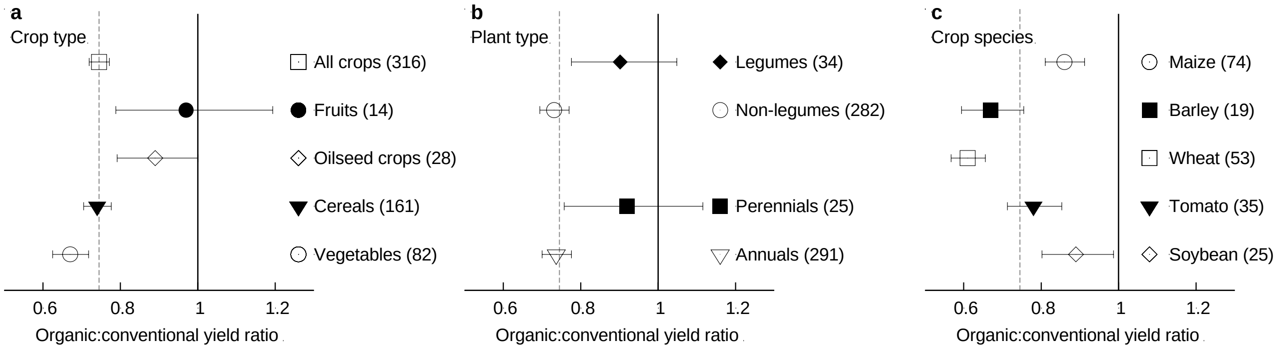


Figure 1. Influence of different crop types (a), plant types (b) and crop species (c) on organic-to-conventional yield ratios. Only those crop types and crop species are shown that were represented by at least 10 observations and 2 different studies. Values are effect sizes with 95% CIs. A significant response is when the CI does not overlap 1. The number of yield observations in each class is shown in parentheses. The dotted line indicates the cumulative effect size across all classes.

The performance of organic systems varies substantially across crop types and species (Fig. 1a-c). Only categories with a significant QB are presented in Figures. Organic yields of fruits and oilseed crops show a small, but not statistically significant, difference to conventional crops (their CI overlap zero), whereas organic cereals and vegetables have significantly lower yields (-26% and -33% respectively) (Fig. 1a). These differences seem to be related to the better organic performance (referring to the relative yield of organic to conventional systems) of perennial over annual crops and of legumes over non-legumes (Fig. 1b). Marked differences can, however, also be observed between crop species of the same crop type -- maize outperforms wheat and barley yields under organic management (Fig. 1c).

Part of the yield response can be explained by differences in the amount of nitrogen (N) input received by the two systems (Fig. 3a). When organic systems receive higher quantities of N than conventional systems, organic performance improves, whereas conventional systems do not benefit from more N. In other words, organic systems appear to be N limited, whereas conventional systems are not. To achieve yields that are comparable to conventional systems, organic agriculture thus appears to require higher N inputs. This could be due to organic N inputs being less readily available to plants. Even if the total amount of N in soils managed with organic or conventional methods do not differ, the composition of the N pools often do (Stockdale et al., 2002). Soils under organic management often have high organic matter and organic N pools but low mineral N content (Stockdale et al., 2002). The release of plant-available mineral N from these organic pools is slow and does often not keep up with the high crop N demand during the peak growing period (Berry et al., 2002; Pang and Letey 2000). Nitrogen availability has thus been found to be a major yield-limiting factor in many organic systems (Berry et al., 2002; Clark et al., 1999).

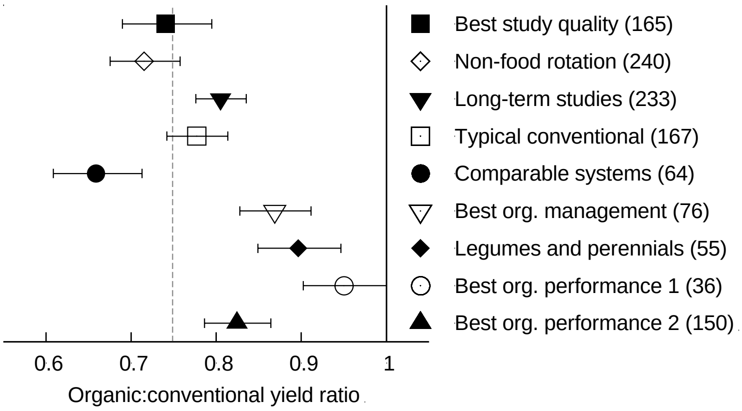


Figure 2. Sensitivity study of organic-to-conventional yield ratios: 'Best study quality' - studies from peer-reviewed journals using an appropriate study design and making appropriate inferences; 'non-food rotation' - studies where the organic and conventional systems rotations have a similar duration of non-food crops; 'long-term studies' - excludes studies that are both very short and on recently converted plots; 'typical conventional' - restricted to commercial conventional systems with conventional yields comparable to local averages; 'comparable systems' - studies that use appropriate study design and make appropriate inferences, where organic and conventional have the same length of non-food rotation and receive similar amounts of N inputs; 'best org management' - excludes studies without BMP or crop rotations; 'legumes and perennials' - only legumes and perennials; 'best org performance 1' - rainfed legumes and perennials on weak-acidic to weak-alkaline soils; 'best org performance 2' - weak-acidic to weak-alkaline soils under rainfed conditions.

The better performance of organic legumes and perennials is, instead, not because they received more N, but rather because they seem to be more efficient at using N. Legumes are not as dependent on external N sources as non-legumes, whereas perennials, owing to their longer growing period and extensive root systems, can achieve a better synchrony between nutrient demands and the slow release of N from organic matter (Crews 2005).

Organic crops perform better on weak-acididc to weak-alkaline soils (that is, soils with a pH between 5.5 and 8.0, Fig. 3e). A possible explanation is the difficulty of managing phosphorus (P) in organic systems. Under strongly alkaline and acidic conditions, P is less readily available to plants as it forms insoluble phosphates, and crops depend to a stronger degree on soil amendments and fertilisers. Organic systems often do not receive adequate P inputs to replenish the P lost through harvest (Oehl et al., 2002). To test this hypothesis we need further research on the performance and nutrient dynamics of organic agriculture on soils of varying pH.

Studies that reported having applied best management practices (BMP) in both systems show better organic performance (Fig. 3c). Nutrient and pest management in organic systems rely on biological processes to deliver plant nutrients and to control weed and herbivore populations. Organic yields thus depend more on knowledge and good management practices than conventional yields.

It is often reported that organic yields are low in the first years after conversion and gradually increase over time, due to improvements in soil fertility and management skills (Martini et al., 2004). This is supported by our analysis: organic performance improves in studies that either lasted for more than two seasons, or were conducted on plots that have been organic for at least three years (Fig. 2, Fig. 3d).

Water relations also influence organic yield ratios -- organic performance is -35% under irrigated, but only -17% under rainfed conditions (Fig. 3e). This could be due to a relatively better organic performance under variable moisture conditions in rainfed systems. Soils managed with organic methods have shown better water-holding capacity and water infiltration rates and have produced higher yields than conventional systems under drought conditions and excessive rainfall (Lotter et al., 2003). This has been attributed to the higher soil organic matter content and the increased aggregate stability of soils managed with organic methods (Stockdale et al., 2001). On the other hand, organic systems are often nutrient-limited (see earlier discussion), and thus probably do not respond as strongly to irrigation as conventional systems.

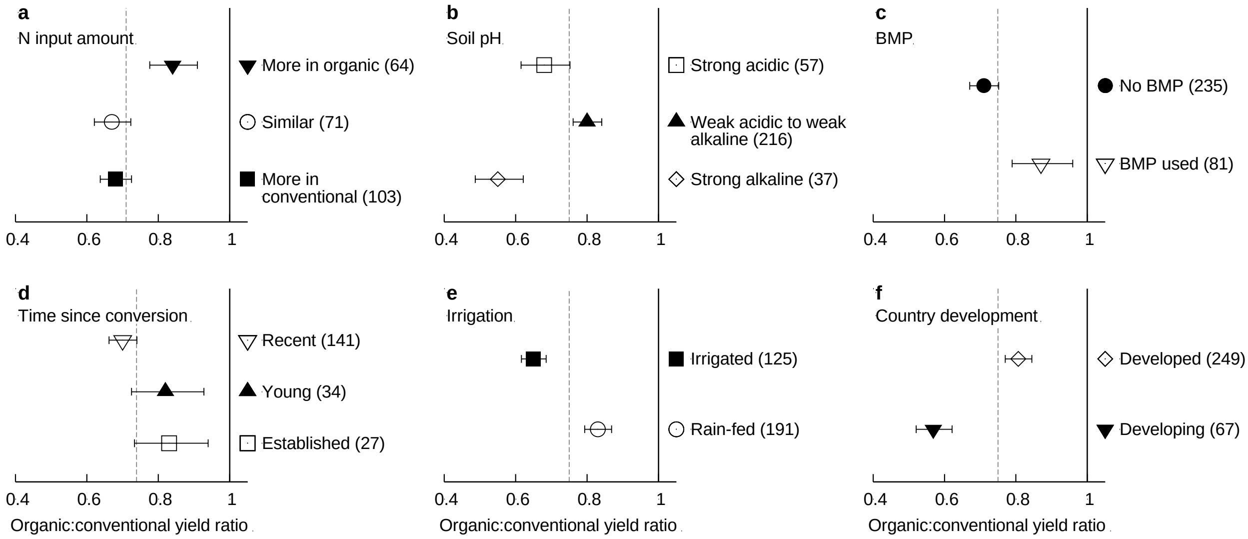


Figure 3. Influence of the amount of N input (a), soil pH (b), BMP (c), time since conversion to organic management (d), irrigation (e) and country development (f) on organic-to-conventional yield ratios.

The majority of studies in our meta-analysis come from developed countries (Fig. 3f). Comparing organic agriculture across the world, we find that in developed countries organic performance is, on average, -20%, whereas in developing countries it is -43% (Fig. 3f). This poor performance of organic in developing countries may be explained by the fact that a majority of the data (58 of 67) from developing countries seem to have atypical conventional yields (>50% higher than local yield averages), coming from irrigated lands (52 of 67), experimental stations (54 of 67) and from systems not using BMP (67 of 67). In the few cases from developing countries where organic yields are compared to conventional yields typical for the location or where the yield data comes from surveys, organic yields do not differ significantly from conventional yields because of a wide confidence interval resulting from the small sample size (N = 8 and N = 12 respectively).

The results of our meta-analysis differ dramatically from the previous results of Badgley et al., (2007). While our organic performance is lower than Badgley et al., (2007) in developed countries (-20% compared to -8%), our results are markedly different in developing countries (-43% compared to +80%). (But note that these figures are not directly comparable, as the Badgley et al., study used a simple arithmetic mean, while we used a weighted effect size to analyse the central tendency of the data.) This is because they mainly included yield comparisons from conventional low-input subsistence systems, while our dataset mainly includes data from high-input systems for developing countries. However, Badgley et al., (2007) compared subsistence systems to yields that were not truly organic, and/or from surveys of projects that lacked an adequate control. Not a single study comparing organic to subsistence systems met our selection criteria and could be included in the meta-analysis. We cannot, therefore, rule out the claim (Scialabba and Hattam 2002) that organic agriculture can increase yields in smallholder agriculture in developing countries. But owing to a lack of quantitative studies with appropriate controls we do not have sufficient scientific evidence to support it either. Fortunately, the Swiss Research Institute of Organic Agriculture (FiBL) recently established the first long-term comparison of organic and different conventional systems in the tropics (FiBL 2011). Such well- designed long-term field trials are urgently needed.

4. Discussion

Our analysis shows that yield differences between organic and conventional agriculture do exist, but that they are highly contextual. When using best organic management practices yields are closer to (-13%) conventional yields (Fig. 2). Organic agriculture also performs better under certain agroecological conditions – e.g., organic legumes or perennials, on weak-acidic to weak-alkaline soils, in rainfed conditions, achieve yields that are only 5% lower than conventional yields (Fig. 2). On the other hand, when only the most comparable conventional and organic systems are considered, the yield difference is as high as 34% (Fig. 2).

Although we were able to identify factors contributing to variations in organic performance, several other potentially important factors could not be tested due to a lack of appropriate studies. For example, we were not able to analyse tillage, crop residue or pest management. Also, most of the studies included in our analysis experienced favourable growing conditions. Performance of organic agriculture under dry climates, short growing seasons and on unfertile soils should be studied more thoroughly and the potential mechanistic differences with conventional agriculture under these biophysical conditions examined. In addition, for a farming system comparison it would be desirable to examine a system metric like total human-edible calorie or net energy yield of the entire crop rotation rather than biomass yield of a single crop species. Probably most importantly, however, more studies on organic agriculture need to be conducted that are representative of the agricultural reality of the majority of farming systems. In this meta-analysis organic systems were mostly compared to commercial high input systems (which had predominantly above-average yields and came from irrigated agriculture in developing countries). However, 75% of global cropland is not irrigated (Portmann et al., 2010) and 50% of cropland receives fertiliser rates of less than 2.5 kg N ha^{-1} (Potter et al., 2010). The conventional systems in our database received on average 126 kg N ha^{-1} and the organic systems 118 kg N ha^{-1}.

To better understand the performance of organic agriculture, we should therefore: (1) systematically analyse the long-term performance of organic agriculture under different management regimes; (2) study organic systems under a wider range of biophysical conditions; (3) evaluate the productive performance of farming systems through holistic system metrics and (4) examine the yield performance of organic and conventional agricultural systems of smallholder agriculture.

Yields are, however, only part of a suite of economic, social and environmental factors that need to be considered when gauging the benefits of different farming systems. In developed countries, the central question is whether the environmental benefits of organic crop production would offset the costs of any lower yield (such as increased food prices and reduced food exports). Studies have shown that organic practices can have a reduced environmental impact (Bengtsson et al., 2005; Siegrist et al., 1998). However, although the overall environmental performance of organic agriculture is often positive, the environmental impact per unit output or per unit input is not always better than in conventional agriculture (Leifeld et al., 2009).

In developing countries, instead, a key question is whether organic agriculture can help alleviate poverty for small farmers and increase food security. On the one hand, it has been suggested that organic agriculture may improve farmer livelihoods due to cheaper inputs, higher and more stable prices, the possibility of integrating traditional knowledge and risk diversification (Scialabba and Hattam 2002). On the other hand, organic agriculture in developing countries is often an export-oriented system tied to a certification process by international bodies, and its profitability can vary in different locations and years (Raynolds 2004; Valkila 2009).

5. Conclusion

The discourse on organic agriculture needs to move away from an ideologically charged to a more balanced debate that is informed by empirical evidence. On the one hand, benefits and problems of organic agriculture need to be assessed objectively and organic practices improved accordingly. The biophysical, economic and social conditions under which organic agriculture would be favourable should be investigated. A less dogmatic approach to organic agriculture might also be helpful. Organic certification systems should reconsider the roots of organic agriculture as a farming system concerned about environmental outcomes, not about ideological prohibitions of specific inputs. Organic agriculture should be first of all a management system that uses best environmental practices. If, as the results of this meta-analysis imply, organic fertilisation requires higher nitrogen inputs than conventional agriculture for achieving high yields, mineral fertilisers might be a better option for increasing poor farmers productivity. Even those that promote the adoption of alternative, agro-ecological framing practices caution against the total exclusion of mineral fertilisers (Schutter 2010). On the other hand, conventional farming systems should take advantage of the experience gained in succesful organic systems and implement those practices that can improve the environmental sustainability while maintaining the productivity of the system. Mixed approaches that combine the use of mineral fertilisers with succesful practices from organic management, like cover cropping, mulching, use of crop residues and increased crop diversity, can show both high yields and reduced environmental problems (Tonitto et al., 2006).

From an agricultural perspective to achieve sustainable food security we need to produce more food at affordable prices, ensuring livelihoods to farmers and using management practices that reduce the environmental costs of food production. Considering the scale of the challenge ahead of us there is a strong need to enhance our efforts of understanding how different farming systems and management practices contribute to the provision of sustainable food.

6. References

Avery, A. A., 2007. 'Organic abundance' report: fatally flawed. Renew. Agric. Food Syst. 22, 321-323.

Badgley, C., Moghtader, J., *et al.*, 2007. Organic agriculture and the global food supply. Renew. Agric. Food Syst. 22, 86.

Bengtsson, J., Ahnström, J.Weibull, A.-C., 2005. The effects of organic agriculture on biodiversity and abundance: a meta-analysis. J. Appl. Ecol. 42, 261-269.

Berry, P., Sylvester Bradley, R., *et al.*, 2002. Is the productivity of organic farms restricted by the supply of available nitrogen? Soil Use Manage. 18, 248-255.

Clark, M. S., Horwath, W. R., *et al.*, 1999. Nitrogen, weeds and water as yield-limiting factors in conventional, low-input, and organic tomato systems. Agr. Ecosyst. Environ. 73, 257-270.

Connor, D. J., 2008. Organic agriculture cannot feed the world. Field Crops Res. 106, 187-190.

Crews, T., 2005. Perennial crops and endogenous nutrient supplies. Renew. Agric. Food Syst. 20, 25-37.

FiBL, 2011, 22.07.2011. Farming System Comparison in the Tropics. Retrieved 23.08.2011, from http://www.systems-comparison.fibl.org/.

Foley, J. A., Defries, R., *et al.*, 2005. Global consequences of land use. Science 309, 570-574.

Foley, J. A., Ramankutty, N., *et al.*, 2011. Solutions for a cultivated planet. Nature 478, 337-342.

Hedges, L. V., Gurevitch, J.Curtis, P. S., 1999. The meta-analysis of response ratios in experimental ecology. Ecology 80, 1150-1156.

Leifeld, J., Reiser, R.Oberholzer, H.-R., 2009. Consequences of Conventional versus Organic farming on Soil Carbon: Results from a 27-Year Field Experiment. Agron. J. 101, 1204.

Lotter, D., Seidel, R.Liebhardt, W., 2003. The performance of organic and conventional cropping systems in an extreme climate year. Am. J. Alternative Agric. 18, 146-154.

Martini, E., Buyer, J. S., *et al.*, 2004. Yield increases during the organic transition: improving soil quality or increasing experience? Field Crops Res. 86, 255-266.

Oehl, F., Oberson, A., *et al.*, 2002. Phosphorus budget and phosphorus availability in soils under organic and conventional farming. Nutr. Cycl. Agroecosys. 62, 25-35.

Pang, X.Letey, J., 2000. Organic farming: challenge of timing nitrogen availability to crop nitrogen requirements. Soil Sci. Soc. Am. J. 64, 247-253.

Portmann, F. T., Siebert, S.Döll, P., 2010. MIRCA2000—Global monthly irrigated and rainfed crop areas around the year 2000: A new high-resolution data set for agricultural and hydrological modeling. Global Biogeochemical Cycles 24.

Potter, P., Ramankutty, N., *et al.*, 2010. Characterizing the spatial patterns of global fertilizer application and manure production. Earth Interactions 14, 1-22.

Raynolds, L. T., 2004. The globalization of organic agro-food networks. World Development 32, 725-743.

Rosenberg, M. S., Gurevitch, J.Adams, D. C., 2000. MetaWin: Statistical Software for Meta-analysis: Version 2, Sinauer.

Schutter, O. d., 2010. Report submitted by the Special Rapporteur on the right to food. New York, United Nations: 21.

Scialabba, N.Hattam, C., Eds. (2002). *Organic agriculture, environment and food security*. Environment and Natural Resources Series. Rome, Food and Agriculture Organization of the United Nations (FAO).

Siegrist, S., Schaub, D., *et al.*, 1998. Does organic agriculture reduce soil erodibility? The results of a long-term field study on loess in Switzerland. Agr. Ecosyst. Environ. 69, 253-264.

Stockdale, E., Lampkin, N. H., *et al.*, 2001. Agronomic and environmental implications of organic farming systems. Adv. Agron. 70, 261-262.

Stockdale, E., Shepherd, M., *et al.*, 2002. Soil fertility in organic farming systems-fundamentally different? Soil Use Manage. 18, 301-308.

Tonitto, C., David, M. B.Drinkwater, L. E., 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. Agriculture Ecosystems & Environment 112, 58-72.

Trewavas, A., 2001. Urban myths of organic farming. Nature 410, 409-410.

Valkila, J., 2009. Fair trade organic coffee production in Nicaragua--Sustainable development or a poverty trap? Ecol. Econ. 68, 3018-3025.

Willer, H.Kilcher, L., 2011. The World of Organic Agriculture - Statistics and Emerging Trends 2011, IFOAM & FiBL, Bonn & Frick.

ENVIFOOD Protocol: launch of the collectively-agreed sectorial methodology for assessing the environmental performance of food and drink products in Europe

Camillo De Camillis[1,*], Jean-Christophe Bligny[2], David Pennington[1], Balázs Pályi[3], Christian Bauer[4], Urs Schenker[5], Henry King[6], Lars Lundquist[5], Nicolas Martin[7], Nicole Unger[6], Laura Draucker[8], Øyvind Vessia[9], Coen Blomsma[10], Nathan Pelletier[1], Hans Blonk[11], Rudi Sueys[12], Anna Papagrigoraki[13], Pascal Gréverath[14], Herbert Aichinger[15], Jean-Pierre Rennaud[16]

[1] European Commission, Joint Research Centre (JRC), Institute for Environment and Sustainability (IES), via Enrico Fermi 2749, 21027 Ispra (Va), Italy
[2] Danone, 11 Avenue Géneral Dupas, 74500 Evian, France
[3] FoodDrinkEurope, Avenue des Arts 43, 1040 Brussels, Belgium
[4] SIG Combibloc, Rurstrasse 58, 52441 Linnich, Germany
[5] Nestec Ltd, Nestlé Research Center, Vers-chez-les-Blanc, 1000 Lausanne 26, Switzerland
[6] Unilever Safety and Environmental Assurance Centre, Colworth Park, Sharnbrook, MK44 1LQ Bedford, UK
[7] FEFAC, Rue de la Loi, 223, Bte 2 B-1040 Bruxelles, Belgium
[8] World Resources Institute (WRI), 10 G St. NE, Suite 800, 20002 Washington, DC, USA
[9] European Commission, DG Energy, BRU-DM24 04/109, 1040 Brussels, Belgium
[10] FEDIOL, Avenue de Tervuren 168, 1150 Bruxelles, Belgium
[11] Blonk Milieu Advies, Gravin Beatrixstraat 34, 2805 PJ Gouda, The Netherlands
[12] Coca-Cola Europe, Bergense Steenweg 1424, 1070 Brussels, Belgium
[13] CEFS, Avenue de Tervuren 182, B-1150 Brussels, Belgium
[14] Nestlé, Avenue Nestlé 55, 1800 Vevey, Switzerland
[15] European Commission, DG Environment, Avenue de Beaulieu 5, 1049 Brussels, Belgium
[16] Danone Institute, Route Départementale 128, 91767 Palaiseau, France
[*] Corresponding author. E-mail: *camillo.de-camillis@jrc.ec.europa.eu*

ABSTRACT

Lack of consistency in both the substance and application of methodologies and guidelines for assessing and communicating the environmental performance of food and drink products has the potential to confuse consumers and other stakeholders involved in the food and drink supply chains. Furthermore, it poses an unnecessary burden on those organisations requested to evaluate the environmental performance of their product according to several different methodologies.

In order to address this issue, business associations and other food supply chain partners have involved the European Commission in establishing the European Food Sustainable Consumption and Production (SCP) Round Table. Since 2009, Round Table members have been working together on a commonly-agreed and science-based framework for assessment and communication of the environmental performance of food and beverage products in Europe. The first milestone was reached in 2010 when Round Table members agreed on a set of principles to guide working groups in harmonizing relevant approaches for assessment and communication. On this basis, the Round Table Working Group 1 reached agreement on key methodological aspects for environmental assessment at a scientific workshop in 2010. A detailed analysis of relevant data, methodologies and guidelines for assessing the environmental performance of foods and drinks was conducted following the 2010 workshop. This analysis identified aspects of Life Cycle Assessment (LCA) practice where existing methodologies agree, as well as inconsistencies among existing approaches. A second workshop was organized in 2011 to reach consensus on the outstanding inconsistencies. Remaining unresolved issues were further discussed within Working Group 1, leading to the release of an advanced draft of what will be the future Round Table methodology for environmental assessment named the ENVIFOOD Protocol. A public consultation scheduled during the last quarter of 2012 will be followed by the publication of the ENVIFOOD Protocol in 2013. In conjunction with its publication, the ENVIFOOD Protocol will be tested through a range of pilot studies and the feedback used for refining the guidance.

The ENVIFOOD Protocol provides guidance to support those environmental assessments of food and drink products in Europe conducted in the context of:

- Business-to-business as well as business-to-consumer communication (focus of the Round Table Working Group 2);
- The identification of environmental improvement options (focus of the Round Table Working Group 3).

This paper gives a general overview of the process to arrive at the ENVIFOOD Protocol. The paper also illustrates possible next steps of the Round Table Working Group 1. Round Table members are, in fact, discussing how to develop and adopt Product Category Rules (PCR) in line with the ENVIFOOD Protocol. In parallel, Round Table members are also discussing how to obtain adequate data for assessment and streamlined tools. The establishment of the ENVIFOOD Protocol and PCRs will allow the development of user-friendly and affordable tools for assessment and communication of environmental impacts, thus reducing the burdens of undertaking such assessments, in particular for SME's.

Keywords: Round Table, sustainable consumption and production, harmonisation, assessment methodology, ENVIFOOD Protocol

1. Introduction

Members of the European Food Sustainable Consumption and Production Round Table (SCP Round Table) are committed to helping consumers and other stakeholders make informed choices by providing them with accurate and understandable information on relevant product characteristics, including environmental performance.

In the framework of the Round Table, Working Group 1 (WG1) has been developing a harmonised methodology called the Protocol for the Environment Assessment of Food and Drink Products (ENVIFOOD Protocol). The ENVIFOOD Protocol is intended to provide common guidance for assessing the environmental performance of food and drink products, hence overcoming the difficulties associated with the current proliferation of standards and guidance documents on this subject.

The main purpose of the ENVIFOOD Protocol is the development of a common framework that will increase the scientific reliability and consistency in studies intended to support informed choice. In particular, this will support environmental assessments conducted in the context of business-to-business as well as business-to-consumer communication, and the identification of environmental improvement options.

This paper provides an overview of the multi-stakeholder approach to develop the ENVIFOOD Protocol, an overview of the Protocol itself, and possible next steps.

2. ENVIFOOD Protocol: a stepwise process involving a broad spectrum of stakeholders

As a first step towards the ENVIFOOD Protocol, the European Commission Joint Research Centre (JRC) Institute for Environment and Sustainability (IES) hosted a scientific workshop in 2010 aimed at arriving at a common understanding of what is required for: reliable and robust environmental assessment of food and drink product supply chains; current limitations; and how to move towards more straightforward/focused criteria/guidance/tools (Peacock et al., 2011).

On this basis, and taking into account the Guiding Principles of the Round Table (Food RT, 2010), WG1 elaborated a Road Map for the development and dissemination of the ENVIFOOD Protocol. Major tasks planned in the Road Map over the time period 2010-2012 were: a detailed analysis of data gaps and of methodological issues; drafting of the ENVIFOOD Protocol; public consultation; and revision of the draft Protocol. In order to validate the ENVIFOOD Protocol and facilitate its application, the following additional steps were included in the Road Map: testing and fine-tuning of the ENVIFOOD Protocol; and provision of product specific guidelines, criteria, tools and datasets. Tasks beyond the ENVIFOOD Protocol testing and fine-tuning are currently out of the WG1 mandate and thus subject to future discussion and agreement.

As the goal of the ENVIFOOD Protocol was to be built on the existing science for the environmental assessment of food and drink products, rather than inventing a new methodology from scratch, the detailed analysis was focused on:

- Review of methodologies and guidance documents for the environmental assessment of food and drink products;
- Existing datasets related to food and drink products.

As a first step for the detailed analysis, WG1 focussed on inventorying reference methodologies, guidelines and assessments. The starting point for this inventory was a survey of existing initiatives dealing with sustainability issues worldwide. This inventory was then complemented by: a literature search of both scientific databases (e.g. SCOPUS, Google Scholar and Web of Science) and proceedings of the international conferences on agri-food LCA; and a WG1 internal consultation. Existing and upcoming life cycle-based technical standards and guidance documents were then grouped in the following classes:

- General methodologies (i.e. reference standards and general guidance documents on LCA; general life cycle-based standards and technical agreements on footprints and environmental disclosures);
- Sectorial methodologies (i.e. sector-specific guidance documents and technical agreements on environmental assessment and environmental disclosures);
- Sub-sectorial methodologies (i.e. sub-sector-specific guidance documents and technical agreements on environmental assessment and footprinting; and sub-sectorial rules for product environmental declarations);
- Product-specific guidelines (i.e. product category rules (PCR) for environmental declarations for food and drink products);
- Other methodological inputs (i.e. European Union (EU) laws giving guidance on certain life cycle approaches; relevant life cycle-oriented assessments by international authoritative bodies; any other methodology and guideline linked to food and drink products).

In order to complement the overview in tabular form of carbon footprint methodologies – study prepared for the European Commission's Directorate General (DG) Environment (Ernst&Young and Quantis, 2010) – a new methodology overview was outlined by WG1 drafting group (De Camillis et al., 2011a) scanning a large number of the methodologies and guidelines found in the literature against the applicable characterisation criteria from the aforementioned review for DG Environment.

In parallel to the detailed analysis of methodologies and guidelines, a data gap analysis was also carried out because data availability and data quality were detected by WG1 as corner stones of any robust assessment. Based on the insights from the data mapping for the French environmental footprinting initiative, data issues were identified and a technical report (De Camillis, 2011) was released accordingly to support development of the ENVIFOOD Protocol and possible future WG1 activities.

To facilitate the process of deriving methodological recommendations from the methodology overviews in tabular form (Ernst&Young and Quantis, 2010; De Camillis et al., 2011), the WG1 drafting group came up with a set of key methodological issues on which the ENVIFOOD Protocol is expected to give guidance (Schenker et al., 2011). These methodological issues were identified by: analysing the methodology overviews in tabular form; scanning the key findings from the critical reviews of environmental assessment case studies in the agri-food sector; and taking into account inputs from the technical report on data gaps.

Methodologies and guidelines from the methodology overviews in tabular form (Ernst&Young and Quantis, 2010; De Camillis et al., 2011) were then scanned against the methodological issues identified by WG1 drafting group (Schenker et al., 2011). The purpose of this further analysis (De Camillis et al., 2011b) was to identify which recommendations/rules diverge across the reference documents analysed and, thus, signal where guidance must be harmonised in the ENVIFOOD Protocol. Particular emphasis was given to those methodologies and guidelines that have somehow involved stakeholders in their own development process.

A lack of consensus on several methodological aspects was found across the methodologies reviewed (De Camillis et al., 2011b). In order to analyse those aspects where consensus does not exist at present and to identify specific methodological approaches to be incorporated in the ENVIFOOD Protocol, the JRC IES hosted another workshop in 2011. The workshop was relatively successful because several approaches for assessment were agreed by Round Table members (De Camillis et al., 2012a, 2012b).

Nevertheless, some issues remained and they were discussed within WG1 before coming out with a draft ENVIFOOD Protocol.

The ENVIFOOD Protocol is not a self-supporting guide. Rather, it provides additional guidance to ISO 14044, specific to food and drink products. Moreover, there is the intention of alignment with the European Commission's methodologies on Environmental Footprint.

The Protocol gives recommendations and guidance on functional unit, system boundaries, data requirements and how to deal with data gaps. In addition, it also provides guidance on how to handle multifunctional processes and how to identify relevant environmental impact categories.

Given that the ENVIFOOD Protocol is intended to be applicable to the entire agri-food sector and hence impact on multiple stakeholders, it is important that the input of scientific and food sector experts guide refining the current draft version of the Protocol. Towards this end, a public consultation will be launched in the last quarter of 2012. In parallel, the Protocol will be tested with the support of business associations or individual companies. According to the feedback collected from both the public consultation and pilot tests, the ENVIFOOD Protocol will be revised in 2013.

In the coming years, the Round Table members may play a central role in the development/adoption of PCRs addressing food and drink products. As specific issues at product level cannot be directly addressed by the Protocol, Product Category Rules (PCRs) may complement the Protocol by providing further detailed guidance where necessary. This might include, for example, rules for solving multifunctionality problems (allocation). Such detailed guidance should contribute to the establishment and diffusion of a single coherent methodology for environmental assessment of specific food and drink products. Future mandates on the development/adoption of PCRs are to be further discussed and agreed by Round Table members.

3. Conclusions

The European Food Sustainable Consumption and Production Round Table works for the harmonisation of the assessment methodologies for the agri-food sector. This paper describes the process implemented so far to develop the ENVIFOOD Protocol. In addition, this paper represents an evidence of the collaboration established between the Round Table and the wider scientific community. We take in fact this opportunity to invite the scientific community, food sector experts and other stakeholders to join the public consultation process of the Protocol that is taking place in the last quarter of 2012.

Inputs from the wider scientific community are also welcome when supporting companies in the testing phase due to be launched in early 2013. More information on the public consultation and testing phase of the Protocol is available on www.food-scp.eu

At present, Round Table members are discussing how to obtain adequate data for assessment and streamlined tools. The establishment of the ENVIFOOD Protocol and PCRs will allow the development of user-friendly and affordable tools for assessment and communication of environmental impacts, thus reducing the burdens of assessments, particularly for SMEs.

Future further interactions with the Round Table will be possible if PCRs and other deliverables are drafted and made available for consultation.

4. References

Peacock, N., De Camillis, C., Pennington, D., Aichinger, H., Parenti, A., Rennaud, J.-P., Raggi, A., Brentrup, F., Sára, B., Schenker, U., Unger, N. and Ziegler, F., 2011. Towards a harmonised framework methodology for the environmental assessment of food and drink products. Int. J. Life Cycle Ass. 16(3):189-97.

Food RT, 2010. Guiding Principles. Voluntary environmental assessment and communication of environmental information along the food chain, including to consumers.The European Food Sustainable Consumption and Production Round Table, Brussels, Belgium.

Ernst&Young and Quantis, 2010. Product carbon footprinting – a study on methodologies and initiatives. Final report for DG Environment, European Commission.

De Camillis, C., Unger, N., Schenker, U., Blok, M.C., Blomsma, C., Vessia, O., Blonk, H., 2011a. Detailed analysis of environmental assessment methodologies and guidelines. Technical report. Deliverable (No.) 1.6 Part 1: Methodology overview. The European Food Sustainable Consumption and Production Round Table, Working Group 1, Brussels, Belgium.

De Camillis, C., Schenker, U., Unger, N., Blomsma, C., Blonk, H., Bligny, J.-C., 2011b. Detailed analysis of environmental assessment methodologies and guidelines. Technical report. Deliverable (No.) 1.6 Part 3: Analysis of methodological convergences. The European Food Sustainable Consumption and Production Round Table, Working Group 1, Brussels, Belgium.

De Camillis, C., 2011. Data gap analysis. Technical report. Deliverable (No.) 1.5. The European Food Sustainable Consumption and Production Round Table, Working Group 1, Brussels, Belgium.

Schenker, U., Bauer, C., Unger, N., Bligny, J.-C., De Camillis, C., 2011. Detailed analysis of environmental assessment methodologies and guidelines. Technical report. Deliverable (No.) 1.6 Part 2: key methodological issues. The European Food Sustainable Consumption and Production Round Table, Working Group 1, Brussels, Belgium.

De Camillis, C., Bligny, J.-C., Pennington, D., Bauer, C., Ponsioen, T., Schenker, U., Sueys, R., Unger, N., Lundquist, L., Brandão, M., Hamill, S., Peacock, N., Pelletier, N., 2012a. Towards the ENVIFOOD Protocol: deriving scientifically-sound rules from existing methodological alternatives. Summary report of the second scientific workshop of the European Food Sustainable Consumption and Production Round Table Working Group 1. Publications Office of the European Union, Luxemburg.

De Camillis, C., Bligny, J.-C., Pennington, D., Pályi, B., 2012b. Outcomes of the second workshop of the Food Sustainable Consumption and Production Round Table Working Group 1: deriving scientifically sound rules for a sector-specific environmental assessment methodology. Int. J. Life Cycle Ass. 17(4):511-5.

Water footprint of pastoral farming systems in New Zealand

Marlies A. Zonderland-Thomassen[1*], Stewart F. Ledgard[1], Mark Lieffering[2]

[1]AgResearch Ltd, Private Bag 3123, Hamilton 3240, New Zealand
[2]AgResearch Ltd, Palmerston North, New Zealand
[*] Corresponding author. E-mail: *marlies.zonderland-thomassen@agresearch.co.nz*

ABSTRACT

Global livestock production has a major challenge ahead in meeting increased food requirements without increasing the environmental burden. A need exists to assess the impacts of water consumption associated with New Zealand (NZ) livestock products. A water footprint (WF) approach compliant with Life Cycle Assessment (LCA) principles was used to assess the stress-weighted WF using a water stress index. The stress-weighted WF was 0.01 and 7.1 L H_2O-eq/kg fat-and-protein-corrected milk (FPCM) for dairy farming in the Waikato and Canterbury regions respectively. The stress-weighted WF of NZ beef was 0.2 L H_2O-eq/kg live weight (LW), whereas for sheep meat it was 0.1 L H_2O-eq/kg LW. Water consumption associated with irrigation dominated the WF of dairy farming in the Canterbury region, as well as the WF of beef and sheep meat, which highlights the importance of targeting water use efficient practices in irrigated regions.

Keywords: water footprint, water stress index, eutrophication potential, pastoral farming

1. Introduction

Global livestock production has a major challenge ahead in securing food production without increasing the environmental burden. Water consumption and pollution are associated with a range of activities, and at a global scale, most of the water use occurs in agricultural production (Mekonnen and Hoekstra, 2012). Agriculture is a water-intensive human activity and water availability can be limited in certain areas and seasons. Globally, a key agricultural system is dairying and New Zealand (NZ) is the world's largest dairy exporting country trading at least 95% of the 16,500 million litres of milk that was processed in NZ in 2010, equal to 26% of NZ's total export revenue. Milk production is inevitably linked with water consumption for animal feed production (Neal et al., 2011) and can be a significant contributor to water pollution (Monaghan et al., 2007; Ledgard et al., 2009). Sheep and beef farming is also dominant in NZ agriculture. Red meat and related products contributed NZ$6.5 billion or 15% of NZ's total export revenues in 2009. It is important that NZ milk and red meat suppliers are prepared with information on the water footprint (WF) of their products. This will enable them to understand the potential for reducing their WF and achieving a comparative advantage relative to products from other countries.

2. Methods

A literature review showed various WF approaches exist or are under development (e.g., Bayart et al., 2010; Berger and Finkbeiner, 2010). For the red meat and dairy supply chains, both water quality and water quantity are relevant. A WF approach compliant with Life Cycle Assessment (LCA) principles was used to assess the stress-weighted WF following Ridoutt et al., (2010). The eutrophication potential (EP) of NZ dairy, beef and sheep farming systems (Guinée et al., 2002) was also assessed.

New Zealand red meat is derived from sheep and cattle that are raised under a variety of mixed production systems across a range of climatic conditions. To deal with variation in production systems, survey data from seven representative sheep and beef farm systems from throughout NZ were used (B+L NZ statistics 2009/2010). These data were used to calculate a weighted average sheep and beef farm representative of beef and sheep meat production for NZ. Across the seven farm classes, the average total effective land area ranged from 245 to 8,872 ha for class 7 (South Island intensive finishing) to class 1 (South Island high country), respectively (Table 1). The average rate in stock units for beef cattle and sheep per effective hectare ranged from 1.1 (class 1) to 10.6 (class 7). The predominant source of feed for sheep and beef animals was grazed pasture. Low levels of feed supplements made on-farm (ranging between 0.4 kg DM/ha in farm class 1 and 292 kg DM/ha in farm class 7) were fed to the animals to overcome pasture shortages in summer or winter and to optimise production.

For dairy farm systems, data from the two contrasting regions of Waikato (North Island, non-irrigated, moderate rainfall) and Canterbury (South Island, irrigated, lower rainfall) were used (Zonderland-Thomassen and Ledgard, 2012). The annual rainfall was 677 mm in Canterbury compared to 1264 mm in the Waikato region (Table 1). Most Canterbury farms, therefore, apply irrigation between spring and autumn (on average 565 mm/ha/year).

Table 1. Farm characteristics of the dairy farm, and beef and sheep systems

Parameters	Unit	Dairy Waikato	Dairy Canterbury	Beef and sheep NZ
Grassland area	ha	94.6	200.2	245 - 8,872
Stocking rate	SU[1]/ha	2.96	3.11	1.1 – 10.6
Feed supplements	kg DM[2]/ha	798	1,892	0.4 - 291.5
Irrigated area	% of total ha	0	71	0 – 0.5
Rainfall	mm/yr	1264	677	688 - 1500

[1] Stock units in dairy cows or sheep and beef cattle equivalants e.g., based on 550 kg DM intake for a mature sheep
[2] Dry Matter

Although annual dry matter intake from pasture by animals on farm was similar for the two systems (11,670 kg DM/ha for the Waikato system and 11,260 kg DM/ha for the Canterbury region), the Canterbury dairy system used more brought-in supplementary feed (1,892 kg DM/ha vs. 798 kg DM/ha) and relied on cows grazing off-farm over winter, which resulted in a higher annual milk production (13,183 L milk/ha vs. 10,514 L milk/ha).

Water losses associated with evapotranspiration from irrigated pasture, as well as nitrate leaching and phosphate runoff (from all soluble and soil-adsorbed sources) were computed using the hydrological sub model in the OVERSEER® nutrient budget model (Wheeler et al., 2003).

We used nitrate and phosphate loss when computing the water quality indicator “eutrophication potential” (EP), while excluding gaseous emissions, as preliminary results showed that ammonia and nitrous oxide emissions dominated the EP which is questionable for NZ conditions. Data are expressed in phosphate equivalents (PO_4^{3-}-eq).

The cradle-to-farm-gate life cycle required for the production of milk, beef, and sheep meat were analysed: for the production of inputs to products leaving the farm-gate, i.e. excluding transport or processing of raw milk or animals. Water abstraction and consumption associated with the production of machinery, buildings, and medicines were excluded. Economic allocation was applied when dividing the WF between milk and meat. Biophysical allocation based on feed intake was used when dividing the WF between beef cattle and sheep, while economic allocation was used when dividing the WF for sheep between meat and wool.

In the impact assessment, the water stress index (WSI) following Pfister et al., (2009) was applied to get a stress-weighted WF. Data were normalised against the global WSI and expressed in H_2O-equivalents (Ridoutt and Pfister, in press).

3. Results

The stress-weighted WF of NZ beef (excluding beef from culled dairy cows) was 0.2 L H_2O-eq/kg live weight (LW) (Table 2). Blue water losses from the grazed system were low and consequently the main losses were associated with bull calf rearing (57%) and blue water losses associated with evapotranspiration from irrigated pasture (36%). The stress-weighted WF of NZ sheep meat was 0.1 L H_2O-eq/kg LW. Table 2 shows that the stress-weighted WF was 0.01 and 7.1 L H_2O-eq/kg fat-and-protein-corrected milk (FPCM) for the Waikato and Canterbury farm systems respectively. Water consumption associated with irrigation dominated the WF of the Canterbury dairy system, whereas water consumption associated with hydro-electricity supply was a hotspot in the WF of the Waikato dairy system.

Table 2. The stress-weighted water footprint (WF) of beef and sheep meat (based on a NZ weighted average), and milk produced in the Waikato and Canterbury regions respectively.

L H_2O-eq/kg unit	Unit	WF
Beef (NZ weighted average)	LW[1]	0.2
Sheep meat (NZ weighted average)	LW	0.1
Milk in Waikato region	FPCM[2]	0.01
Milk in Canterbury region	FPCM	7.1

[1] Live Weight sold
[2] Fat-and-Protein Corrected Milk

The EP of NZ beef (excluding beef from culled dairy cows) was 12.2 g PO_4^{3-}-eq/kg LW (Table 3). The EP of NZ sheep meat was 4.9 g PO_4^{3-}-eq/kg LW. Nitrate and phosphate loss at the farm (71% and 75% respectively) and the eutrophying pollutants to waterways associated with the production of artificial fertilisers (20% and 22% respectively) dominated the EP. Table 3 shows that the EP was 1.94 g PO_4^{3-}-eq/kg FPCM and 1.49 g PO_4^{3-}-eq/kg FPCM for the average Waikato and Canterbury dairy farm systems respectively. Nitrate and phosphate loss within the dairy farm (94-98%) dominated the EP of milk in these two regions.

Table 3. The eutrophication potential excluding gaseous emissions (EP) of beef and sheep meat (based on a NZ weighted average), and milk produced in the Waikato and Canterbury regions respectively.

g PO_4^{3-}-eq/kg unit	Unit	EP
Beef (NZ weighted average)	LW[1]	12.2
Sheep meat (NZ weighted average)	LW	4.9
Milk in Waikato region	FPCM[2]	1.94
Milk in Canterbury region	FPCM	1.49

[1] Live Weight sold
[2] Fat-and-Protein Corrected Milk

4. Discussion

A stress-weighted water footprint accounts for differences in water scarcity between the regions. The stress-weighted WF of Australian beef produced in six different geographically defined production systems varied between 3.3 and 221 litres per kg live-weight (Ridoutt et al., 2012), which was higher than the stress-weighted WF of NZ beef (0.2 L H_2O-eq/kg LW). When converting live-weight into meat, using the factor that animals contain approximately 40% of meat, this stress-weighted WF of NZ beef (0.51 L H_2O-eq/kg meat) was lower than the stress-weighted WF of beef produced in England (which was in the range of 15.1 - 20.0 L H_2O-eq/kg meat, dependant on production system) (EBLEX, 2010). The stress-weighted WF of NZ sheep meat (0.25 L H_2O-eq/kg meat) was also lower than the stress-weighted WF of sheep meat produced in the UK (which was in the range of 8.4 -23.1 L H_2O-eq/kg meat, dependant on production system) (EBLEX, 2010). The WSI of NZ regions where livestock was produced varied between 0.01 and 0.013, whereas the spatially averaged WSI for England was 0.27. However, the distribution of the livestock was not uniform, and the weighted WSI for beef cattle was estimated at 0.19 (T. Hess, personal communication). The low stress-weighted WF illustrates the benefits of NZ beef and sheep meat produced in regions with low water stress levels from a possible marketing perspective.

The EP of the average Waikato and Canterbury dairy farm systems (1.9 g PO_4^{3-}-eq/kg FPCM and 1.5 g PO_4^{3-}-eq/kg FPCM respectively) were much lower compared to the EP of organic (670 g PO_4^{3-}-eq/kg FPCM) and conventional milk (1080 g PO_4^{3-}-eq/kg FPCM) produced in the Netherlands (Thomassen et al., 2008). This is mainly due to a difference in methodology, as the Dutch study included gaseous emissions.

Choice of allocation had little effect on the water footprint results. Economic allocation was applied when dividing the water footprint between milk and meat. If biophysical allocation had been used it would have decreased the stress-weighted WF by 7% and 11% for dairy farming in the Waikato and Canterbury respectively.

The stress-weighted WF of 0.14 L H_2O-eq/kg milk solids for Waikato dairy farming was nearly one thousand of the 94 L H_2O-eq/kg milk solids for Canterbury dairy farming. The Waikato dairy WF was 10-fold lower while the Canterbury dairy WF was 6.5-fold higher than the 14.4 L H_2O-eq/ kg total milk solids reported for non-irrigated South Gippsland, Australia (Ridoutt et al., 2010). This highlights the benefits, from a water consumption perspective, of milk production in non-irrigated regions and of targeting water use efficiency practices to dairy farming in irrigated regions.

5. Conclusion

We can conclude that the stress-weighted WF is a useful indicator to assess the impact of pastoral farming systems on freshwater availability and EP is a feasible indicator to assess water degradation impacts of pastoral farming systems, mainly resulting from leaching or runoff of eutrophying pollutants to waterways when the gaseous emissions (such as ammonia, nitrous oxide and nitrogen oxides) are excluded. New Zealand has large regions with low water stress, although seasonal droughts can occur in many areas. From an international marketing perspective, beef and sheep meat produced in NZ, as well as milk produced in the rain-fed region Waikato have a possible advantage for water consumption compared to overseas pastoral farming systems.

The impact of NZ pastoral farming on freshwater availability can be reduced by practices that decrease water use, increase feed conversion efficiencies, increase the use of non-irrigated feed supplements, and reduce irrigation needs. The impact of NZ pastoral farming on water quality can also be reduced by efficient nutrient management. Other water quality impacts of pastoral farming are relevant to consider in future studies, e.g., the impact of microbial pollution on waterways.

References

Bayart, J-P., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., Koehler, A., 2010. A framework for assessing off-stream freshwater use in LCA. International Journal of LCA. 15, 439-453

Berger, M., Finkbeiner, M., 2010. Water Footprinting: How to address water use in Life Cycle Assessment? Sustainability 2, 919-944

B+L NZ Statistics 2009/2010. Survey data of farm classes throughout NZ provided by Beef + Lamb NZ to AgResearch in 2011/2012.

EBLEX, 2010. Testing the water. The English beef and sheep production environmental roadmap, phase 2.www.eblex.org.uk

Guinée, J. B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. A., de Bruijn, H., van Duin, R., Huijbregts, M. A. J., Lindeijer, E., Roorda, A. A. H., van der Ven, B. L., Weidema, B .P., 2002. Handbook on Life Cycle Assessment; Operational Guide to the ISO Standards. Centre of Environmental Science – Leiden University (CML); Kluwer Academic Publishers., Leiden, the Netherlands

Ledgard, S.F., Schils, R., Eriksen, J., Luo, J., 2009. Environmental impacts of grazed clover/grass pastures. Irish Journal of Agricultural Research 48, 209-226

Mekonnen, M.M., Hoekstra, A.Y., 2012. A global assessment of the water footprint of farm animal products, Ecosystems, 15(3), 401–415

Monaghan, R.M., Wilcock, R.J., Smith, L.C., Tikkisetty, B., Thorrold, B.S., Costall, D., 2007. Linkages between land management activities and water quality in an intensively farmed catchment in southern New Zealand. Agriculture, Ecosystems and Environment 118, 211-222

Neal, J.S., Fulkerson, W.J., Hacker, R.B., 2011. Differences in water use efficiency among annual forages used by the dairy industry under optimum and deficit irrigation. Agricultural Water Management 98 (5), 759-774

Pfister, S., Koehler, A., Hellweg, S. 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environmental Science & Technology 43, 4098-4104

Ridoutt, B.G., Williams, S.R.O., Baud, S., Fraval, S., Marks, N. 2010. Short communication: the water footprint of dairy products: Case study involving skim milk powder. Journal of Dairy Science 93, 5114-5117

Ridoutt, B.G., Sanguansri, P., Freer, M., Harper, G.S., 2012. Water footprint of livestock: comparison of six geographically defined beef production systems. International Journal of LCA 17, 165-175

Ridoutt, B.G., Pfister, S. In Press. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. International Journal of LCA.

DOI 10.1007/s11367-012-0458-z

Thomassen, M.A., van Calker, K.J., Smits, M.C.J., Iepema, G.J., de Boer, I.J.M. 2008. LCA of conventional and organic milk production in the Netherlands. Agricultural Systems 96, 95-107

Wheeler, D.M., Ledgard, S.F., De Klein, C.A.M., Monaghan, R.M., Carey, P.L., McDowell, R.W., Johns, K.L., 2003. OVERSEER® nutrients budgets- moving towards on-farm resource accounting. Proceedings New Zealand Grassland Association 65, 191-194

Zonderland-Thomassen, M.A., Ledgard, S.F., 2012. Water footprinting – a comparison of methods using New Zealand dairy farming as a case study. Agricultural Systems 110, 30-40.

Water footprint accounting of organic and non-organic strawberries including ancillary materials: a case study

Manzardo Alessandro*, Anna Mazzi, Monia Niero, Sara Toniolo, Antonio Scipioni

CESQA (Quality and Environmental Research Centre), University of Padova, Department of Industrial Engineering, Via Marzolo 9, 35131 Padova, Italy
* Corresponding author. E-mail: *alessandro.manzardo@unipd.it*

ABSTRACT
Fresh water scarcity has become a main issue debated at international level. In order to support a better management and use of water, the methodology of Water Footprint has been developed. In this paper the Water Footprint of a 330g Strawberry Jam produced within Europe and sold in Italy is presented. The objectives of the research were: 1) to compare the Water Footprint accounting of two different strawberries farming (organic and non-organic); 2) to determine whether the contribution of the ancillary materials and processes to the overall product water footprint was significant. The water footprint accounting of organic farming resulted to be bigger than the non-organic one. Moreover ancillary materials and processes resulted responsible for over the 10% of the overall product water footprint. Therefore, to support the choices and decisions in the field of water resources management, it is important to consider also ancillary materials and processes.

Keywords: Water Footprint accounting, Organic Food, Strawberry

1. Introduction

Climate Changes, world population growth and economic development heavily affected accessibility to freshwater resources in the last decades (UNESCO, 2006; Rockström, 2009). The public interest and concerns on this issue has constantly grown in the recent year so that freshwater scarcity has been debated at local, national and international level (Bates et al., 2008; UNEP, 2011): United Nations included water availability and water access in the “The Millennium Development Goals” (UN, 2010) the European Union is working on a new water policy that will be published within 2012 named “Blueprint to Safeguard Europe's Water Resources” (EU, 2012); moreover a growing number of companies want to be aware of the environmental impacts of their choices, including water use (Ercin et al., 2012; Chapagain and Orr, 2010).

To support the definition of strategies to better manage these resources, the concept and assessment methodology of Water Footprint accounting has been introduced as an indicator of freshwater use (Hoekstra, 2003; Hoekstra et al., 2011).

Latest developments of this methodology are focused on water use impact assessment of products and organisations within the framework of a more comprehensive Life Cycle Assessment (LCA) (ISO 2006, ISO 2011).

Many studies on food and agricultural products have been published accounting water footprint indicator but only a few include ancillary materials and processes (overheads), such as packaging, transports, energy etc. although these are usually considered in LCA studies (Ridoutt et al., 2009, Mila I canals et al., 2010; Ercin et al., 2012; Chapagain and Orr, 2010). These materials and processes have usually non-relevant contribution to the overall Water Footprint accounting of a product (Ercin et al., 2012; Ridoutt et al., 2009, Mila I canals et al., 2010; Ercin et al., 2012; Chapagain and Orr, 2010).

In this study, conducted in 2011, a water footprint of a jar of 330g organic Strawberry Jam produced within Europe and sold in Italy by a food Italian company is presented. The objectives of the research were:

1) to compare the water footprint accounting of organic and non-organic strawberry farming;

2) to determine if the contribution of the ancillary materials and processes (overheads) to the overall product water footprint can be significant.

2. Methods

To achieve the objectives of the research the methodology presented by the Water Footprint Network has been used (Hoekstra et al., 2011) adopting a life cycle approach.

This study is structured in the following steps: 1) goals and scope of the study; 2) water footprint accounting of the organic and traditional strawberry farming; 3) water footprint of the 330g organic Strawberry Jam. The blue, green and grey water footprint of the processes and product under study were assessed. The water footprint accounting of the organic farming was used as basic building block to determine the final product water footprint.

A life cycle approach was adopted to determine the boundaries of the study in order to include the ancillary materials and processes. Results at product level have also been characterised using Water Stress Index

(WSI) (Pfister at al., 2009). Most of the input data were primary data, obtained from the Italian company records.

3. Results

3.1. Goals and scope of study

The goals of this water footprint study were to quantify:

1. the water footprint accounting of two alternative strawberry farming (non-organic and organic). The strawberries are produced in one site in Bulgaria owned by the Italian company that produces the jam. Primary data of the planting and harvesting period were collected.
2. the direct and indirect water footprint of a 330g organic strawberry Jam produced by the Italian company. A cradle to gate perspective was adopted, considering all the processes and materials that contribute to the production of the product. Secondary distribution, use and end of life were excluded.

The water footprint accounting of the alternative farming is expressed in l/kg of strawberries.

The functional unit used to represent the product water footprint is 330g of organic strawberry jam sold in Italy. Data used in the study refers to a 15 months period between 2009 and 2010: from strawberry plantation to final product primary distribution. The Italian company owns and manages the farming site in Bulgaria and the production and distribution sites in Italy. Table 1 reports the main ingredients contained in 1 kg of organic strawberry jam.

Table 1. Main ingredients of the product under study

Ingredient	Quantity (grams)
Organic Strawberries	600
Organic Apple pulp	380
Pectin	20

Figure 1 represents the boundary of the production system of the 330g Organic Strawberry jam under study. The diagram shows only the major production steps included in the study.

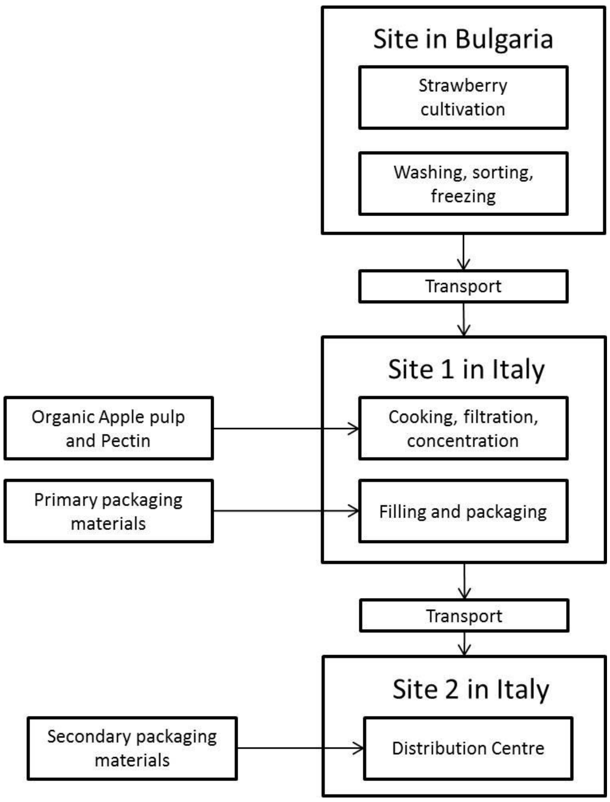


Figure 1. Boundary of the production system under study

3.2. Water Footprint accounting of organic vs. non-organic strawberry farming

Non-organic and organic strawberry farming were analysed in this study. The production site is located in Bulgaria. Its extension is 47 ha; 34 ha are dedicated to the production of organic strawberries that are used to produce the organic strawberry jam; 13 ha are dedicated to the production of non-organic strawberries used for other products. Both of the productions are irrigated (drip irrigation) and open air. The two farming methods differ only on fertilisers and pesticides. Table 2 reports the main characteristics of the farming.

Table 2. Main characteristics of the product under study

Fruit	Planting	Harvesting	Yield (ton/ha)
Organic strawberries	July year 1	June year 2	20
Non-organic strawberries	July year 1	June year 2	26

The green and blue water footprint of the crops are obtained using CROPWAT model (Allen et al., 1998; FAO, 2009). Climate data are obtained from a climate station located in the field. Crop parameters were obtained from Allen et al., (Allen et al., 1998). Yield data were obtained from site records. All data were used in CROPWAT to obtain the crop water use of organic and non-organic strawberries. According to the model presented by Mila I canals and Ridoutt (Mila I canals et al., 2009; Ridoutt et al., 2009), primary data of irrigation water volume were used.

The grey water footprint of non-organic farming took into consideration the impact of the use of chemical (such as fertilisers and pesticides). In the case of organic farming according to EU regulation (EC, 2007) only natural fertiliser are used (compost). Data on the quantity and quality of fertilisers were given directly by the Italian company. The leaching rate is taken from literature and assumed to be 10% with a N limit of 10 mg/l (Ercin et al., 2012). In the case of organic farming, nitrogen from compost contributed to grey water footprint, phosphorus resulted negligible. Table 3 reports the blue, green and grey water footprint of the two farming.

Table 3. Water Footprint accounting of organic and non-organic farming

	Non-organic farming	Organic farming
Green water (l/kg)	90.7	117.9
Blue water (l/kg)	98.1	127.5
Grey water (l/kg)	40	2.5
Total (l/kg)	**228.8**	**247.9**

3.3. Water Footprint of the 330g organic Strawberry Jam

The product water footprint accounting of a 330g organic strawberry jam was then quantified by summing the water used for all the raw materials, ancillary materials and processes included in the study. The blue, green and grey water of these processes were also quantified: washing, sorting and freezing of the strawberries in Bulgaria; transportation of the strawberries to Site 1 in Italy were the jam production processes take place; farming of organic apples and pulping processes; production of pectin; transportation of apple and pectin to Site 1; production of primary and secondary packaging and their transport to Site 1 and Site 2; filling and packaging processes; transportation of end-product to the Site 2 for distribution. The water footprint accounting of energy mix used in the processes is also quantified using the method developed by Mila I Canals (Mila I Canals et al., 2009). The water footprint accounting relevant to the processes of production site waste management, secondary distribution, product use and end of life are not included.

All data on the materials and processes considered in the study are obtained directly from the food company and its suppliers. All the factories have waste-treatment plants that discharge water with effluent characteristics below the legal limits. Therefore it is assumed that grey water footprint is equal to zero. The green water of wood and paper based products used as primary (labelling) and secondary packaging material was assessed.

Water accounting indicator result were characterised using WSI. Therefore the Product Water Footprint resulted to be 88.4 l. Table 4 shows the water footprint of the 330g organic Strawberry jam

Table 4. Water Footprint of the 330g organic Strawberry jam

	Water Footprint accounting (l/functional unit)	Water Footprint (l/functional unit)
Primary Packaging	7.0	1.1
Transport and Secondary packaging	20.8	6.3
Production	9.0	3.0
Farming	200.5	78.0
Total	**237.3**	**88.4**

4. Discussion

The water footprint accounting was conducted to determine the quantity of water used to grow organic and non-organic strawberries in a site in Bulgaria. In this study the organic farming method resulted to be more water intensive then the non-organic one. This result strongly depends on the different yield of the two farming method in this specific production site (Table 2). Other studies confirm the importance of this value when determining the environmental performances of agricultural products (Ercin et al., 2012; Blengini and Busto, 2009). The values of the blue and green water footprint of the two farming methods resulted to be very similar. These results depend on the quantity of water used for irrigation and the effective precipitation. Grey water of non-organic farming is higher because chemicals fertilisers and pesticides (+93.75%) are used. This value do not consider environmental impacts but it is determined through a water dilution approach (Jeswani and Azapagic, 2011)

The product water footprint of a 330g strawberry jam was then quantified. Fig.2 represents the contribution of macro-processes to the overall Water Footprint. The farming processes resulted to be responsible for the majority of the product water footprint (84.49%) then follows transport and secondary packaging, production and primary packaging. The value of the transport and secondary packaging depends on the use of cardboard and wood to pack the final product. Figure 3 represent the contribution of different processes after characterisation using WSI (Pfister et al., 2009). Changes in contribution depend on the% of blue water of the different materials/processes considered. Even in this case farming is still responsible for the majority of the water used. Other processes account for over the 11% of the product overall water footprint.

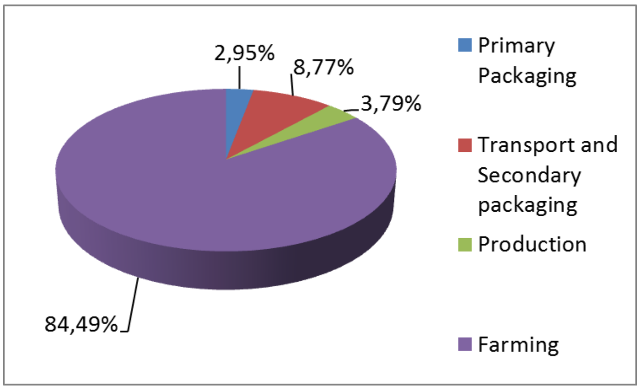


Figure 2. Contribution of processes to the product Water Footprint accounting

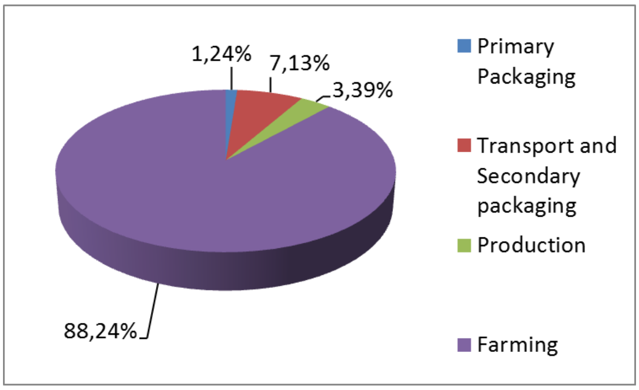


Figure 3. Contribution of processes to the product Water Footprint after characterisation with WSI

5. Conclusion

Fresh water scarcity is a central issue debated at international levels. It affects a growing number of people and companies worldwide. To support the definition of strategies to better manage water resources the concept and the methodology of Water Footprint has been introduced.

Food and agriculture are recognized to be water intensive sectors. Therefore many studies have been published to determine the water footprint of many food products. Most of these studies adopt life cycle perspective in the definition of processes to be considered in the water footprint quantification, but consider ancillary materials and processes (such as energy use or transportation) to be non-significant.

In this study, conducted in 2011, a water footprint of a 330g organic strawberry jam is conducted. The method used is the one presented by the Water Footprint Network. The objectives of the study were:

- to compare the water footprint accounting of organic and non-organic strawberry farming in cultivated a site in Bulgaria;
- to assess the product water footprint of the 330g strawberry jam and to determine the contribution of ancillary materials and processes (overheads). Primary data were collected from the Italian Food company and its suppliers.

Characterisation was performed using WSI.

In this case study, results showed that organic strawberry farming has higher water use than non-organic strawberry farming. The main reason is the different yield of the two farming method.

The water footprint of the 330g organic strawberry jam resulted to be 88.4 l. Other studies confirm that farming is the process with higher water footprint. Overheads water footprint contribution resulted to be over 10%. The Italian Food company should work mainly on paper based packaging to reduce the water footprint of overheads. These results suggest that even ancillary materials and processes should be considered when looking at strategies to reduce a product water footprint.

6. References

Allen R.G., Pereira L.S., Raes D., Smith M., 1998. Crop Evapotranspiration – Guidelines for Computing Crop Water Requirements FAO Irrigation and Drainage Paper 56. Food and Agriculture Organization, Rome.

Bates, B.C., Kundzewicz Z.W., Wu S., Palutikof J.P., , 2008. Climate Change and Water. Technical Paper of the Intergovernmental Panel on Climate Change. IPCC Secretariat, Geneva.

Blengini G.A., Busto M., 2009. The life cycle of rice: LCA of alternative agri-food chain management systems in Vercelli (Italy). Journ. Env. Man. 90, 1512-1522.

Chapagain A.K., Orr S., 2010. Water Footprint of Nestlè's Bitesize Shredded Wheat. A pilot study to account and analyse the water footprints of Bitesize Shredded Wheat in the context of water availability along its supply chain.

EC (European Council), 2007. Council Regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91.

Ercin A., Aldaya M.M., Hoekstra A.Y., 2011. Corporate Water Footprint Accounting and Impact Assessment: The Case of the Water Footprint of a Sugar-Containing Carbonated Beverage. Wat. Res. Man. 25 (2), 721-741.

Ercin A., Aldaya M.M., Hoekstra A.Y., 2012. The water footprint of soy milk and soy burger and equivalent animal products. Ecol. Ind. 18, 392-402.

FAO, 2009. CROPWAT 8.0 Model. Food and Agriculture Organization, Rome.

Hoekstra A.Y., 2003. Virtual water trade. Proceedings of the International Expert Meeting on Virtual Water Trade. Value of Water Research Report Series No. 12. UNESCO-IHE, Delft, the Netherlands.

Rockström, R., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature 461, 472–475.

Hoekstra A. Y., Chapagain A.K., Aldaya M.M , Hoekstra M.M.M., 2011. The water footprint assessment manual: Setting the global standard. Earthscan.

ISO (International Organization for Standardization), 2006. ISO 14040: Environmental management - Life cycle assessment - Principles and framework. Switzerland, Geneva.

ISO (International Organization for Standardization), 2011. ISO/WD2 14046: Water Footprint requirements and guidelines. Switzerland, Geneva.

Jesvani H.K., Azapagic A., 2011. Water footprint: methodologies and a case study for assessing the impacts of water use. J. of Clean. Prod. 17 (13), 1228-1235.

Ridoutt B.G., Eady S.J., Sellahewa J., Simons L., Bektash R., 2009. Water footprinting at the product brand level: case study and future challenges. J. of Clean. Prod. 19, 1288-1299.

Milà I Canals L., Chenoweth J., Chapagain A., Orr S., Antón A., Clift R., 2009. Assessing freshwater use impacts in LCA: Part I—inventory modelling and characterisation factors for the main impact pathways. Int J Life Cycle Ass. 14 (1), 28-42.

Milà I Canals L., Chenoweth J., Chapagain A., Orr S., Antón A., Clift R., 2010. Assessing freshwater use impacts in LCA, part 2: case study of broccoli production in the UK and Spain. Int J Life Cycle Ass. 15 (6), 598-607.

Pfister S., Koehler A., Hellweg S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Tech. 43 (11), 4098-4104.

UN (United Nations), 2010. The Millennium Development Goals Report. New York.

UNEP (United Nations Environment Programme), 2011. Towards a Green Economy: Pathways to Sustainable Development and Poverty Eradication.
UNESCO-WWAP (United Nations Educational Scientific and Cultural Organization-World Water Assessment Programme), 2006. Water: A Shared Responsibility: The United Nations World Water Development Report 2
EU (European Union), 2012. Blueprint to Safeguard Europe's Water Resources.

Comprehensive database for use in LCA-based water footprinting: new results and findings from case studies

Samuel Vionnet*, Lindsay Lessard, Sebastien Humbert

Quantis, Parc scientifique de l'EPFL, Bât. D, 1015 Lausanne Switzerland
* Corresponding author: E-mail: *samuel.vionnet@quantis-intl.com*

ABSTRACT

There is a rapidly growing demand for information regarding environmental impacts mediated by changes in water quantity and water quality from the perspective of product and corporate footprints. From 2009-2010, Quantis - working with the ecoinvent center and a large number of industry partners such as Molson Coors, L'Oreal, Steelcase, Kraft Foods, Groupe Danone, Veolia Environment, Natura and Unilever - led a collaborative project to create a water footprinting framework and inventory and impact assessment database. This database is now finished and compatible with ecoinvent. These new inventory flows for water are currently being implemented in the upcoming version 3 of the ecoinvent database. This new database identified some of the main limitations in current (traditional) water footprint assessment, especially in terms of grey (i.e., polluted) water assessment, as well as in terms of local competition for consumptive water ("blue water footprint", which competes with other uses).

This presentation will give an overview of the project's results after one year of application and some of the major lessons about assessing grey water and impact assessment associated with water comsumption throughout the supply chain of products and companies. These include examples from industries such as textiles, food and beverage, personal health care, furniture and information technology, among others. Several of these examples have been used within the context of the AFNOR/ADEME French labelling experiment, and key lessons will be presented. The presentation will also show where research and data collection is still needed to have robust water-footprint results so that this metric can be used in a robust way for LCA-based decision making.

Keywords: water footprint, water use in LCA, database

Monthly characterisation factors for water consumption and application to temporally explicit cereals inventory

Stephan Pfister*, Julia Baumann

Ecological Systems Design, Institute of Environmental Engineering (IfU), ETH Zurich, 8093 Zürich, Switzerland
* Corresponding author. E-mail: *pfister@ifu.baug.ethz.ch*

ABSTRACT

We calculated monthly water stress indices for >11000 watersheds with global coverage. In comparison with annual assessments these indices show large differences in many loctions although the general spatial distribution of water scarcity remains quite stable. Analysing wheat and rice with monthly and annual indicators shows that the crop growing period has a considerable influence and shifting crop planting dates (or crops with different calendars) can help to relieve water stress. The main limitation of the improved temporal resolution is the lack of detail in quantifying inter-monthly natural and man-made storage effects. The produced maps and data allow better capturing of water scarcity with temporal resolution, but many issues are raised and further research is required.

Keywords: temporal resolution, rice, wheat, water footprint, regionalisation

1. Introduction

Water scarcity is affecting a large part of the population and will increase in future due to population growth, requiring a shift in spatial distribution of water consumption (Ridoutt and Pfister 2010a). Therefore water footprint and impact assessment of water in LCA has gained wide interest and is heavily discussed (Berger and Finkbeiner 2010), and different methods to quantify the impacts have been created and compared (Kounina et al., 2012). Relevance of spatial resolution for assessing water consumption and related impacts of plant has been highlighted in previous research (e.g. Pfister et al., 2011a). However, the temporal dimension of crop cultivation and related impacts has been neglected so far, although different crop options can shift irrigation water consumption within a year and hence lead to higher or lower water stress in the region. Furthermore, in some regions the temporal dimension is crucial, especially in cases with high variability of water use and availability. Consequently, annual assessment might be misleading in guiding crop choices within and among different regions. Temporal resolution is therefore essential for proper LCA of crop production.

The definition of a water footprint (WFP) has led to confusion as some consider and report pure aggregation of water consumption volumes to be proper WFP (e.g. Hoekstra et al., 2012), while we consider a water footprint according to carbon footprint and LCA including impact assessment as the only useful aggregated number informing about water scarcity issues related to products and services (Pfister and Hellweg 2009, Ridoutt and Pfister 2010b). This is especially relevant for processes in the supply chain, which are often major contributors to the overall WFP, as shown by Feng et al., (2011). A recent report highlighted the similarity of the approaches and revealed the shortcomings of the purely volumetric approach (UNEP 2012).

Several indicators for water scarcity have been developed recently and we focus our development based on the approach of Pfister et al., (2009). For a selection of 405 watersheds Hoekstra et al., (2012) calculated monthly water stress indicators following a different approach from the one we used.

In this work we developed water stress index (WSI) on a monthly basis for more than 11000 watersheds with global coverage. In a second step, WFP are calculated by multiplying monthly WSI ($WSI_{monthly}$) with monthly crop water consumption.

2. Methods

The original, annual WSI based on the approach of Pfister et al., (2009) includes a term for monthly variability of water availability in order to account for increased pressure in watersheds with unstable water supply over time. This factor has been excluded as it is explicitly covered by applying monthly WSI. Only the inter-annual variability is accounted for by the geometric standard deviation (s^*_{year}) of annual precipitation data during the "climate normal period" (1961-1990) within each river basin. Consequently the WSI function on monthly resolution of each watershed is adjusted to the reduced variability factor s* by increasing the exponent factor of -6.4 to -9.8:

$$WSI_{monthly} = \frac{1}{1 + e^{-9.8 \cdot WTA^*_{monthly}} \left(\frac{1}{0.01} - 1\right)} \quad \text{Eq. 1}$$

$$WTA^*_{monthly} = WTA_{monthly} \cdot s^*_{year} \quad \text{Eq. 2}$$

$WTA_{monthly}$ is the monthly withdrawal to availability ratio. It is determined by aggregating data from the 0.5 arc-degree model by Fekete et al., (2002) to watershed level and deriving factors of monthly WTA to annual WTA for each month. In a second step, these monthly factors are applied to the annual data from “WaterGAP” Alcamo et al., (2003) which are used in the original WSI (Pfister et al., 2009) to derive values for $WTA_{monthly}$ that are consistent with the annual factors.
The irrigation water consumption of the crops (IWR) is calculated according to Pfister et al., (2011a) and related to kg harvested crop through location-specific attainable yields (Fischer et al., 2000) in order to derive monthly WFP ($WFP_{monthly}$) and compare to the annual assessment (WFP_{annual}).

3. Results

Fig. 1 shows a comparison of monthly and annual WSI, indicating large differences for some watersheds, while the general trend of spatial distribution is not changed dramatically (Figure 2). Still the variation is relevant especially due to the fact that most water is used in periods where water stress is rather high. This is also a logical consequence of the equation.

For the WFP analysis this has some additional implication, as not only the place of water use but also the timing is crucial for the assessment. This is shown by the case of wheat (Figure 3): In some areas wheat is planted as “winter wheat” and therefore has a growing phase that is prior to most other crops and consequently the average water consumption is rather in months with less water scarcity. Although the difference of annual and monthly water WFP is not changing the pattern of the global maps, it can be seen that in Southern Spain the Winter season is limiting the $WFP_{monthly}$ compared to WFP_{annual}. These effects are better shown in Figure 4, where maps of the ratio $WFP_{monthly}$ and WFP_{annual} shows that for rice and wheat the ratio can be quite different due to different growing seasons, pointing out the relevance of temporal resolution.
The scatter plots (Figure 5) indicate the strong correlation but show relatively high variability in regions with average water footprints, especially for wheat.

These results and relevant data (incl. Google Earth layers) are made available on a webpage (ESD 2012).

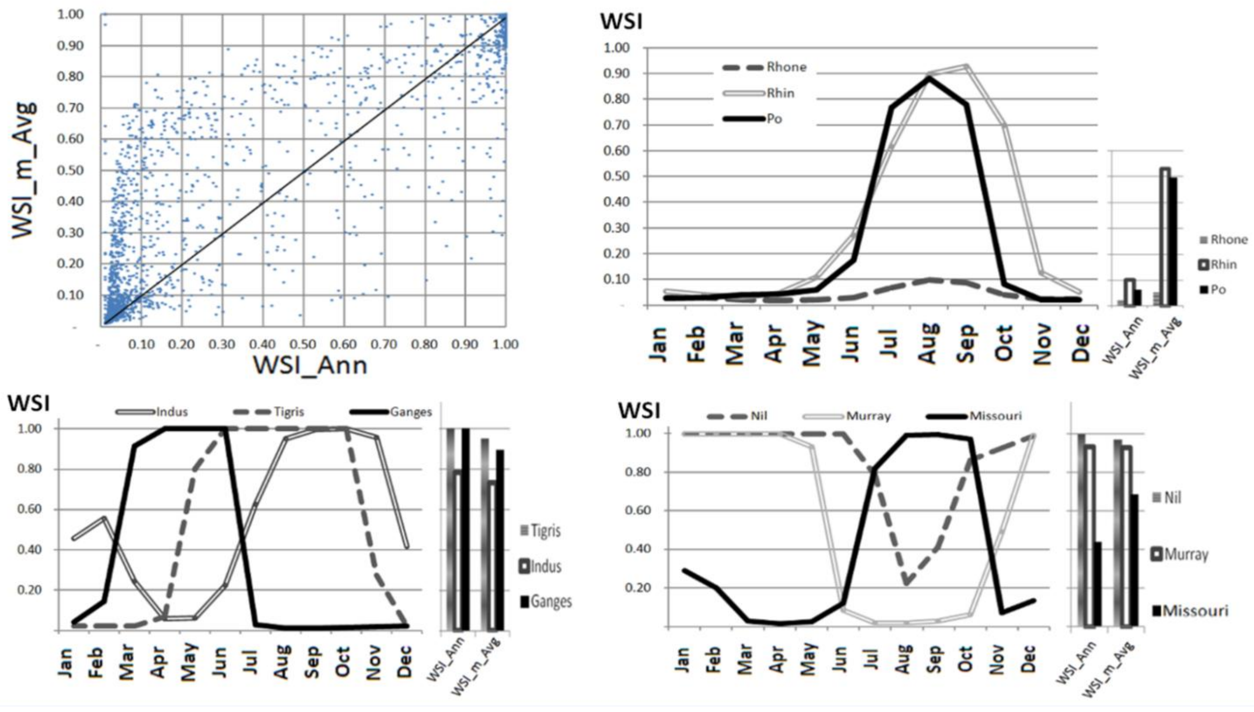


Figure 1. The graphs show an analysis of annual WSI (WSI_Ann) versus water-use weighted annual average of $WSI_{monthly}$ (WSI_m_Avg). The scatter plot shows all values for all watersheds and reveals that WSI_m_Avg is higher for many low WSI_Ann. The remaining graphs show the monthly WSI and the annual values for selected rivers for illustration of the variabilities in many watersheds.

WSI WSI

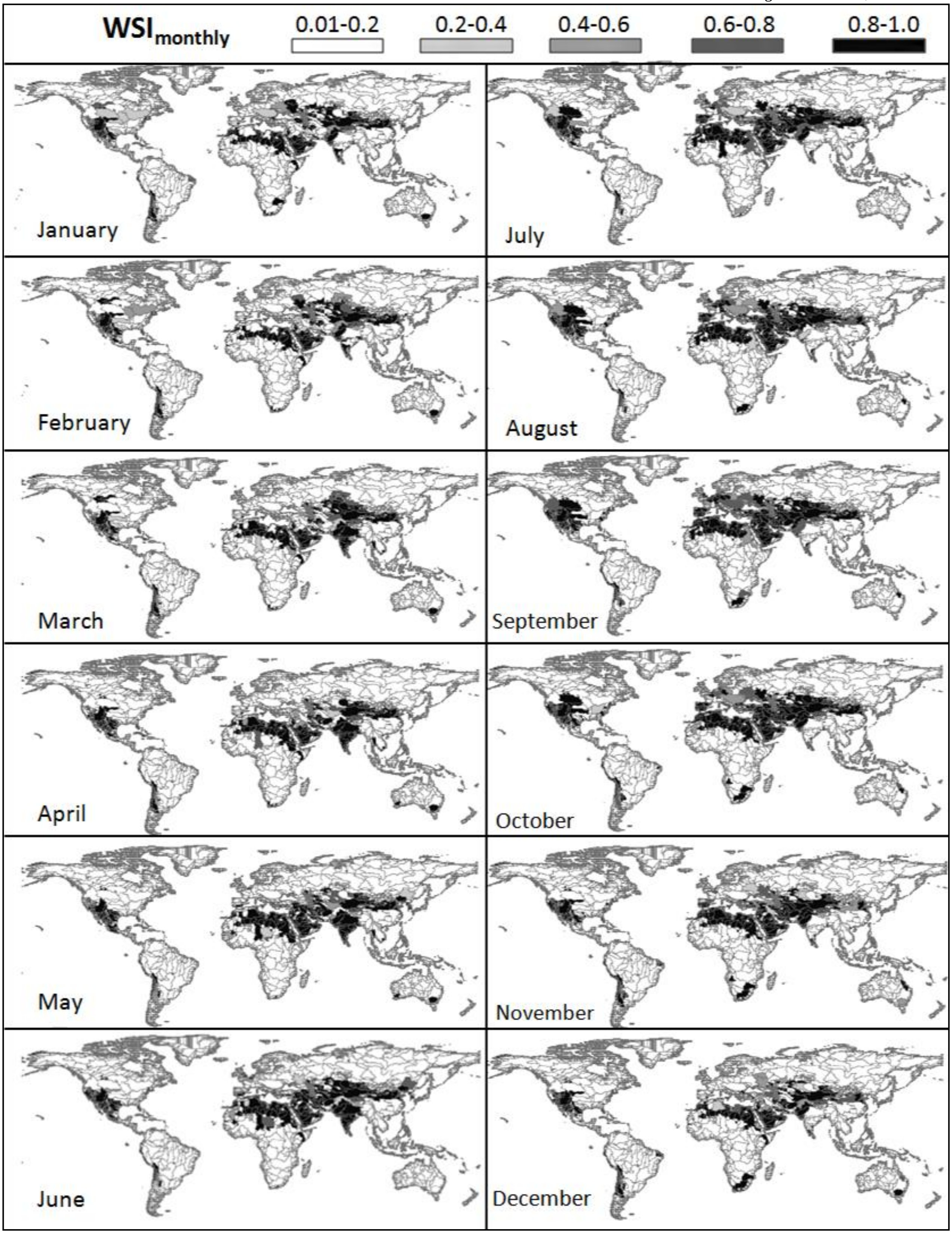


Figure 2. These maps show the monthly WSI for each watershed. In many areas there is no or only minor water stress while some areas consistently experience high water stress. However, some regions such as large parts of Europe have highly variable WSI for different months.

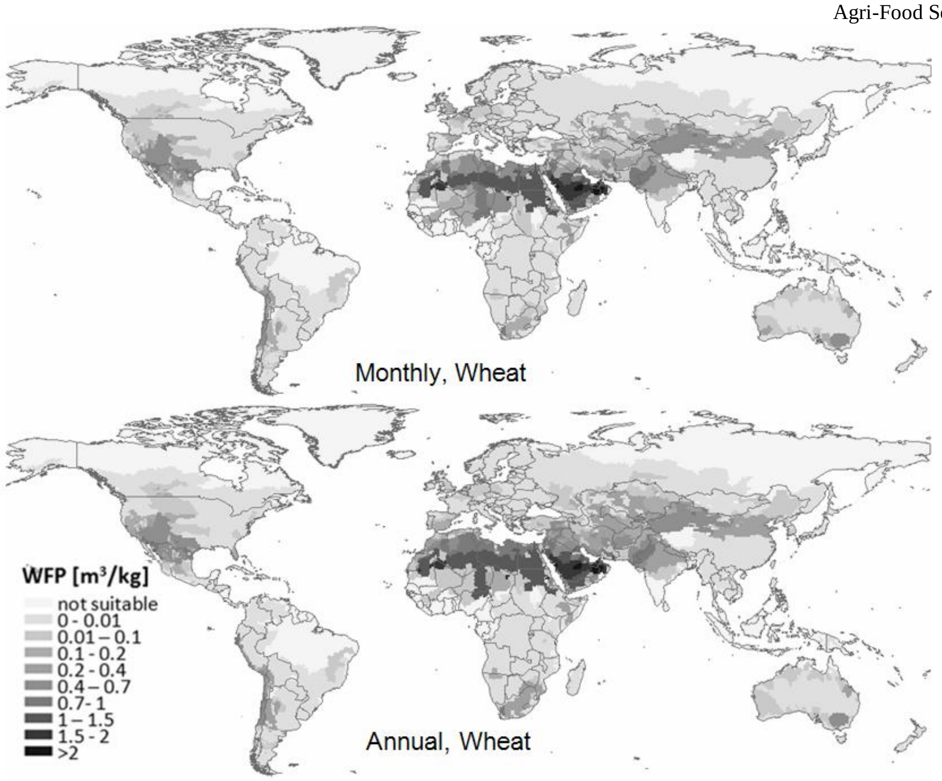


Figure 3. These maps show the water footprint (WFP) of wheat based on monthly (top) and annual (bottom) Assessment. In many areas there is no or only minor differences as the general trend is the same. However some watersheds have significantly different impacts (Fig. 4).

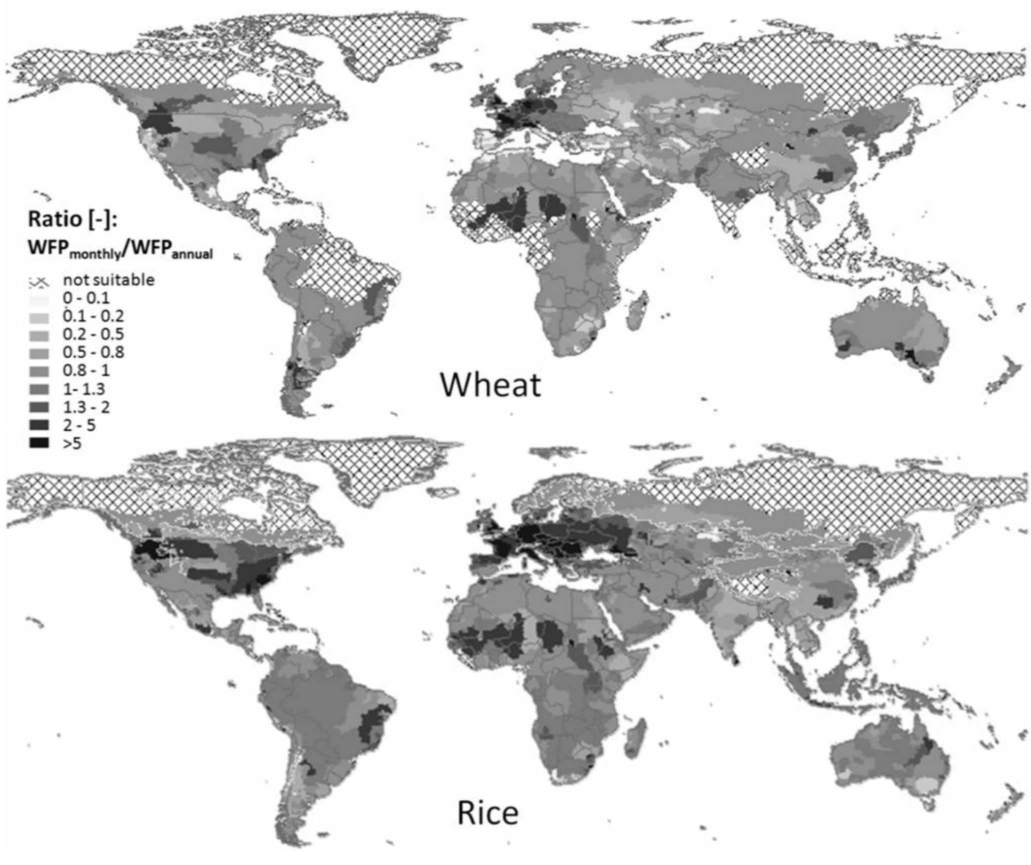


Figure 4. These maps show the ratio monthly to annual water footprint calculation of wheat top and rice (bottom). In many watersheds the impacts vary significantly. Note that although there is a similar pattern, rice and wheat differ regionally due to different growing seasons.

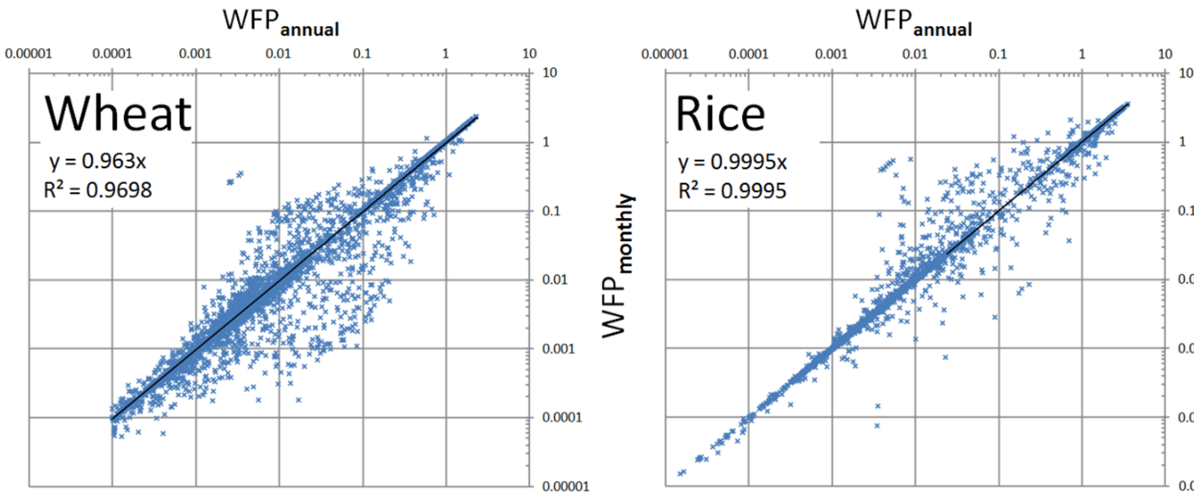


Figure 5. The scatter plots represent individual WFP values for wheat and rice production based on annual (WFP_{annual}) and monthly assessments ($WFP_{monthly}$) for each watershed. The linear regression indicates that monthly WFP are generally lower than WFP_{annual} for wheat, and quite consistent for rice. The high variability for the case of wheat (up to 2 orders of magnitude) reveals the importance of temporal explicit WFP.

4. Discussion

The results presented here are providing additional insights in water scarcity assessment and WFP but are having deficits. Uncertainties have shown to be extremely high in some areas (orders of magnitudes confidence intervals) even on annual WSI (Pfister and Hellweg 2011) and are assumed to be much higher for monthly factors. Additionally the inter-monthly storage needs to be accounted for and groundwater sources have to be distinguished from surface water in order to capture the relevant hydrological features. Reservoirs are also not addressed here. They could significantly change the situation of monthly availabilities, but are causing losses which can be significant as shown for dams for hydropower (Pfister et al., 2011c). All these features are also lacking in the monthly indicators for 405 watersheds by Hoekstra et al., (2012). However, those factors are based on water consumption instead of water use and therefore might capture the quantitative issue better. On the other hand also degradative water use contributes to water scarcity and therefore a mixture of water use and water consumption might be the best basis for a single water stress indicator.

The monthly WFP allows assessing more accurately the water consumption impacts and related management options such as evaluating different crop rotations as discussed in Nunez et al., (2012). The higher detail can also better analyse future crop schemes and reveal potentials for feeding the mankind in 2050. While shifting crop locations might considerably reduce water stress (Pfister et al., 2011b), shifting crop planting dates could mitigate peak water stress periods. This is an important potential as relocation of production sites might be problematic due to supply chain management of existing processing facilities as shown in Chiu et al., (2011) and reluctance of people to move their agricultural activities to completely different places. However, shifting cultivation periods might also reduce yield or increase irrigation water demand and consequently lead to higher water footprints. One limitation of the presented inventory is the exclusion of unproductive irrigation-water losses, which can lead to significant additional water consumption (Faist et al., 2011) and should be considered in future work.

Integration of water quality aspects potentially included in WFP should also consider watershed characteristics and temporal variation for impact assessment of emissions in LCA such as shown for heat emissions (Verones et al., 2011). This is a crucial step for consistently addressing the temporal dimension in LCA and aggregate monthly water footprint figures as suggested by Ridoutt and Pfister (2012). Finally the same temporal issues also concern land occupation as discussed and addressed in Pfister et al., (2010), since occupying land in winter is different than in summer. Beyond this consideration of natural growth seasons, other parameters for temporally-explicit land quality assessment are needed, especially for evaluating food products.

5. Conclusion

Analysing wheat and rice with monthly and annual indicators shows that the crop growing period has a considerable influence and shifting crop planting dates or crops with different calendars can help to relieve water stress. The main limitation of the improved temporal resolution is the lack of detail in quantifying inter-monthly natural and man-made storage effects. The produced maps and data allowing for better capturing water scarcity with temporal resolution, although further improvement is required.

6. References

Alcamo, J., Doll, P., Henrichs, T., Kaspar, F., Lehner, B., Rosch, T., Siebert, S., 2003. Development and testing of the WaterGAP 2 global model of water use and availability, Hydrolog. Sci. J., 48 (3), 317-337.

Berger M, Finkbeiner M (2010) Water footprinting: How to assess water use in life cycle assessment? Sustainability 2:919-944

Chiu, Y-W; Suh, S; Pfister, S; Hellweg, S; Köhler, A (2011) Measuring ecological impact of water consumption by bioethanol using life cycle impact assessment. Int J LCA, DOI 10.1007/s11367-011-0328-0

ESD (2012). Water inventory data. http://www.ifu.ethz.ch/ESD/downloads/WATER_DATA

Faist Emmenegger, M.; Pfister, S.; Koehler, A.; de Giovanetti, L.; Arena, A.; Zah, R. (2011) Taking into account water use impacts in the LCA of biofuels: an Argentinean case study. The International Journal of Life Cycle Assessment, 16(9), 869-877

Fekete, B.M., Vorosmarty, C.J., Grabs W., 2002. High-resolution fields of global runoff combining observed river discharge and simulated water balances, Global Biogeochem. Cycles, 16 (3): 15-1 to 15-10.

Feng, K; Chapagain, A; Suh, S; Pfister, S; Hubacek, K; (2011) Comparison of bottom-up and top-down approaches to calculating the water footprints of nations. Economic Systems Research, 23(4), DOI 10.1080/09535314.2011.638276

Fischer, G., van Velthuizen, H., Nachtergaele, F., Medow, S., 2000. Global agro-ecological zones. FAO and IIASA, http://www.iiasa.ac.at/Research/LUC/GAEZ/index.htm, Accessed August 3, 2009.

Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E. and Richter, B.D. (2012) Global monthly water scarcity: Blue water footprints versus blue water availability, PLoS ONE 7(2): e32688.

Kounina A, Margni M, Bayart JB, Boulay AM, Berger M, Bulle C, Frischknecht R, Koehler A, Mila-i-Canals L, Motoshita M, Núñez M, Peters G, Pfister S, Ridoutt BG, van Zelm R, Verones F, Humbert S (2012) Review of methods addressing freshwater availability in life cycle inventory and impact assessment. Int J Life Cycle Ass (in review)

Nunez, M; Pfister, S; Anton, A; Munoz, P; Hellweg, S; Koehler, A; Rieradevall, J. (2012) Assessing the environmental impacts of water consumption by energy crops grown in Spain. J Ind Ecol (available online)

Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol., 43(11), 4098–4104

Pfister S and Hellweg S (2009) The water "shoesize" vs. footprint of bioenergy. PNAS 106:E93-E94

Pfister S, S Hellweg. 2011. Surface water use – human health impacts. Report of the LC-IMPACT project (EC: FP7). Available at. http://www.ifu.ethz.ch/ESD/downloads/Uncertainty_water_LCIA.pdf

Pfister, S.; Curran, M.; Koehler, A.; Hellweg, S. (2010) Trade-offs between land and water use: regionalized impacts of energy crops. Published in Proceedings of the 7th International Conference on LCA in the Agri-Food Sector. 2010, Università degli studi di Bari, Aldo Moro, Itlay. ISBN 978-88-88793-29-0

Pfister, S., Bayer, P., Koehler, A., Hellweg, S., 2011a. Environmental impacts of water use in global crop production: hotspots and trade-offs with land use. Environ. Sci. Technol., 45(13), 5761–5768

Pfister, S.; Bayer, P.; Koehler, A.; Hellweg, S. 2011b. Projected water consumption and land use in future global agriculture: scenarios and related impacts. Science of the Total Environment, 2011, doi:10.1016/¬j.scitotenv.2011.07.019

Pfister S, Saner D, Koehler A (2011c) The environmental relevance of water consumption in global power production. Int J Life Cycle Ass 16: 580-591

Ridoutt BG, Pfister S (2010a) Reducing humanity's water footprint. Environ Sci Technol 44:6019-6021

Ridoutt BG, Pfister S (2010b). A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Glob Environ Chang 20:113-120

Ridoutt BG, Pfister S (2012). A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone indicator, Int J LCA, (accepted for publication).

UNEP (2012) Measuring water use in a green economy. McGlade, J., Werner, B., Young, M., Matlock, M., Jefferies, D., Sonnemann, G., Aldaya, M., Pfister, S., Berger, M., Farell, C., Hyde, K., Wackernagel, M., Hoekstra, A., Mathews, R., Liu, J., Ercin, E., Weber, J.L., Alfieri, A., Martinez-Lagunes, R., Edens, B., Schulte, P., von Wirén-Lehr, S., Gee, D. ISBN: 978-92-807-3220-7

Verones, F.; Hanafiah, MM.; Pfister, S.; Huijbregts, MAJ; Pelletier; GJ; Koehler, A. Characterisation factors for thermal pollution in freshwater aquatic environments. Environ. Sci. Technol, 2010, 44(24), 9364-9369

Holistic sustainability assessment of winter oilseed rape production using the AgBalance™ method – an example of 'sustainable intensification'?

Markus Frank[1]*, Jan Schöneboom[2], Martijn Gipmans[3], Peter Saling[2]

[1] BASF SE,APD/S, Speyrer Strasse 2, 67117 Limburgerhof, Germany
[2] BASF SE, Sustainability Center, ZZS/SE – Z7, 67056 Ludwigshafen, Germany
[3] BASF Plant Science Company GmbH, c/o metanomics GmbH, Tegeler Weg 33, 10589, Berlin, Germany
* Corresponding author. E-mail: *markus.frank@basf.com*

ABSTRACT

AgBalance™ is a holistic method to quantify the sustainability performance of agricultural production systems. It combines LCA with environmental, economic and social impact indicators, generalized to varying spatial scales. AgBalance comprises up to 70 sustainability indicators, based on a significantly larger number of input data and parameters. The indicators are grouped into the three dimensions environmental, social and economic sustainability impacts. Both detailed in-depth results of individual impact indicators, as well as aggregated results are output of AgBalance. We present herein results of a case study on oilseed rape production , comparing the average good agricultural practices of 1998 and 2008 at the state level (State Mecklenburg-West Pomerania, North-East Germany). For the functional unit of providing one ton of rapeseed a the field border, the aggregated results indicate significant improvements in the overall sustainability single score and particularly in the environmental aggregated score, for the2008 production protocol. Using scenario analysis, the use of fertilisers including denitrification-inhibitors such as ENTEC® 26, was investigated, which is linked to potentially significant reductions in N_2O field emissions.

Keywords: ecoefficiency analysis, Social Life Cycle Assessment, scenario analysis, winter oilseed rape, denitrification inhibitors

1. Introduction and goal

Most of the existing methods to assess agricultural sustainability for farm or crop management (e.g. Bockstaller et al., 2008, 2009; Field to Market 2009; Grenz et al., 2009) are not based upon a life cycle approach or do not address all dimensions of sustainability in a holistic manner. In building on the environmental impacts and economic costs assessed in BASF's Eco-efficiency Analysis (Saling et al., 2002) and the social impact indicators in SEEBALANCE® (Schmidt et al., 2004; Kölsch et al., 2008), BASF has designed AgBalance™ as a new method to assess the sustainability performance of agricultural production systems. AgBalance in addition contains a range of new agriculture-specific indicators, which were identified and developed in a dialogue with various stakeholders, specifically addressing biodiversity and soil quality. This new method is created to assess and manage continuous improvement in sustainable agriculture at several levels: (1) for the farmer, by assessing current practices and developing scenarios for improved processes, (2) for the agri-food value chain, by assessing agriculture's contribution over the complete product life cycle and developing options for improvement, and (3) for policy makers, by assessing the impact of regulations on products and farming practices.

Here, we present results of an AgBalance™ case study on winter oilseed rape production at the geographical level of Northern German state Mecklenburg-West Pomerania. Recent reports indicate that oilseed rape production has seen a substantial increase in productivity and profitability for the farmers over the last decade (AgMRC 2012). Whether or not this intensification has come at the expense of increased impacts on environment and society, however, remained unclear. The goal of the study therefore was to investigate the sustainability performance of state-of-the art winter oilseed-rape production in the study region, and compare it to the period ten years before, by assessing holistically environmental, social and economic sustainability impacts with a focus on the on-farm activities.

2. Methods and scope of the study

The AgBalance methodology (Schoeneboom 2012a, Schoeneboom 2012b) is based on BASF's Eco-Efficiency Analysis and SEEBALANCE® methods for quantitative sustainability impact assessment (Saling et al., 2002, Schmidt et al., 2005, Kölsch 2008). It includes an environmental LCA for several impact categories, based on mandatory and optional parts of the ISO 14040 and 14044 standards for life cycle assessment. Furthermore, recommendations of the UNEP/SETAC working group for social LCA as well as the SA8000 and ISO26000SR standards were followed in the development of the methodology. The results from the individual impact categories were weighted and aggregated as outlined in Saling et al., (2002) and Kicherer et al., (2007). The method received independent assurance of functionality and coherence from DNV Business Assurance, TÜV Süd and the National Sanitation Foundation (BASF 2011).

For the present case study on winter oilseed rape production in the Northern German state Mecklenburg-West Pomerania, the main data sources were data sets from state office for agricultural research, particularly the “reference farm monitoring network”, and the state’s annual Agricultural Report (LFA Mecklenburg-Vorpommern 2009); moreover interviews conducted on five farms, each with 100 – 500 ha of agricultural area planted with winter oilseed rape. The study’s scope was the sustainability performance of winter oil-seed-rape production in the study region in comparison to production ten years before. We have collected data sets for the time period 2008 (five-year averages) and 1998; the two alternative product systems compared are thus the ‘good agricultural practice 2008’ and ‘good agricultural practice 1998’. All assumptions were verified with the state office for agricultural research (Gülzow) and selected farmers. The functional unit was defined as the ‘production of 1 ton of oilseed rape, cradle to field border, in Mecklenburg-Vorpommern, Germany, product water content below 9%, data from 2008 and 1998’. As no by products (straw) are usually harvested all impacts were allocated to the rape seed. The system boundaries of both the alternative production scenarios (2008, 1998) are shown in Fig. 1.

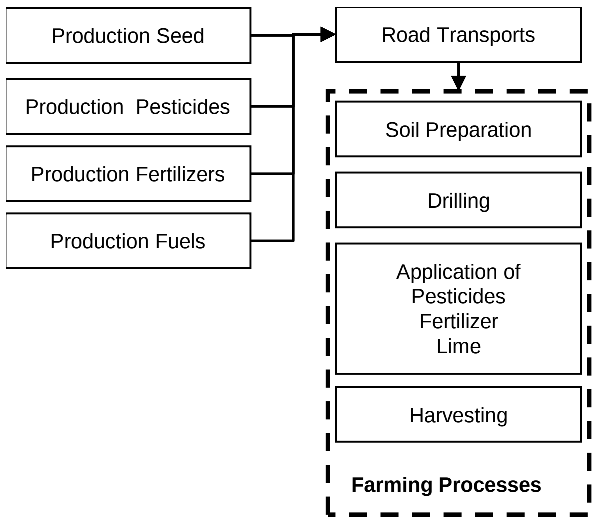


Figure 1. System boundaries of the AgBalance case study

3. Results and discussion

We focus here on the discussion of environmental impacts, as these showed the biggest difference between the two studied systems. The assessment of environmental impacts comprises the global warming potential, acidification -, ozone depletion -, and photochemical ozone creation potential, as well as emissions to water. Resource efficiency indicators include pimary energy demand, abiotic resource depletion, (Saling 2002) land use, (Koellner and Scholz 2007), and water consumption (Pfister 2009). Moreover, we assessed the aquatic ecotoxicity potential resulting from emissions of pesticides and fertilisers according to Saling et al., (2005); Summarizing these results, the impacts were significantly lower in the 2008 compared to the 1998 production system. This partly reflects the fact that rapeseed production on the reference monitoring farms has seen substantial increase in yields from 1998 to 2008, on average from 2.7 ton/ha to 4.1 ton/ha. As the functional unit is defined as one ton of rapeseed, the emission flows and exchanges in relation to the reference flow were lower in the 2008 production system.

AgBalance™ however additionally requires the evaluation of a set of agri-environmental indicators pertaining to specific biodiversity impacts in agricultural areas. These indicators are quantified per hectare and year, and have no numerical correlation with the reference flow, in order to show in a transparent way the management of the agricultural soil.

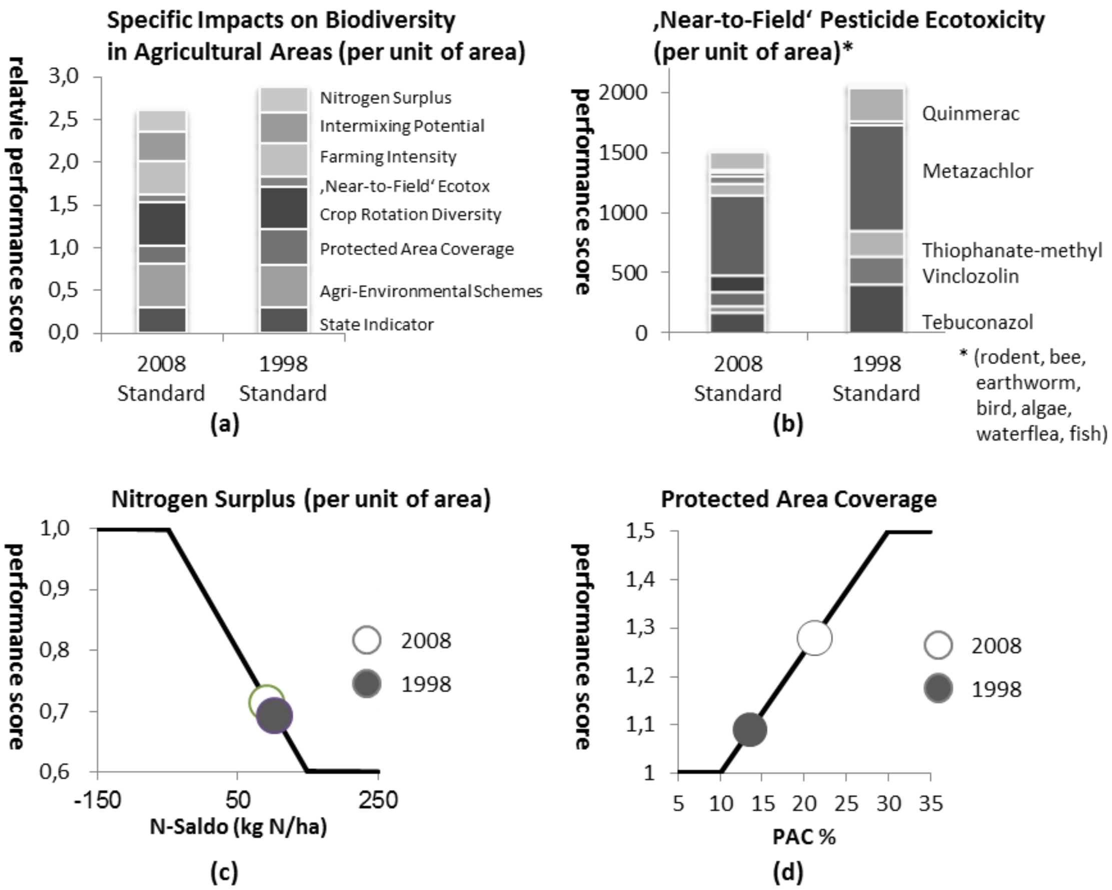


Figure 2. Assessment of specific impacts on biodiversity in agricultural areas: (a) aggregated result, (b) evaluation of combined pesticide ecotoxicity profile for endpoints rodent, bee, earthworm, bird, algae, waterflea, fish, (c) evaluation of nitrogen surplus, (d) evaluation of protected area coverage.

The results of this analysis are summarized in Figure 2 (a). The group of sub-indicators comprises: a state indicator, describing the vulnerability of the ecosystems in the study region, adoption of agri-environmental schemes, protected area coverage, the pesticide ecotoxicity profile relating to several endpoints, the crop rotation diversity, the intermixing potential of crops, farming intensity, and nitrogen surplus. The main contribution to differences between the 1998 and 2008 production systems in the overall biodiversity indicator are, in the present case, the protected area coverage, the ecotoxicity profile, and the nitrogen surplus. The protected area coverage evaluates the coverage of Natura2000 and FFH areas in the study region (in this case, the state) against a given target value, as well as the fraction of area enrolled in agri-environmental schemes. Improvements have been mainly attributed to the first category. The ecotoxicity profile is shown in Fig. 2 (b). The relative improvement is mainly due to a more efficient use of state-of-the-art pesticides with a lower ecotoxicity profile. Finally, nitrogen surplus has decreased in the 2008 case, due to improvements in the nitrogen saldo; we note, however, that both systems still exhibit large and positive nitrogen balances.
Relative results of all indicators are shown on a normalised scale in the environmental fingerprint diagram, in Fig. 3. Relative improvement in each impact is represented by smaller values on the respective axes; hence the smaller the fingerprint, the better the relative performance of the corresponding alternative. As discussed above, the assessment of specific impacts on biodiversity in agricultural areas show an improvement in the biodiversity potential, caused by both an increased assignment of protected areas in the region as well as a better ecotoxicity profile of state-of-the-art agrochemical inputs (i.e. a shift from organophosphates and carbendazim to modern insecticides and fungicides). Moreover, improvements for most of the other environmental impact categories were revealed for “2008 Standard”, such as “energy consumption”,”water use” or “resource depletion”, to a large extent due to a more efficient land use (Fig. 3a). Although individual social sustainability indicators exhibited changes between 1998 and 2008, the aggregated social sustainability score remained nearly unchanged, due to compensation of positive and negative impacts. In particular, rationalisation efforts resulted in fewer working hours in agriculture (Fig. 3b).

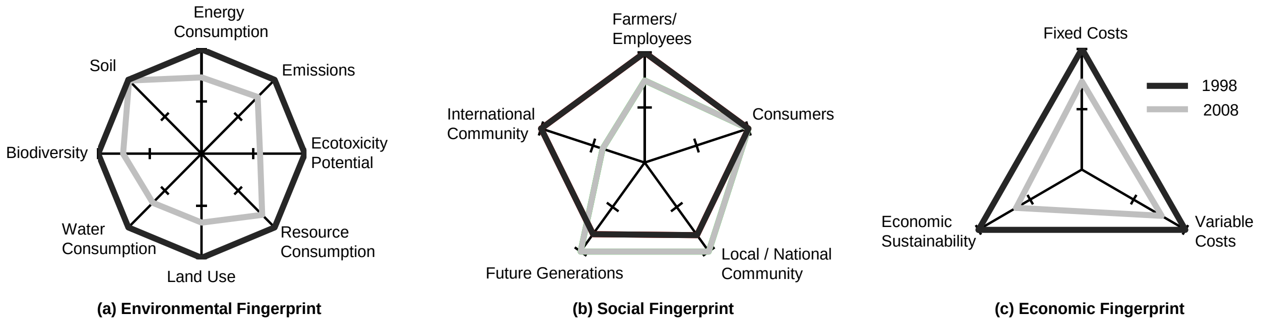


Figure 3. Representation of relative impact category results for alternatives in environmental (a), social (b) and economic (c) fingerprint diagrams. Relative improvement in each impact category is represented by smaller values on the respective axes; hence the smaller the fingerprint, the better the relative performance of the corresponding alternative.

On the other hand, rationalisation drove to a large extent the improved performance on the part of "fixed costs" (Fig. 3c), which also resulted in a reduced number of working hours within the agriculture industry in the state of Mecklenburg-West Pomerania. This reduction in hours worked mainly contributes to the lower performance in the impact category "local/national community" (Fig 3b). On the part of "Economy", both higher commodity prices and higher yields contributed to an improved profitability (and overall economic sustainability; Fig. 3c). The reason for the higher yields is the market introduction and adoption of open-pollinating varieties as well as of hybrid varieties resulted in a more intensive use of fertilisers and plant protection compounds. Moreover, in 2008 a larger portion of farmers adopted conservation tillage practices than in 1998. All these changes caused lower variable and fixed costs per functional unit (Fig 3c).
Scenario analysis was carried out to investigate potential improvements in emission of greenhouse-gases by changing to a different nitrogen fertiliser, including a de-nitrification inhibitor (i.e. ENTEC® 26).

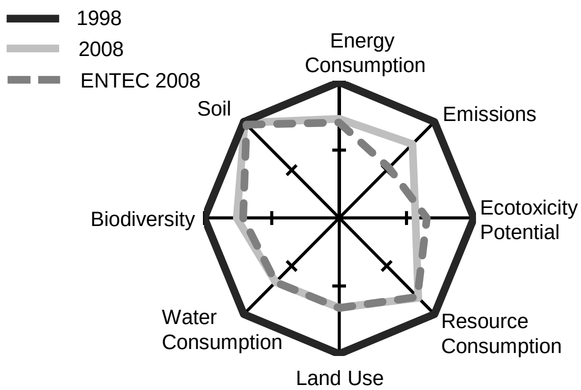


Figure. 4. Environmental fingerprint with scenario 'ENTEC', assuming that N-fertilisation is exclusively done with ENTEC26, containing a denitrification inhibitor

The results of the scenario indicate a significant improvement of the 2008 sustainability performance (Fig. 4). This increase was mainly conferred by improvements in the indicators "greenhouse gas emissions", and "water emissions" as well as in "nutrient balance" of the soil. The dimensions "Economy" and "Society" stayed largely unchanged. This finding indicates that a switch to denitrification inhibitor-based fertilisers would be a useful measure to improve the sustainability of oilseed rape production, especially against the background of a high nitrogen demand of oilseed rape in comparison to other crops (Malagoli et al., 2005).

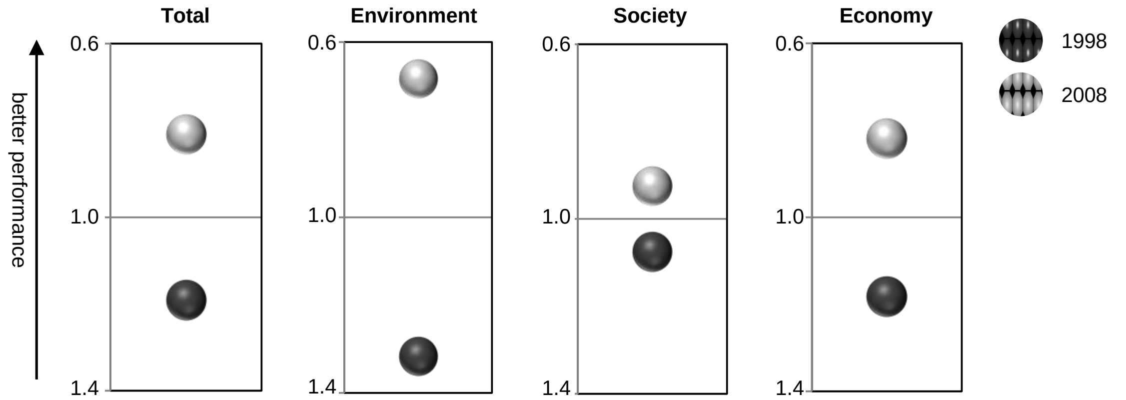


Figure 5. Representation of single score results. The alternative with the relative better performance is at the top, the one with the worst performance at the bottom of the graph.

4. Conclusions

As Fig. 5 summarises, this AgBalance case study on winter oilseed rape production in Northern Germany provides an example for a sustainable intensification in modern agriculture. The state-of-the-art agricultural practice from 2008 outperformed the 1998 practice in all three dimensions of sustainability. The amount of detailed information on various aspects of the sustainability performance of the production system together with scenario analysis make AgBalance a powerful tool to derive recommendations for optimized crop production protocols. It needs to be stated, however, that these recommendations require further verification and substantiation through field testing.

5. References

AgMRC, 2012. Agricultural Marketing Center – Rapeseed. Available online at: http://www.agmrc.org/commodities__products/grains__oilseeds/rapeseed/. Accessed: 10 May 2012.

BASF (2011) AgBalance http://www.agro.basf.com/agr/AP-Internet/en/content/news_room/index . Accessed: 10 May 2012.

Bockstaller C., Guichard L., Makowski D., Aveline A., Girardin P., Plantureux S., 2008. Agri-environmental indicators to assess cropping and farming systems. A review. Agron. Sustain. Dev. 28:139-149.

Bockstaller C., Guichard L., Keichinger O., Girardin P., Galan M.B., Gaillard G., 2009. Comparison of methods to assess the sustainability of agricultural systems. A review. Agron. Sustain. Dev. 29:223-235.

AF Bouwman, LJM Boumans, NH Batjes, Global Biochemical Cycles, Vol. 16, No. 4, p. 1080, Table 3.

Döhler et al., 2002: 'BMVEL/UBA Ammoniak-Emissionsinventar der deutschen Landwirtschaft und Minderungsszenarien bis 2010'

Field to Market, 2009. Environmental Resource Indicators for Measuring Outcomes of On-Farm. Agricultural Production in the United States, First Report, The Keystone Alliance for Sustainable Agriculture, Available online at http://keystone.org/spp/env-sustain_ag.html. Accessed 10 January 2012.

Grenz J., Thalmann C., Stämpfli A., Studer C., Häni F., 2009. RISE – a method for assessing the sustainability of agricultural production at farm level. Rural Dev. News 1: 5-9.

IPCC 2006: 2006 IPCC Guidelines for National Greenhouse Inventories, Vol. 4, Agriculture, Forestry, and Other Land Use, S. Egglestone, L. Buendia, K. Miwa, T. Ngara, K. Tanabe (eds.), IGES, Japan.

Kicherer A., Schaltegger S., Tschochohei H., Ferreira Pozo B., 2007. Combining Life Cycle Assessment and Life Cycle Costs via Normalization. Int. J. Life Cycle Assess. 12:537-534.

Köllner T, Scholz R (2007) Assessment of Land Use Impacts on the Natural Environment, Part 1: An Analytical Framework for Pure Land Occupation and Land Use Change. Int. J. Life Cycle Assess. 12:16-23; Köllner T, Scholz R (2008) Assessment of Land Use Impacts on the Natural Environment, Part 2: Generic Characterization Factors for Local Species Diversity in Central Europe. Int. J. Life Cycle Assess. 13:32-48.

Kölsch D., Saling P., Kicherer A., Grosse-Sommer A., Schmidt I., 2008. How to measure social impacts? A socioeco-efficiency analysis by the SEEBALANCE® method. Int. J. Sustain. Dev. 11:1- 23.

LFA Mecklenburg-Vorpommern, 2009. Available online at: http://www.landwirtschaft-mv.de/cms2/LFA_prod/LFA/content/de/Fachinformationen/Betriebswirtschaft/Allgemein/index.jsp?&artikel=1531. Accessed: 10 May 2012

Malagoli P., Laine P., Rossato L., Ourry A., 2005. Dynamics of Nitrogen Uptake and Mobilization in Field-grown Winter Oilseed Rape (Brassica napus) From Stem Extension to Harvest. II. An ^{15}N-labelling-based Simulation Model of N Partitioning Between Vegetative and Reproductive Tissues. Ann Bot., 95 (7), 1187-1198.

Pfister, S, Koehler A, Hellweg S (2009) Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43: 4098–4104.

Saling P., Kicherer A., Dittrich-Kraemer B., Wittlinger R., Zombik W., Schmidt I., Schrott W., Schmidt S., 2002. Eco-efficiency Analysis by BASF: The Method. Int. J. Life Cyle Assess. 7:203-218.

Saling P., Maisch R, Silvani M, König N (2005) Assessing the Environmental-Hazard Potential for Life Cycle Assessment, Eco-Efficiency and SEEbalance® Int. J. Life Cycle Assess., 10:364-371.

Schoeneboom 2012a: Schoeneboom, J.; Saling, P.; Gipmans, M. 2012. AgBalanceTM – Technical Background Paper. Available at http://www.agro.basf.com/agr/AP-Internet/en/function/conversions:/publish/upload/sustainability/AgBalance/307736_BASF_Tech-E_Paper-AgBalance.pdf

Schoeneboom 2012b: AgBalance™ - Holistic Assessment of Sustainability in Agriculture Part 1: Method Framework. Schoeneboom, J.; Saling P.; van Gelder, R. 2012. Submitted to Int. J. Life Cycle Assess.

Schmidt I., Meurer M., Saling P., Reuter W., Kicherer A., Gensch C.O., 2004. SEEBALANCE®: Managing sustainability of products and processes with the socio-eco-efficiency analysis by BASF, Greener Management International, 45:79 - 94.

Assessment of existing and potential cereal food and non food uses by combining E-LCA and S-LCA

Florence Van Stappen[1 *], Alice Delcour[2,4], Stéphanie Gheysens[3], Virginie Decruyenaere[3], Didier Stilmant[4], Philippe Burny[2], Fabienne Rabier[5], Jean-Pierre Goffart[2]

[1] Biomass, Bioproducts and Energy Unit, Walloon Agricultural Research Centre (CRA-W), 146 chaussée de Namur, 5030 Gembloux, Belgium
[2] Crops Production Systems Unit, CRA-W, 4 rue du Bordia, 5030 Gembloux, Belgium
[3] Animal Breeding, Quality Production and Welfare Unit, CRA-W, 8 rue de Liroux, 5030 Gembloux, Belgium
[4] Farming Systems, Territories and Information Technology Unit, CRA-W, 100 rue du Serpont, 6800 Libramont, Belgium
[5] Agricultural Machines and Facilities Unit, CRA-W, 146 chaussée de Namur, 5030 Gembloux, Belgium
[*] Corresponding author. E-mail: *vanstappen@cra.wallonie.be*

ABSTRACT

In Wallonia (Belgium), 60% of the arable cropped area is dedicated to cereals. More than a half of the cereal chains are currently turned towards animal feed. Direct human food uses barely reach 10% of the grain production. Non food uses are growing, with 16% of the Walloon cereal production converted into bioethanol or biogas. Based on a comprehensive description of the Walloon cereal sector, the project ALT-4-CER considers opportunities for food, feed, fuel and fibre uses of Walloon cereal resources (“4F” scenarios) through an exhaustive comparison of existing and potential conversion chains. Scenarios are evaluated regarding environmental and socio-economic aspects through Life Cycle Analyses fed by region-specific data. Environmental LCA aims at identifying territorial differences regarding the cultivation step in comparison with data from commonly used databases. Beside E-LCA, a Social LCA methodology is being elaborated. Environmental and socio-economic assessment results will then be integrated thanks to multi-criteria analysis.

Keywords: cereals, life cycle assessment (LCA), environmental impact, socio-economic impact, decision making

1. Introduction

In the present sustainable development framework, agriculture raises major concerns in terms of productions and farmers’ income diversification, competition for arable land between food and non food uses, employment preservation or creation in rural areas, climate change mitigation and natural resources protection.

Current environmental policies and regulations aim at moving towards more sustainable production and consumption pathways. This transition requires the improvement of the energy, environmental and social balances of considered processes and reference systems. This shift also involves the search for new and alternative uses of agricultural resources, products and co-products.

In Wallonia (Belgium), more than 60% of the arable cropped area is dedicated to cereals (DGSIE 2010). In the current context of non renewable energy sources depletion and growing world population, competition for cereal resources requires the identification of the most sustainable scenarios for agricultural products and co-products use.

Nowadays most Walloon cereal chains are classically turned towards human food and animal feed. Key opportunities for non food uses are considered in a sustainable development perspective. But this can only be achieved through an exhaustive comparison of environmental and socio-economic impacts of existing and potential cereal chains.

2. Objectives

Comparing current and potential cereal uses in Wallonia, the overall goal of the 3-year project “Alternatives for Cereals – ALT-4-CER”, started in March 2011, is to define and to evaluate alternative scenarios for food and non food uses of cereal resources in Wallonia with the support of involved stakeholders.

In order to depict comprehensively the Walloon cereal landscape the project firstly aimed at elaborating scenarios for food, feed, fuel and fibre uses of cereal resources in Wallonia (so-called “4F” scenarios). Scenarios definition was supported by the consultation of all involved actors (producers, wholesalers, processors, consumers, and decision-makers) in order to ensure a further scientific approach based on realistic existing and potential cases. Scenarios take into account interactions between chains and their co-products, as well as potential impacts on the agricultural landscape and land occupation in Wallonia.

On the basis of these scenarios the project is evaluating their environmental impacts through the development of Environmental Life Cycle Assessments (E-LCA) adapted to the local context, fed with specific data collected for Wallonia. Furthermore methodological choices regarding system boundaries, technological options, data quality, co-product treatment, or result interpretation must be justified and their sensitivity tested.

Simultaneously, a Social Life Cycle Analysis methodology (S-LCA) is being developed in order to grasp socio-economic impacts of defined scenarios. This method, complementary to E-LCA but not as well developed yet, requires more methodological adjustments.

Finally the relevance of the selected scenarios will be evaluated through multi-criteria analysis supported by the sector. Stakeholders will again be convened in order to help weighting sustainable indicators identified in the E-LCA and S-LCA processes.

3. Walloon cereal current and potential uses

3.1. The Walloon agricultural context

Wallonia is divided into 10 agricultural regions, according to soil texture and landscape of the country (Sneppe 2002). The agricultural productions of Wallonia are closely related to the opportunities offered by the soil properties or the landscape. The Northern part of Wallonia (loam area, sandy-loam area, and clayed-loam area), with its particularly fertile and deep soils, is dedicated to large-scale crops, such as cereals, sugar beet and potato. The central part of Wallonia has poorer and less deep soils and is therefore dominated by forage crops (mainly forage maize). The Southern part, with a lower population density, is mostly covered by woods and forests, where agricultural lands are devoted to meadows and pastures. The Eastern Belgium area, with the highest a.s.l. elevations in Belgium, is dedicated to pastures and dairy productions; orchards and bocage are also found in this area.

Agricultural statistics show that more than 60% of the Agricultural Area Utilised for Farming (AAUF) is dedicated to cereals in Wallonia (DGSIE 2010). Main cereal crops are winter wheat (36%), forage maize (16%) and winter barley (10%). Thanks to its hardiness, spelt is also common in the Southern area with its less fertile soils, more cold and wet climate and steeper landscape.

Trends regarding main crops in Wallonia indicate that areas dedicated to wheat have been stable for the past 15 years, while barley and spelt areas are slowly decreasing. This can be related to climate becoming milder, allowing more productive wheat to overtake other cereals. Within the same period, grain maize areas have increased due to climate evolution and genetic selection. Forage and grain maize progressions seem to be made at the expense of sugar beet and temporary meadows.

The ALT-4-CER project team has therefore decided to focus on the following prevailing cereal crops in Wallonia: wheat (*Triticum aestivum* L.), barley (*Hordeum vulgare* L.), forage and grain maize (*Zea mays* L.), and spelt (*Triticum aestivum* subsp. *spelta* L.).

3.2. Current cereal uses in Wallonia

More than a half of the cereal chains are currently turned towards animal feed. Direct human food uses, i.e. milling and brewing, barely reach 10% of the grain production, mainly because of low prices paid for food varieties, less favourable climate conditions and scattered plots of land. Non food uses are growing too, with 16% of the Walloon cereal production converted into bioethanol or biogas (Table 1, Figure 1).

Table 1. Production and uses of Walloon cereal resources in 2010

Use	Grain		Forage maize		Straw		TOTAL	
	1000 T FM[a]	%	1000 T FM[a]	%	1000 T FM[a]	%	1000 T DM[b]	%
Food	125	8	0	0	0	0	108	4
Feed	724	46	2601	98	68	11	1544	57
Fuel (ethanol and biogas)	501	32	53	2	0	0	449	16
Fibre (animal litter)	0	0	0	0	547	89	465	17
Export	227	14	0	0	0	0	160	6
TOTAL	**1577**		**2654**		**616**		**2725**	

[a]FM = fresh matter
[b]DM = dry matter

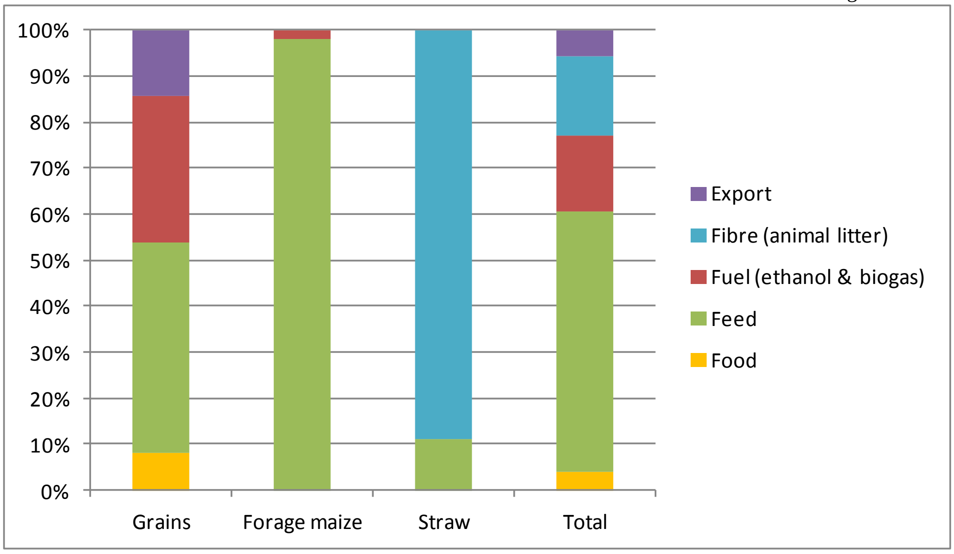


Figure 1. Uses of Walloon cereal resources in 2010.

3.3. Scenario definition

The definition of "4F" scenarios is the basis for the subsequent steps of the project and was thus effectively supported by involved stakeholders. Cereal flows (production, export, import, stock) have been evaluated thanks to expert consultation and available relevant data and literature. These flows, integrating historic trends and projections by 2030, helped defining the baseline scenario (2010) as well as future potential scenarios for cereal uses in Wallonia (2030).

Scenarios consider interactions between chains and their co-products, as well as potential impacts on the agricultural landscape and land occupation in Wallonia. They have been established with contrasting hypotheses: (1) "Business-as-usual" scenario: current trends are extrapolated from the past 15 years; (2) "Strategic" scenario: environmental, economic and social optimisation of current system; (3) "Localisation" scenario: development of new cereal conversion units in Wallonia (added value relocated within the region) and increased autonomy; and (4) "Globalisation" scenario: world demand drives cereal resources outside Wallonia (massive export), and Wallonia focuses on high added-value products (biorefinery, bio-based chemistry) (Figure 2).

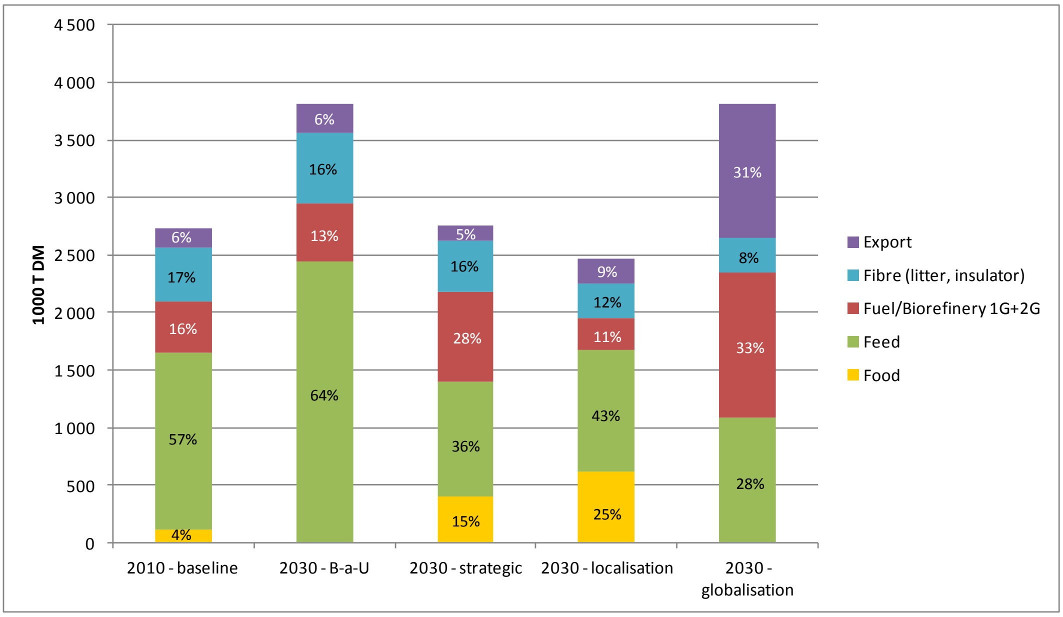


Figure 2. Production and uses of main Walloon cereal resources (wheat, barley, spelt, grain maize and forage maize) in the 4 scenarios (DM = dry matter).

Details on Walloon cereals flows and scenario definition will be available in an article to be submitted to *Biotechnology, Agronomy, Society and Environment* (Eds. Presses agronomiques de Gembloux).

4. Scenarios evaluation with LCA

Scenarios are now being evaluated regarding environmental and socio-economic aspects through LCA fed by region-specific data adapted to the local context.

The bottleneck to estimate production chain sustainability is the lack of knowledge regarding material and energy flows, and environmental and socio-economic impacts, from raw material extraction to waste disposal. Among available evaluation tools, LCA is nowadays the most commonly used method (Moras 2007). Using LCA can have two complementary goals: either to compare products, processes or services according to their impacts, or to identify, within a production and use chain, key points to be improved in order to minimize any given impact. The ISO 14044:2006 standard (ISO 2006) defines three impact category groups called “Areas of Protection”: (i) natural and abiotic resources use, (ii) human health consequences and (iii) ecological consequences. Beside E-LCA, recent developments in LCA tend to integrate socio-economic aspects in S-LCA, in order to grasp all three pillars of sustainable development. With respect to S-LCA, a fourth Area of Protection is usually added: human dignity and well-being (Jørgensen, Bocq et al., 2008).

The use of local and specific data is crucial when conducting LCA. Data uncertainty is closely related to data reliability and completeness, but also to geographical, temporal and technological correlation (Frischknecht, Jungbluth et al., 2007). Supported by long-term expertise and wide-spread contact network of the research team, the ALT-4-CER project has committed itself to collect and use specific data, adapted to the Walloon context.

5. Scope and goal definition

5.1. Consequential *versus* attributional LCA

Two types of LCA methodologies are used according to the objective(s) of the study: attributional LCA (A-LCA) and consequential LCA (C-LCA). A-LCA describes the relevant physical flows entering and exiting a product system. C-LCA expresses how these flows will evolve in response to decisions or changes (Finnveden, Hauschild et al., 2009). When C-LCA is useful to assess the consequences of individual decisions, A-LCA enables distinguishing between systems having important impacts. These complementary objectives allow both A-LCA and C-LCA methodologies to be valid for decision-making.

Regarding these definitions, the goal of the ALT-4-CER project can be formulated as follows: to assess the consequences, in terms of environmental and socio-economic impacts, of potential changes in Walloon cereal uses by 2030, in comparison with the current situation (2010). Consequential LCA is therefore the appropriate choice with respect to this objective.

5.2. System boundaries and functional unit definition

The first step of any LCA is to define the goal and scope of the study (ISO 2006). This essential stage lays the basis of the study by answering key questions such as “What do we study? In what purpose? Who is the targeted audience?”. System boundaries need to be cut between impacts considered as relevant and others. Besides, a functional unit (FU) needs to be chosen. The FU aims at providing a reference to which all input and output flows in the assessment are normalized (Weidema et al., 2004). In a comparative study, the FU shall be the same for all compared product systems. This is a prerequisite for ensuring equivalence among the product systems (ISO, 2006).

Food and non food uses considered in ALT-4-CER are classified in “4F” categories:

1. Human (Food) uses (i.e. flour mills, beer products, starch products used in agro-food industries, etc.);
2. Animal (Feed) uses (i.e. feed ingredients for animal rations, agro-food and biofuel industry co-products, grain and/or straw self-consumption on farm, etc.);
3. Energy (Fuel) uses (i.e. ethanol production from starch, second generation ethanol production from straw, biogas production from maize, straw direct combustion, etc.);
4. Material (Fibre) uses (i.e. straw for animal litter, non food uses of starch, straw use as isolation material, biorefineries, etc.).

Those four categories imply various end-uses and very different functions of the initial cereal resources.

(De Boer 2003) considers that the kilogram is a convenient FU in order to take into account both production efficiency and impacts. Normalizing impacts according to land occupation on the contrary does not account for production efficiency. On the other hand, both the mass and surface can be used as FU for assessing global impacts, whether the hectare is preferred for regional impact assessment (Basset-Mens 2005).

Considering the multiple functions of the various systems studied here the proposed FU is "any useful output per hectare in an average year" (Rettenmaier, Köppen et al., 2010). This option uses a FU which is the same for all 4F scenarios and illustrates the competition for land between food and non food. On the other hand, the function of agriculture is not to occupy land. Nor is this solution convenient to compare different production systems, such as organic farming *versus* conventional farming, or different land use intensities, because it does not account for the lesser productivity of organic farming or less intensive production systems. A sensitivity analysis will thus test conclusions strength according to another FU defined as "any useful output per kilogram of a given cereal in an average year". Besides, regarding S-LCA, a FU considering working hours can be a useful unit for specific socio-economic concerns.

6. Environmental life cycle assessment

Environmental Life Cycle Analysis (E-LCA) aims at identifying territorial differences regarding the cultivation step in comparison with generic data found in commonly used databases.

Cultivation pathways and agricultural work processes are modelled on the basis of current research in the Region. New cropping practices such as no tillage or direct sowing are being explored. Machines and tractor consumptions are adapted according to common practices in Wallonia. Methods assessing direct field emissions relating to agricultural inputs application during and after plant production are also closely surveyed, especially since local climate, practices and soil characteristics can have a huge impact on environmental performances. Other aspects such as inputs manufacture and management, and animal feeding and husbandry are scrutinized too. Downstream of the cultivation step, different conversion processes will be studied, with a focus on existing facilities.

7. Socio-economic life cycle assessment

Besides E-LCA, a Social Life Cycle Analysis (S-LCA) methodology is being elaborated in order to evaluate socio-economic impacts of scenarios. S-LCA studies currently assess social performances and generally not proper social impacts (Macombe, Feschet et al., 2011). Similarly to E-LCA, consequential LCA will be used in order to evaluate specific social impacts due to decision alternatives. In that case S-LCA assesses social impacts caused by choosing decision alternatives (Jorgensen et al., 2011).

The ultimate goal of this study is to contribute to the drawing up of a S-LCA methodology for the Walloon cereal value chains. This S-LCA aims at answering the question: "What are the similarities, differences and future trends in terms of added value and working environment for cereal Walloon chains?" The scope includes the agricultural step down to the first conversion in Food, Feed, Fuel, Fibre sectors using Walloon cereals. At the cultivation level geographical differences will consider crop choice linked to territorial specificities.

S-LCA implies the definition of a range of particular stakeholders (farmers, firms, workers and local communities) and workable indicators (number of work-related accidents, employment, training, qualification, etc.) specific to cereal production. Farmers constitute a specific category because they have a particular status: they are at the same time consumers, sellers, workers and managers. Based on political priorities, a methodology evaluating wellness from both the economic and social dimensions is being developed, based respectively on added value and work time distribution.

Two impact categories are developed for this study: (i) distribution of added value along the agricultural chain and (ii) working environment. These impact categories aim at being as quantifiable as possible because results are focused on stakeholders of Walloon cereal chains directly concerned by cereal resources. Added value is evaluated thanks (i) to regional database of agricultural accounts and (ii) balance sheets freely available for the rest of the actors. Working environment will be evaluated through the "Bilan Travail" methodology (INRA, l'Elevage et al., 2008) and thanks to several indicators and interviews. "Bilan Travail" is dedicated to animal breeding and must therefore be adapted for cereal sectors. "Bilan Travail" estimates work hours through interviews with farmers and other farm workers, dedicated to (i) repetitive and postponed work, (ii) seasonal work and (iii) mutual aid work. Available time for administrative tasks, leisure time and others are calculated by the difference between legal working days and days occupied by works listed above.

Others indicators such as numbers of work accidents, employment by value chain or distribution of work contracts are used.

8. Expected results

Supported by key methodological tools such as Life Cycle Analysis and Multi-criteria Analysis applied to alternative scenarios for cereal resources uses fed with local data, the ALT-4-CER project will provide clues to answer key questions raised today in human Societies, such as “What type of agriculture do we want for tomorrow? Is it ethically, environmentally and economically sustainable to dedicate cereals resources to other uses than human food?”

In particular, the project will (i) support Walloon cereal chains by optimizing their production and use choices, (ii) help decision-makers by providing them with scientific support in the drawing of new environmental, agricultural and energy policies, and (iii) assist the consumer in his choices to contribute to sustainable development.

9. References

Basset-Mens, C., 2005. Propositions pour une adaptation de l'Analyse de Cycle de Vie aux systèmes de production agricole - Mise en oeuvre pour l'évaluation environnementale de la production porcine. Thèse de doctorat INRA. Rennes, Agrocampus Rennes: 244p.

De Boer, I. J. M., 2003. Environmental impact assessment of conventional and organic milk production. Livestock Production Science 80, 69-77.

DGSIE, 2010. SPF Economie - Direction générale statistique et information économique - Recensements agricoles de 1995, 2000, 2005, 2007, 2008, 2009, 2010.

Finnveden, G., Hauschild, M. Z., et al., 2009. Recent developments in Life Cycle Assessment. Journal of Environmental Management 91, 1-21.

Frischknecht, R., Jungbluth, N., et al., 2007. Overview and Methodology. ecoinvent report No. 1. Dübendorf, Swiss Centre for Life Cycle Inventories.

INRA, Institut de l'Elevage, Florac, SupAgro, 2008. L'organisation du travail en élevage - Enseigner la méthode Bilan Travail - Guide pédagogique. Dijon.

ISO, 2006. NBN EN ISO 14040:2006 – Management environmental - Analyses de Cycle de Vie - Principes et cadre. Geneva, Switzerland, International Organization of Standardization.

ISO, 2006. NBN EN ISO 14044:2006 - Management environnemental - Analyses du cycle de vie - Exigences et lignes directrices, IBN.

Jørgensen, A., Bocq, A., Nazarkina, L., Hauschild, M., 2008. Methodologies for social life cycle assessment. The International Journal of Life Cycle Assessment 13, 96-103.

Jørgensen, A., et al., 2011. The effect of three different SLCA methods. International Seminar on social LCA. Montpellier, Section for Quantitative Sustainbility Assessment. Technical University of Denmark.

Macombe, C., Feschet, P., Garrabé, M., Loeillet, D., 2011. 2nd International Seminar in Social Life Cycle Assessment - recent devlopements in assessing the social impacts of product life cycles. The International Journal of Life Cycle Assessment 16, 940-943.

Moras, S., 2007. Analyse comparée du cycle de vie des filières de production d'énergie renouvelable issue de la biomasse. Thèse de doctorat. Gembloux, Faculté universitaire des Sciences agronomiques: 343p.

Rettenmaier, N., Köppen, S., Gärtner, S.O., Reinhardt, G.A., 2010. Life cycle assessment of selected future energy crops for Europe. Biofuels, Bioproducts and Biorefining 4, 620-636.

Sneppe, C., 2002. La structure de l’industrie sucrière. Retrieved in September 2007, from www.subel.be/emc.asp?pageId=17.

Weidema, B. P., Wenzel H., Petersen, C., Hansen, K., 2004. The Product, Functional Unit and Reference Flows in LCA. Environmental News No. 70 2004. Danish Ministry of the Environmen. Danish Environmental Protection Agency: 46p.

Life cycle costing of farm milk production – cost assessment of carbon footprint mitigation strategies

Anne C. Asselin-Balençon*, Olivier Jolliet

School of Public Health (SPH), University of Michigan, 6622 SPH Tower, 1415 Washington Heights, Ann Arbor, Michigan 48109-2029, United States

* Corresponding author. E-mail: *aasselin@umich.edu*

ABSTRACT

One point of concern with the present Life Cycle Costing (LCC) approach is the arbitrary choice of the discount rate, which does not differentiate between economic objectives and effective economic performance. Here, we propose an alternate methodology and apply it to two examples. A multi-year method is used to simultaneously assess cumulated cost and impact variations. On the cost aspect, our method assesses the net present value (NPV) of the investment, as well as its internal rate of return (IRR). We graphically compare those values to the cumulated environmental impact over the same period. The case studies focus on two energy saving alternatives on the farm and the installation of a digester. The internal rate of return is a synthetic metric of economic performance that avoids the arbitrary choice of a discount rate; this indicator can be compared to an economic objective for decision making purposes.

Keywords: life cycle costing, discount rate, life cycle assessment

1. Introduction

Agriculture is a significant contributor to environmental impacts. A study by the University of Arkansas showed that 70% of the carbon footprint of US milk occurs at or before the farm gate (Thoma et al. 2012). Milk is also a significant contributor to water use and land use, as shown by an on-going comprehensive milk life cycle study (University of Michigan, 2011).

For decision making, however, environmental impact assessment should be consistently complemented by an economic assessment. Environmental Life Cycle Costing (LCC), as described in the introduction of Ciroth et al. (2008), combines environmental and economic assessments over the life cycle of the product: "Environmental life cycle costing summarizes all costs associated with the life cycle of a product that are directly covered by one or more of the actors in that life cycle; these costs must relate to real money flows. […] A complementary life cycle assessment (LCA), with equivalent system boundaries and functional units, is also required, Environmental LCC is performed on a basis analogous to that of LCA […]." One of the most useful features of a combined LCA and LCC approach is in the assessment of a mitigation scenario relative to a reference situation, enabling comparison of both costs and environmental impact between these scenarios.

For products with a life time longer than a year, a multi-year assessment has to be performed. Regarding the cost part, "the use of discounted cash flows for money flows occurring at different times within a product life cycle is commonly applied" (Huppes et al., 2008) .

One point of concern of that approach is the arbitrary choice of the time value for money, also known as the discount rate, which does not differentiate between economic objectives and effective economic performance. On the other hand, the environmental performance (e.g. 20% CO_2 saved) is clearly estimated and reported independently from the environmental objective (e.g. objective of 10% minimum impact reduction). Huppes et al. (2008) acknowledges this fact and recommends sensitivity analysis and peer review.

In this paper, we propose an alternate methodology to avoid the use of discount rate, focusing on three main objectives:

1. Elaborate a cost metric that is independent of any arbitrary discount rate,
2. Compare consistently environmental and cost performance cumulated over the lifetime of the equipment
3. Apply the approach to two case studies to determine the carbon footprint and cost performances of energy saving equipment and of an anaerobic digester.

2. Methods

2.1. LCC Actors and perspectives

Life cycle costing summarizes all the costs of the product. It can bear different points of view. If we consider milk, consumers will be concerned with the product cost in the retail shop, and with the costs associated with milk storage and preparation (e.g. refrigeration cost). The dairy farmer, on the other hand, will account for the price of feed, dairy infrastructure, and equipment, trying to lower his milk production costs and maximize

his margin on milk sales. The feed producer will be concerned with the cost of land, production inputs, and the possible cost of irrigation, with his goal being to lower his feed production costs.
The scope of this paper focuses on the perspective of the person that implements the mitigation scenario, in this case the dairy farmer. We also aim at looking at reduction in costs and impacts associated to mitigation measures, assessing the relative change rather than the absolute values. Applying and comparing the different economic assessment methods and metrics described below, we thus get an initial estimate of the costs or savings that can be expected from the mitigation scenario.

2.2. Cost assessment approaches and metrics

Two cost assessment approaches were used:

a) The "Net present value (NPV)" approach: in this case, the producer has to initially invest for the scenario; during operations, the producer may get cost reductions or additional revenues. Using the method of discounted cash flows, we compute the NPV of the alternate scenario's costs or savings as:

$$\text{Cost NPV (r)} = \sum_{i=0}^{i=LT} \frac{CF_i}{(1+r)^i} \qquad \text{Eq.1}$$

where LT = lifetime of investment
r = discount rate
CF_i= total cash flows of year i

b) The "internal rate of return (IRR)" approach: this method is complementary to the cost NPV method. The IRR corresponds to the discount rate that makes the NPV equal to zero.

$$\text{IRR that makes} \quad \text{Cost NPV (IRR)} = 0$$

$$\text{where Cost NPV (IRR)} = \sum_{i=0}^{i=LT} \frac{CF_i}{(1+IRR)^i} = 0 \qquad \text{Eq. 2.}$$

The higher the IRR of a scenario, the more profitable it is. The IRR is directly comparable to the minimum target rate of profitability for the scenario. If it is higher than the target, then it will be acceptable according to the standards of the farmer. If it is lower, then it will not be acceptable according to the same standards. The advantage of the IRR is that it is independent of the chosen discount rate as well as the interest rate. It can be compared to the interest rate (e.g. 6%) that represents a break-even situation or to a higher minimum profitability objective (e.g. 10%). A similar approach can be defined for the environmental assessment, clearly separating performance and objectives for the reduction of environmental impacts.

2.3. Cumulated cost and impact assessment approach

The cost assessment approach is compared to the environmental impact assessment, keeping the two metrics separate and reporting a) the change in environmental performance from year 0 to the end of life of the equipment as a function of b) the corresponding cumulated costs, discounted at the interest rate.

a) The total environmental impact is computed differentially over the lifetime between the reference situation and the alternate situation. It is equal to the sum of the yearly impact differential. Environmental impacts are not discounted; we assume that a change in impacts in year n has the same value as a change in impact in year zero. Only the carbon footprint impact is studied in this paper. To ensure consistency, externalities due to reduction in GHG emissions are not accounted for in the cost computations, as they are computed and reported separately as environmental impact.
b) The total discounted costs are computed over the lifetime of the investment with the cost NPV formula (Eq. 1), with a discount rate, *r*, equal to a representative long-term loan rate that the farmer could obtain on the market.

The internal rate of return (IRR) computation (Eq.2) is also computed and reported separately on the graph.

3. Results and discussion: energy saving case study

3.1. Description

In 2010, in collaboration with Dairy Management Inc., Ensave performed an energy audit of a farm located in New York state. The audit encompassed all sources of energy used for dairy activities, such as electricity, fuel and propane. The recommendations of the audit were twofold:

- Replace existing lighting fixtures with High Performance (HP) T8 fluorescent fixtures or T5 High-Output (HO) fluorescent fixtures, both especially designed for agriculture applications. These fixtures and bulbs are more energy efficient.
- Replace existing barn ventilation fans with more energy efficient 36” circulation fans.

This case study assesses the differences in GHG impact and in costs for those installations compared to the reference scenario. Table 1 lists the main data for the LCA computations.

Table 1. LCA data for the energy efficient equipment case study

Type of data	Value	Reference
Total herd	1,710 cows	Ensave report
Annual milk production	21,233,225 kg/yr	Ensave report- assumed FPCM
GHG impact of electricity	0.84 kg CO_{2e}/kWh	Asselin-Balençon et al., 2012
GHG impact of $1 lighting equipment	0.93 kg CO_{2e}/$ produced	US 1998 Input Output data for “lighting fixtures and equipment” section
GHG impact of $1 ventilation equipment	0.60 kg CO_{2e}/$ produced	US 1998 Input Output data for “electrical” section (SimaPro databases manual)

The cost data used are listed in .

Table 2. Most of the specific data come from the Ensave report. Expected lifetime of equipment has been assessed by expert judgement (farmer, Dec 2011). The discount rate has been set at the long-term interest rate.

Table 2. Cost characteristics for the energy efficient equipment case study

	Unit	Lighting investment	Ventilation investment
Cost of investment	$	39,219	21,700
Expected lifetime	Year	10	7
Accounting lifetime	Year	5	5
Estimated annual energy saving	kWh/year	144,768	34,298
Total electricity consumed on farm in reference scenario	kWh/year	1,244,440	
Initial price of electricity	$/kWh	$0.073 (Ensave, 2010)	
Discount rate= Long term interest rate	%	6	
Annual inflation (2001-2011)	%	2.5 (U.S. DoL, 2012)	
Annual variation of electricity price (2000-2010)	%	3.7 (U.S. EIA, 2012)	

3.2. Cumulated results and Net Present Values

The cumulated costs and carbon footprint differentials are shown in Fig. 1. Year 0 represents the year of investment. There is a positive carbon footprint differential due to the manufacturing of the new equipment, and a positive cost differential due to the cost of the new equipment. Year after year, the new equipment generates energy savings, which translates into both cost and carbon footprint savings; those savings are cumulated. For the year equal to the lifetime of the investment, the final datapoint gives the total cumulated cost and impact for the scenario.

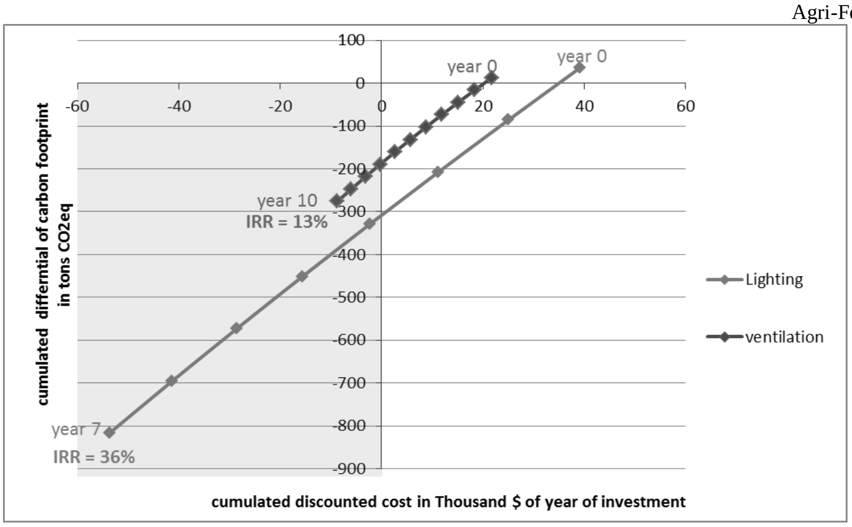


Figure 1. Cumulative carbon footprint as a function of the cumulated discounted costs at the long term interest rate, differentiated over the expected lifetime of investment. The corresponding internal rate of return (IRR) is given below the final cumulated value at end of life.

Over the lifetime of the investment, the ventilation equipment presents a discounted cumulated profit of \$9,000, equal to 40% of the initial investment cost. The IRR of this investment is equal to 13%. It also generates a significant reduction in GHG of nearly 300 tons CO_2e over the lifetime of the equipment.

The lighting equipment reaches the economic break-even point early, at year 3. Over the lifetime of the equipment, it generates a discounted cumulated profit (NPV) of \$ 54,000, equal to 137% of the initial investment cost. The IRR of this investment is very high at 36%. The effect on the reduction of GHG is over 800 tons of CO_2e over 7 years. We can graphically see that the lighting investment is economically more interesting than for the ventilation equipment. This difference is directly reflected in the IRR. The ventilation scenario's IRR is 13%, whereas the lighting IRR is 36%. However, both scenarios generate a profit over their lifetime, and this can be identified by the fact that their IRR is above the discount rate of 6%. A scenario with an IRR equal to 6% would generate a cumulated profit equal to zero. A typical minimum expected rate of return is 10%.

The discount rate value selected has an important influence on the assessment of economic performance; in most cases, most of the costs occur in the first years, and most of the benefits occur in the following years. On the graph, the discount rate is taken equal to the long-term interest rate, so that the yearly evolution represents the effective gain or loss actualised at year 0. This is equal to a net present value of \$ 54,000 for the lighting and \$9,000 for the ventilation. A higher discount rate of e.g. 10% can lower the economic benefit of the use phase and would lead to a lower net present value of \$41,000 for the lighting and \$3,500 for the ventilation. The advantage of calculating the internal rate of return is that it is independent of the arbitrary choice of a discount rate.

4. Discussion and sensitivity study for the energy saving case study

Two main parameters directly influence the profitability of the scenario: the expected lifetime of the equipment and the price of electricity. We study the sensitivity of the IRR to different values of these parameters, varying them from low to high lifetimes, and from low to high electricity price. Table 3 compiles the resulting internal rates of return.

Table 3. IRR of the scenarios with varying equipment lifetime and electricity price.

		Lighting			**Ventilation**		
		Low	**Base case**	**High**	**Low**	**Base case**	**High**
Estimated lifetime		5 years	**7 years**	10 years	7 years	**10 years**	12 years
Price of electricity	\$0.07/kWh	13%	22%	27%	-3%	6%	9%
	\$0.10/kWh	29%	**36%**	40%	6%	**13%**	16%
	\$0.12/kWh	39%	45%	48%	12%	18%	20%

For lighting, the IRR of 36% is much higher than the long-term interest rate of 6%, and is thus financially very interesting; this scenario remains of interest for shorter lifetimes and/or for lower electricity prices. The IRR is very sensitive to a lower price of electricity: it drops to 22% if the price of electricity is lower by

30%, down to $0.07 per kWh, and even to 13% if the life time is equal to 5 years. For ventilation, the IRR is lower. Three cases have an IRR lower than the long-term interest rate of 6%, and among them, one has a negative IRR. It is crucial for the farmer to ensure a longer lifetime and a higher price of electricity to avoid losing money.

The longer the expected lifetime, the higher the IRR. However, this lifetime must be consistent with the technical characteristics of the project. If one chooses a higher expected lifetime, he has to account for additional maintenance or repair costs that will be needed. In order to avoid this distortion, we propose to use the most likely expected lifetime rather than a conservative lifetime often used in financial accounting method. Sensitivity analysis on this variable can also be carried out.

5. Results and discussion: Digester case study

This section provides a summary of results for a second case study based on a dairy farm of 1100 cows located in upstate New York. In the scenario, the farm changes from slurry manure storage in a top-loaded open tank that would be emptied every 4 to 6 months to a 200 MW anaerobic digester, producing electricity and selling it to the grid.

We study two revenue scenarios: low and high. In the low scenario, the equipment is financed by the farmer, and the price of electricity is an average market price of 0.11 $/kWh. In the high scenario, we added an up-front grant, and we model a higher price of electricity of 0.17 $/kWh, which could be anticipated in the future. The impact reduction from the electricity generated by the digester was accounted for through a system expansion and the substitution of an average US electricity mix.

Table 4 presents the main assumptions and results of this study. The low revenue scenario does not reach a profit over the lifetime of the equipment: the NPV of the scenario is equal to $-540,000; this is also reflected in its IRR, equal to 2% and hence lower than the discount rate of 6%. Conversely, the high revenue scenario is profitable, and displays a profitable NPV equal to $1,020,000 ; its IRR is equal to 16%, well above the discount rate. Both scenarios generate a reduction of GHG of 45 million tons $CO_{2e.}$

Table 4. Main assumptions and results in the anaerobic digester case study

Scenario	Low revenue	High revenue	Units
Initial investment	1,467,000		$
Upfront grant	0	35	% of investment
Estimated lifetime	20	20	year
Discount rate =long term interest rate	6		%
Annual electricity produced	1,090,953		kWh/year
Price of electricity	0.11	0.17	$/kWh
Carbon footprint of US electricity production	0.73 (Rotz et al., 2011)		kg CO_{2e} / kWh
Net present value (at long term interest rate) of total costs and revenues over lifetime	-540,000	+ 1,020,000	$
IRR	2	16	%
total carbon footprint differential over lifetime	45		million tons CO_{2e}

This case study also displays how the economic profitability depends on the background economic condition; namely the grant amount and the price of electricity. In both cases, the reduction in greenhouse gases is significant.

6. Conclusion

In LCC, the internal rate of return enables a direct and straightforward metric for the economic performance of a scenario. Compared to the NPV method, it avoids the arbitrary choice of a discount rate. The higher the IRR, the more profitable the scenario. The IRR can be directly compared to the long-term interest rate as a break-even point for profitability. The IRR can also be compared with a higher financial objective of, e.g., 10% to assess the financial interest of the investment over its lifetime; if the IRR is lower, the scenario is below the objective, and may not be financially interesting.

Determination of the IRR has to be carefully done: specific care has to be taken in defining the cost data, and especially the lifetime of the investment and the most important revenue sources.

In the LCC context, the IRR metric can be combined with the cumulated impact assessment, providing decision makers with a comprehensive performance measurement, both environmental and financial.

7. Acknowledgements

This work was funded by the Innovation Center for U.S. Dairy. We would like to thank Marty Heller of University of Michigan for his constructive comments and careful reading.

8. References

Asselin-Balençon A. C., Popp J., Henderson A., Heller M., Thoma G., Jolliet O., 2012. Dairy farm greenhouse gas impacts: a parsimonious model for a farmer's decision support tool. Int.Dairy J. Submitted.

Ciroth A., Huppes G., Klopffer W., Rudenauer I., Steen B., Swarr T., 2008. Executive Summary in : Hunkeler D., Lichtenvort K., Rebitzer G. (Eds.), Environmental Life Cycle Costing. Setac, New York, US. Page xxvii.

Ensave, 2010. Ensave Farm Energy Audit Report (Spruce Haven) September 2010. Audit financed by Dairy Management Inc.

Huppes G., Ciroth A., Lichtenvort K., Rebitzer G. Schmitdt W., Seuring S. 2008. Modelling for life cycle costing in : Hunkeler D., Lichtenvort K., Rebitzer G. (Eds.), Environmental Life Cycle Costing. Setac, New York, US. Pages 17-34.

Rotz, C. Alan, Michael S. Corson, Dawn S. Chianese, and Colette U. Coiner. 2011. The Integrated Farm System Model (IFSM): Reference Manual Version 3.4. Washington, DC, USA: Pasture Systems and Watershed Management Research Unit - Agricultural Research Service - United States Department of Agriculture. http://www.ars.usda.gov/main/docs.htm?docid=8519.

Thoma, G., Popp, J., Shonnard, D. R., Nutter, D., Ulrich, R., Matlock, M. M., …Mayes, A., 2012. Greenhouse Gas Emissions from Milk Production and Consumption in the United States: A cradle to grave life cycle assessment circa 2008. Int.Dairy J. Submitted.

U.S. DoL (Department of Labor), 2012. Bureau of Labor Statistics. Consumer Price Index, All Urban Consumers - (CPI-U), U.S. city average, All items. ftp://ftp.bls.gov/pub/special.requests/cpi/cpiai.txt

U.S. EIA (Energy Information Administration), 2012. Electric Power Annual 2010. Table ES1. http://www.eia.gov/electricity/annual/

Comparing integral sustainability performance of conventional farms with farms focussing on internal recycling of nutrients

Mark A. Dolman[1,2,*], Marthijn P.W. Sonneveld[3], Herman Mollenhorst[2,4], Imke J.M. de Boer[2]

[1] Agriculture Economics Research Institute, part of Wageningen University and Research centre, The Hague, The Netherlands
[2] Animal Production Systems Group, Wageningen University, Wageningen, The Netherlands
[3] Soil Geography and Landscape Group, Wageningen University, Wageningen, The Netherlands
[4] Business Economics Group, Wageningen University, Wageningen, The Netherlands
[*] Corresponding author. E-mail: *mark.dolman@wur.nl*

ABSTRACT

Sustainable milk production systems require economically viable, environmentally sound and socially acceptable practices. We compared sustainability indicators of a group of conventional farms with a group of farms aiming at improved internal nutrient cycle (INC) on their farm. Economic performance was based on net farm income (NFI) and labour productivity. Environmental performance indicators were derived from a cradle-to-farm-gate life cycle assessment, i.e., land occupation (LO), non-renewable energy use (NREU), global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP). Moreover, soil- and water quality were monitored. Societal performance was quantified with payments for ecosystem services, grazing hours and penalties for aberrant milk composition. INC farms positively differ from conventional farms in a lower non-renewable energy use, higher soil organic carbon and receiving higher payments for ecosystem services, without compromising on economic performance. High standard deviations for other indicators suggest that differences within both groups are mostly higher than between groups.

Keywords: sustainability, FADN, dairy farming, nutrient cycling

1. Introduction

A commonly used definition on sustainability is to meet the needs of current generations without compromising the ability of future generations to meet their needs and aspirations (WCED, 1987). Sustainable milk production systems require economically viable, environmentally sound and socially acceptable practices (Thomassen et al., 2009). Besides delivering marketable products, the delivery of products and services related to the quality of the environment, i.e. ecosystem services, is becoming increasingly important. Such services include the value of the landscape and the contribution to biodiversity, water storage and supply of clean groundwater (MEA, 2005). Soil quality is a key aspect with respect to ecosystem services (Bennet et al., 2010). Land use by agriculture, however, rapidly intensified, causing deterioration of soil structure, impoverishment of the soil and a decline in soil organic matter (EU, 2006). This deterioration of soil quality threatens the future food production and other important ecosystem services. The supply of ecosystem services in the Netherlands is originating from specific landscapes, for example the National Landscape 'Noordelijke Friese Wouden (NFW)' in the North of the Netherlands. Several farmers in this area follow a farming practice, aiming at an improved internal nutrient cycle (INC) on their farm. In order to minimize use of external inputs, INC farms focus on optimizing use of on-farm available resources, for example, soil organic matter, nutrients from manure and home-grown feed production (Van Hees et al., 2009). INC farms, therefore, can make a significant contribution to preservation of food producing capacity, while additionally producing ecosystem services. An integral assessment that quantifies the effect caused by this INC approach, however, is never performed. Several studies have assessed the environmental performance of INC farms in the past (Sonneveld et al., 2008). So far, however, no life cycle assessment (LCA) of INC farms has been performed. Moreover, an integral assessment, including economic, environmental, and societal indicators has never been performed.

The objective of this study, therefore, is to quantify the economic, environmental and societal performance of INC farms in the NFW and, secondly, benchmark with a conventional milk production system comparable in terms of farm size, intensity and site-specific circumstances.

2. Methods

2.1. Data

Data needed to quantify economic, environmental and societal indicators of farms were derived from the farm accountancy data network (FADN) and the Minerals Policy Monitoring Programme. We quantified a two-year average sustainability performance of nine INC farms (2008-2009). For every INC farm, a conventional 'mirror-farm' was composed from farms in FADN, using statistical matching. Central assumption for statistical matching is that farms resembling for the imputation variables, also resemble for the goal variables, in this case sustainability performance indicators (Vrolijk et al., 2005). For every INC farm, ten similar conventional farms were selected, i.e., nearest neighbours, based on farm size (fat and protein corrected milk (FPCM) production), intensity (FPCM production per ha) and site-specific circumstances, i.e., soil nitrogen supply level, percentage of sandy, peat or clay soil, and percentage of poorly, moderately well or well drained land. The ten farms selected per INC farm, are scaled to one, so they make up one farm matching one INC farm.

2.2. Economic performance

To quantify economic performance we quantified net farm income (NFI) and labour productivity. NFI is often used as an indicator for profitability (Dekker et al., 2011, Shadbolt et al., 2009, Blank et al., 2009, Van Calker et al., 2008). We defined NFI as the remuneration for management, family labour and capital that is left after all other costs are paid. To correct for differences in farm size we expressed NFI per 100 kg FPCM. NFI, however, does not account for input of family labour. To give insight in the labour effort to realise the NFI, a measure of labour productivity is required. Labour productivity is a ratio of a volume of output per unit of labour input (OECD, 2001). To enable a comparison of labour productivity in hours of labour among farms differing in scale, we expressed labour productivity in minutes of labour also per 100 kg FPCM.

2.3. Environmental performance

The environmental performance of farms was based on indicators derived from a cradle-to-farm-gate LCA, i.e., land occupation (LO), non-renewable energy use (NREU), global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP), expressed per unit of FPCM. All farms were highly specialised dairy farms, without other types of animal production. We performed an attributional LCA. Whenever a multifunctional process occurred, economic allocation was used. For EP and AP we took characterisation factors from Heijungs et al., (1992), for GWP from IPCC (2007).

Production of feed
On average, 71% of all feed was home grown for the farms in this study. All home grown feed was sampled. Furthermore, all non-monetary input of pesticides, fertilisers, water, and energy was recorded in FADN, as well as the application method and quantity of the animal manure (at field level). In the Netherlands, farms are obliged to use low-emission techniques to apply animal slurry to the land. In the past, INC farms have received a temporary exempt from this regulation because of assumed lower ammonia losses under their specific practices. For each farm, detailed information on purchased feed was available, i.e. exact quantity used per feed product, and dry matter (DM), energy (VEM), nitrogen (N) and phosphorus (P) contents. The average composition of compound concentrates (i.e. main feed type) was based on monthly publications of Nevedi (2008-2009). For each feed ingredient used, the environmental impact of crop cultivation, processing and transport was based on Thomassen et al., (2009) and additional empirical data, literature or expertise from feed processing companies.

Farm specific excretion and gaseous losses
For each individual farm, we computed the excretion of N and P via manure by subtracting N and P fixed in milk and animals from the total uptake of N and P in feed. Uptake of non-grass products was known. In order to quantify the intake of grass, the energy requirement of dairy cattle was computed per farm, based on the level of milk production, breed, pasturing and housing system and number of animals including young stock. Grass intake was computed by subtracting the energy uptake from non-grass feed from total energy requirement. Subsequently, the energy uptake from grass was converted into N and P uptake, based on the energy-nutrient ratio. The fixation of N and P in milk and animals was subtracted from the total uptake, resulting in the gross N and P excretion (Anonymous, 2010a). Subsequently, for each farm gaseous N losses were computed based on the total ammoniacal nitrogen (TAN) in manure. We assumed a TAN value of 68% and 77% of the mineral N for dairy slurry in respectively the winter and summer period. (Van Bruggen et al., 2011). Emission of NH_3 was assumed to depend on housing system (Anonymous, 2010b). The NH_3-emission factor varied between 10.9 and 15.4% of TAN in the winter period, whereas in the summer period it varied between 10.1 and 31.0%. Emissions of NO_x and N_2O were computed as 0.3% of TAN in manure for all housing systems (Velthof et al., 2010; Oenema et al., 2000). Emissions in the pasture were computed based on Velthof et al., (2010) and using the farm specific manure production and an emission factor of 3.5% of TAN for NH_3.

On-farm soil and water quality indicators

To quantify site-specific impacts, we determined on-farm AP and EP per hectare and measured on-farm field parameters, i.e., soil organic matter (0-10 cm (%)), soil phosphorus content (P-Al, mg/100 gram soil) and soil nitrogen supply (kg/ha). Moreover, phosphate and nitrate concentrations in the upper groundwater (mg/l) were monitored. Soil organic matter has a positive effect on water supply, soil structure and nutrient availability. The P-Al number is a measure of the capacity of the soil to supply phosphorus (P) and gives a rough indication of the level of P-saturation and thus the amount of P that potentially may be lost to the environment. The soil nitrogen supply is a measure of the N-supply in an unfertilised situation.

2.4. Societal performance

We quantified societal performance using three indicators: payments for ecosystem services (Euro/ha), grazing (hours/cow) and penalties for aberrant milk composition (%). Data on working conditions and landscape quality are not available in FADN, and, therefore, not included (Van der Veen et al., 2006). Payment for ecosystem services is regarded here as a proxy for the farmers' investment (time or money), for nature conservation or bird protection. The number of penalty cases is provided as a measure of food safety. Farmers get a penalty when the milk contains high somatic cell, or bacterial counts. The number of hours grazing is included as an indicator for welfare and social perception. We acknowledge that pasture hours is a simple indicator for welfare. Current FADN does not have the potential to assess animal welfare using preferred animal-based indicators, such as, the Welfare Quality Protocol (De Vries et al., 2011; Welfare quality, 2009).

3. Results

3.1. Descriptive

Both groups of farms were comparable in terms of farm size, intensity and milk production (Table 1). The proportion of grassland for INC farms was significantly higher (P< 0.05) than for conventional farms. The conventional farms only had sandy and clay soils, INC farms had a small proportion of peat soil as well (P<0.05).

Table 1. Weighted mean and standard deviations (SD) of farm characteristics of conventional farms and farms aiming at an improved internal nutrient cycle (INC) (average 2008-2009).

Characteristic	Conventional mean (SD)	INC mean (SD)	Sig [b]
Farm land			
Utilized agricultural area (ha)	45.9 (21.5)	50.1 (19.0)	
Grassland (%)	87 (11)	97 (6)	*
Arable land (%)	13 (10)	3 (5)	*
Sandy soil (%)	64 (49)	60 (44)	
Peat soil (%)	0 (0)	1 (1)	*
Clay soil (%)	36 (49)	39 (44)	
Poorly drained land (%)	22 (20)	19 (18)	
Moderately well drained land (%)	75 (21)	79 (17)	
Well drained land (%)	3 (9)	2 (5)	
Animal production			
Cows (#)	69.5 (33.8)	74.8 (35.2)	
Total milk production (kg FPCM[a])	553,570 (273,057)	573,525 (255,106)	
Milk production per cow (kg FPCM)	7,960 (859)	7,663 (691)	
Milk production per ha (kg FPCM)	12,092 (1,537)	11,439 (1,565)	

[a] Fat-and-Protein-Corrected Milk. [b] *:P<0.05; ** P<0.01; ***:P<0.001

3.2. Performance indicators

For economic performance, no significant differences were observed. There was a tendency (P=0.19) that INC farms had a higher NFI (Table 2). Less efficient labour productivity partially explains the higher farm income, since no family labour costs were included in NFI.

For environmental performance, on-farm, off-farm and total non-renewable energy use were significantly lower for INC farms. INC farms had a lower diesel use than conventional farms. For the years under investigation, INC farms used aboveground spreading application techniques that are less energy consuming compared to conventional application techniques. Although INC farms had numerical lower impact per unit of

FPCM for most LCA indicators, these differences were not significant. High standard deviations indicate that differences within groups are higher than between groups. For example, the coefficient of variation for the nitrate and phosphorus concentration, is higher than 100% for both groups. INC farms did perform significantly better than conventional farms regarding soil organic carbon (P<0.05). INC farms traditionally have a high soil organic carbon, since almost no grassland is renewed.

For societal performance, INC farms outperformed conventional farms for payment for ecosystem services (P<0.01). INC farms received higher payment for bird protection programme. The payment compensates farmers for lower yields due to harvesting after breading season. The grass yield, however, did not significantly differ between groups. The grass yield on INC farms and conventional farms was respectively 10,450 and 10,700 kg dry matter. For grazing hours and penalties for aberrant milk composition, INC farms not significantly differed from conventional farms.

Table 2. Weighted mean and standard deviations (SD) of sustainability performance of conventional farms and farms aiming at an improved nutrient cycle (INC) (average 2008-2009)

Performance indicator	Conventional mean (SD)	INC mean (SD)	Sig b)
Economic performance			
Labour productivity (minutes labour / kg FPCM)	39 (13)	42 (18)	
Farm income (€ / 100 kg FPCM)	5.71 (6.84)	8.26 (6.54)	
Environmental performance (LCA)			
Land occupation			
On-farm (m^2 / kg FPCM)	0.8 (0.1)	0.8 (0.1)	
Off-farm (m^2 / kg FPCM)	0.6 (0.3)	0.5 (0.1)	
Total (m^2 / kg FPCM)	1.4 (0.3)	1.3 (0.2)	
Non-renewable energy use			
On-farm (MJ / kg FPCM)	1.0 (0.3)	0.8 (0.2)	*
Off-farm (MJ / kg FPCM)	5.0 (1.0)	4.3 (0.8)	*
Total (MJ / kg FPCM)	5.9 (1.0)	5.0 (0.8)	*
Global warming potential			
On-farm (kg CO_2eq / kg FPCM)	0.8 (0.1)	0.8 (0.1)	
Off-farm (kg CO_2eq / kg FPCM)	0.6 (0.1)	0.5 (0.1)	
Total (kg CO_2eq / kg FPCM)	1.4 (0.2)	1.3 (0.2)	
Acidification potential			
On-farm (g SO_2eq / kg FPCM)	5.7 (2.9)	5.4 (3.8)	
Off-farm (g SO_2eq / kg FPCM)	5.0 (1.1)	4.3 (0.8)	
Total (g SO_2eq / kg FPCM)	10.7 (3.3)	9.7 (4.0)	
On-farm (kg SO_2eq / ha)	72 (38)	71 (51)	
Off-farm (kg SO_2eq / ha)	82 (21)	87 (16)	
Total (kg SO_2eq / ha)	76 (21)	77 (30)	
Eutrophication potential			
On-farm (g NO_3-eq / kg FPCM)	32.9 (16.7)	30.3 (20.5)	
Off-farm (g NO_3-eq / kg FPCM)	45.4 (14.9)	35.1 (8.7)	
Total (g NO_3-eq / kg FPCM)	78.2 (24.7)	65.4 (26.2)	
On-farm (kg NO_3-eq / ha)	413 (207)	400 (426)	
Off-farm (kg NO_3-eq / ha)	752 (209)	712 (162)	
Total (kg NO_3-eq / ha)	559 (145)	523 (170)	
Environmental performance (on-farm soil and water quality)			
Soil Organic Carbon grassland (ton / ha)	152 (40)	186 (42)	*
P-Al grassland (mg / 100 gram soil (0-10 cm))	38 (9)	36 (4)	
Soil Nitrogen Supply grassland (kg / ha)	191 (18)	196 (13)	
Nitrate concentration (mg NO_3^- / litre)	22 (22)	12 (12)	
Phosphorus concentration (mg P / litre)	0.2 (0.2)	0.7 (1.1)	
Societal performance			
Grazing (hours / cow)	2,509 (1,305)	2,006 (1,312)	
Payment for ecosystem services (€ / ha)	24 (46)	166 (175)	**
Penalties for aberrant milk composition (%)	2.4 (3.9)	1.4 (3.1)	

[a] Fat-and-Protein-Corrected Milk. [b] *:P<0.05; ** P<0.01; ***:P<0.001

4. Discussion

INC farms were compared to conventional farms, which were comparable in farm size, and site-specific circumstances using statistical matching. Central assumption is that these indicators together resemble sustainability performance. Thomassen et al., (2009) showed that intensity and farm size were positively correlated with economic and environmental performance. Moreover, only nine INC farms were assessed. A higher number of farms is needed to improve robustness of comparing the INC farming system, with conventional practices, especially considering the fact that variability within the groups was substantial.

For economic performance, no significant differences were seen. However, NFI was 44% higher for INC farms. High standard deviations indicated that differences within groups were higher than between groups. It is common that economic performance differs highly among farms. There are also large fluctuations between the years, which also affects the outcomes of our analyses. The year 2008 was a relatively prosperous year, whereas 2009 showed the lowest income over the last decade (De Bont et al., 2009). INC farms received a higher income due to higher payments from bird protection programmes and used a higher amount of own labour, which was not included in NFI.

For environmental performance, INC farms outperformed conventional farms on energy use and SOC. Using above-ground spreading for slurry explains a lower on-farm energy use, because of lower diesel consumption application technique. On-farm acidification potential was expected to be higher due to a higher emission factor for aboveground spreading. INC farms, however, applied less artificial fertiliser and had a housing system with a lower emission factor, compensating the higher emission of applying animal manure. SOC was significantly higher for INC farms, partly because these farms were reluctant in applying regular grassland renovation. Results from other studies however also show that SOC contents are strongly controlled by soil type, drainage level and land use history, factors which are site-specific. Thus, the differences between both groups with respect to SOC content cannot purely be related to differences in current farm management.

For societal performance, current FADN does not have the potential to assess animal welfare using preferred animal-based indicators. Pasture hours, therefore, were used as an indirect indicator for animal welfare. This indicator showed no differences between both farm types. INC farms significantly differed from conventional farms for received payments on ecosystem services. These payments, however, did not reflect the actual gain in improved societal performance. A better indicator should be area based, or expressing biodiversity, e.g. number of plant species, or number of birds. The INC farms aims for improved nutrient cycling, by making use of all available resources on-farm. This approach goes along with social involvement and entrepreneurial complacency. These values, however, were not included in this assessment.

5. Conclusion

INC farms positively differ from conventional farms in a lower non-renewable energy use, higher soil organic carbon and receiving higher payments for ecosystem services, without compromising on their economic performance. However, significant differences between INC farms were limited, despite INC farm follow a farming practice aiming at optimizing use of on-farm available resources. Overall, most performance indicators did not show significant differences. High standard deviations indicate that differences within groups are mostly higher than between groups.

6. References

Anonymous, 2010a. Handreiking bedrijfsspecifieke excretie melkvee, versie per 1 januari 2010 van kracht. Ministry of Economic Affairs, Agriculture and Innovation, The Hague.

Anonymous, 2010b. Regeling Ammoniak en Veehouderij (RAV), article 2. Ministry of Infrastructure and the Environment, The Hague.

Blank, S. C., Erickson, K.W., Nehring, R., Hallahan, C., 2009. Agricultural profits and farm household wealth: a farm-level analysis using repeated cross sections. Journal of agricultural and applied economics 1: 207-225.

Bennet, L.T., Mele, P.M., Annett, S., Kasel, S., 2010. Examining links between soil management, soil health, and public benefits in agricultural landscapes: An Australian perspective. Agriculture Ecosystems & Environment 139: 1-12.

De Bont, C.J.A.M, Van Everdingen, W.H., Van der Knijff, A., Van der Meulen, H.A.B., 2009. Actuele ontwikkeling van resultaten en inkomens in de land- en tuinbouw in 2009. LEI, part of Wageningen UR. The Hague.

Dekker, S.E.M., De Boer, I.J.M., Vermeij, I., Aarnink, A.J.A., Groot Koerkamp, P.W.G., 2011. Ecological and economic evaluation of Dutch egg production systems. Livestock Science 139: 109-121.

De Vries, M., Bokkers, E.A.M., Dijkstra T., Van Schaik, G., De Boer, I.J.M., 2011. Invited review: Associations between variables of routine herd data and dairy cattle welfare indicators. J. Dairy Science: 94:3213–3228.

EU, 2006. Thematic Strategy for Soil Protection. Commission of the European communities, Brussels.

Heijungs, R., Guinee, J.,Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., Wagener Sleeswijk, A., Ansems, A.M.M., Eggels, P.G., VanDuin, R., De Goede, H.P., 1992. Environmental Life Cycle Assessment of Products. Guide and backgrounds. CML, Institute for Environmental Sciences, Leiden.

IPCC, 2007. In: Pachauri, R.K., Reisinger, A.(Eds.), Core Writing Team, Intergovernmental Panel on Climate Change. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva.

Nevedi, 2008-2009. Lineaire programmeringen rundvee-, varkens- en pluimveevoeders. Monthly reports on feed composition computed by Schothorst Feed Research BV. Lelystad.

MEA, 2005. Millenium Ecosystem Assessment. Ecosystems and Human Well-Being: Synthesis. Island Press, Washington.
OECD, 2001. Measuring productivity – OECD Manuel: measurement of aggregate and industry-level productivity growth. OECD Statistics Directorate, Division of structural economic statistics.
Oenema, O., Velthof, G.L., Verdoes, N., Groot Koerkamp, P.W.G., Bannink, A., Monteny, G.J., Van der Meer, H.G., Van der Hoek, K., 2000. Forfaitaire waarden voor gasvormige stikstofverliezen uit stallen en mestopslagen. Alterra, Wageningen UR. Wageningen.
Shadbolt, N., Kelly, T., Horne, D., Harrington, K., Kemp, P., Palmer, A., Thatcher, A., 2009. Comparisons between organic & conventional pastoral dairy farming systems: Cost of production and profitability. Journal of Farm Management 10: 31-45.
Sonneveld, M.P.W., Schröder, J.J., De Vos, J.A., Monteny G.J., Mosquera, J., Hol, J.M.G., Lantinga, E.A., Verhoeven, F., Bouma, J., 2008. A Whole-Farm Strategy to Reduce Environmental Impacts of Nitrogen. J. Environ. Qual. 37(1), p. 186-195.
Thomassen, M.A., Dolman, M.A., Van Calker, K.J., De Boer, I.J.M., 2009. Relating life cycle assessment indicators to gross value added for Dutch dairy farms. Ecological Economics 68, 2278–2284.
Van Bruggen, C., Groenestein, C.M., De Haan, B.J., Hoogeveen, M.W., Huijsmans, J.F.M., Van der Sluis, S.M., Velthof, G.L., 2011. Ammoniakemissie uit dierlijke mest en kunstmest in 2009. Berekeningen met het Nationaal Emissiemodel voor Ammoniak (NEMA). Statutory Research Tasks Unit for Nature and the Environment, Wageningen.
Van Calker, K.J., Berentsen, P.B.M., Giesen, G.W.J., Huirne, R.B.M., 2008. Maximising sustainability of Dutch dairy farming systems for different stakeholders: A modelling approach. Ecological Economics 65: 407-419.
Van der Veen, H., Oltmer, K., Boone, J.A., 2006. Het BIN-nenstebuiten: beschikbare gegevens in het Bedrijven-Informatienet Land- en Tuinbouw. LEI, part of Wageningen UR. The Hague.
Van Hees, E.M., Otto, A.A.C., Van der Schans, F.C., 2009. Van Top-down naar Bodem-up. Review van kringlooplandbouw in de melkveehouderij. Centrum voor Landbouw en Milieu, Culemborg.
Velthof, G.L., Van Bruggen, C., Groenestein, C.M., De Haan, B.J., Hoogeveen, M.W., Huijsmans, J.F.M., 2009. Methodology to calculate ammonia emissions by Dutch agriculture. Statutory Research Tasks Unit for Nature & the Environment, Wageningen.
Vrolijk, H.C.J., Dol, W., Kuhlman, T., 2005. Integration of small area estimation and mapping techniques - Tool for Regional Studies. LEI, part of Wageningen UR, Den Haag.
WCED, 1987. World Commission on Environment and Development: Our common future (The Brundtland report). Oxford University Press, Oxford, United Kingdom.
Welfare Quality®. 2009. Welfare Quality® Assessment Protocol for Cattle. Welfare Quality® Consortium, Lelystad, Netherlands.

Environmental improvement of a chicken product through life cycle assessment methodology

Guillermo Pardo[1,*], Ascensión Ciruelos[2], Nuria López[3], Lorenzo González[4], Saioa Ramos[1], Jaime Zufía[1]

[1,*]AZTI-Tecnalia
[2] CTAEX - Centro Tecnológico Agroalimentario de Extremadura
[3] AIMPLAS – Instituto Tecnológico del Plástico
[4] IAT – Instituto Andaluz de Tecnología
[*] Corresponding author. E-mail: *gpardo@azti.es*

ABSTRACT

Life Cycle Assessment methodology was applied to a chicken product in order to identify the more relevant sources contributing to environmental impact. Several potential improvements were proposed and developed at different stages of the food chain, based on a technical and environmental point of view, with special focus on the application of innovative solutions. Improved food chain was also evaluated through LCA in order to compare results. Significant impact reductions were achieved in the feed production stage by replacing conventional ingredients with tomato by-products in the poultry diet, (freshwater eutrophication decreased 9% and global warming effect 5%). Water consumption and wastewater generation was minimised about 16% in the slaughterhouse and meat processing stage through reutilisation alternatives and pulsed light decontamination. Conventional plastic tray and film were replaced with biodegradable materials avoiding up to 20% of greenhouse gases emissions. Finally product distribution was also optimised by improving refrigerated storage and logistics achieving relevant reductions in electrical and diesel consumption estimated around 10-15%.

Keywords: poultry, broiler, Life Cycle Assessment, ecodesign

1. Introduction

Spain is traditionally one of the major producers and consumers of chicken meat in the European Union. In the last years consumers demand has risen to 1.12 million tons accounting one of the highest consumption ratios of chicken meat in the UE (around 24 kg per capita), which represents 25% of the meat globally consumed in the Spanish market. At the same time the concern with environmental issues related to meat production, with special regard to acceptable levels of nitrogen and phosphorus, or the contribution of the sector to the emission of greenhouse gases, may lead to a significant increase in costs in the poultry sector.

From a multidisciplinary approach the project ECOALIM aimed to perform the eco-design of a meat product as a way to develop more efficient and sustainable food products throughout its whole life cycle. Four different research centres were involved to investigate innovative technologies and processes in their respective areas of expertise under the common objective of reducing the environmental impact of a food product along various stages of the supply chain (Figure 1).

The product chosen for the study was a 600 gr tray of sliced chicken breast fillets packaged in modified atmosphere, which is a commonly commercialised item in the Spanish market. The most significant impact sources were identified using life cycle assessment (LCA) methodology and several improvement approaches were selected at different life cycles stages, according to their potential to reduce the environmental load of the product and their suitability from a technical point of view. The following solutions were proposed:

- Processing of food industry by-products as valuable ingredients for animal feed.
- Pulsed light technology for water decontamination and re-use in food sector
- Development of a biodegradable packaging for meat products
- Energy and resources reduction in the distribution system by means of improved storage, routes optimisation and reverse logistics solutions.

Proposed improvements were studied and developed at each research centre involved in the project, according to their field of knowledge, and feasibility was evaluated for their implementation to the industry. Finally LCA methodology was applied in order to estimate the environmental impact of the new food chain and to compare results with the original scenario.

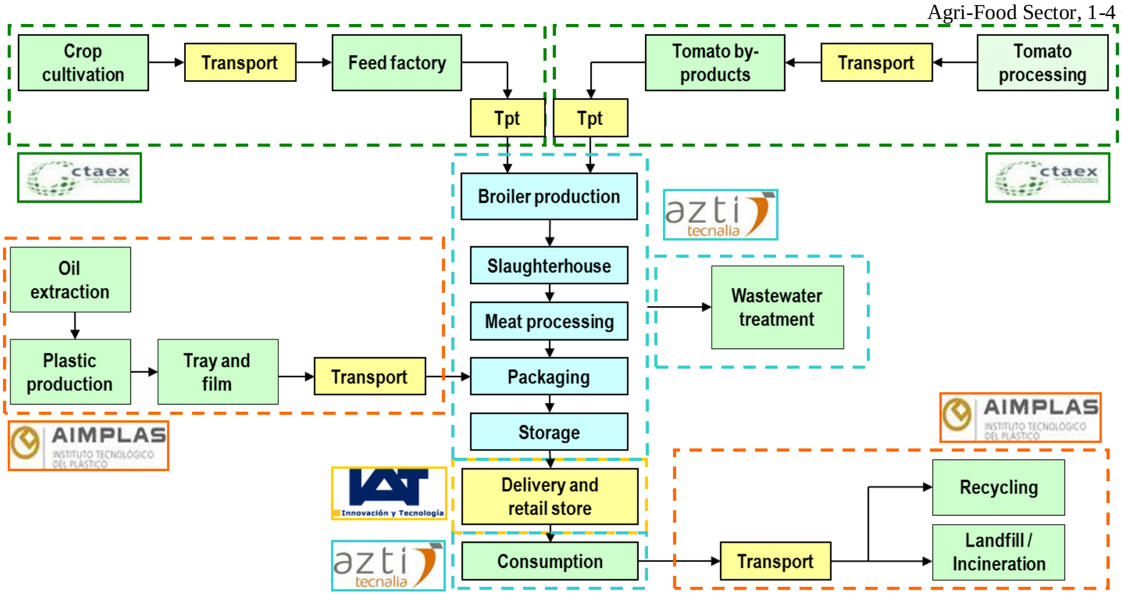


Figure 1. Overview of the life cycle of the chicken product and expertise areas of every partner.

2. Methods

Selected functional unit was a 0.6 kg tray of sliced chicken breast packaged in modified atmosphere. The study ranged all the stages of the life cycle of the product from cradle to grave, including consumption phase and packaging disposal. Mass allocation was performed by accounting the share in quantity of different chicken meat products from broiler production. Table 1 shows the allocation factors used. As can be observed, breast had a high allocation factor which pointed out that a relevant amount of the environmental burden related to the broiler production is ascribed to the analysed product.

Data for inventory were mainly acquired from contacts in industry and companies participating in the project, as well as peer-reviewed literature and goverment publications. Data from four farms (three for broiler production and one for chicks) and two poultry slaughterhouses was collected through questionnaires with regards to water and energy consumption, packaging, fodder use and related outputs. Maize, soybean, wheat, sunflower meal and corrector were identified as the main components in the formulation of poultry fodder. Data for inventory of crop cultivation were based on Nemecek (2007). Direct emissions of ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) from chicken house and poultry manure application were estimated according to EPER (2007) and EMEP-CORINAIR (2007). For soybean cultivation in Brazil the carbon dioxide (CO_2) emissions related to the land transformation from tropical rainforest were estimated according to Jungbluth et al., (2007).

Table 1. Chicken co-products and derived mass allocation factors.

Co-Product	Weight (g)	Allocation factor
Breast	480	0.32
Drumsticks	240	0.16
Thighs	500	0.34
Wings	267	0.18
Total (carcass)	1487	1.00

Average Spanish grid electricity data was modelled adapting ecoinvent electricity production mix. Other data for background systems such as transportation, fertiliser, plastic and fuel production were also derived from ecoinvent database. Plastic tray and logistics were modelled according to realistic packaging details and delivery routes from a Spanish chicken company. Refrigerant leakages and additional energy required for refrigeration during transport were accounted following reviewed literature (Tassou et al.,, 2009). Electricity for cooling at stores was estimated using refrigeration equipment details and product throughputs of ten supermarkets within the distribution area. Finally consumer habits related to purchasing, storage conditions, cooking, washing and waste disposal were modelled according to results from a specific market research survey (Zufía and Pardo, 2011).

Additional energy, water and material requirements for the proposed improvements were estimated mainly from prototypes and pilot scale trials carried out along the project and from contacts in industry and technology suppliers. Tomato by-products (seeds and peels) were proposed to be included in poultry diet as partial substitute of conventional ingredients. As shown in Figure 1, in the alternative feed formulation the proportion of soybean was reduced and the other components where adjusted to keep nutrient levels. However tomato by-products inclusion was limited to 3% due to high fiber content. Energy for drying process was measured on-site at CTAEX facilities.

Polylactide (PLA) based tray and film were studied as alternative packaging materials for chicken meat products. An additional co-extruded poly-vinyl alcohol (PVOH) layer was included to provide the gas barrier proprieties of modified atmosphere packaging. Vinyl acetate copolymer data was used as a proxy for PVOH. Data of the production process for PLA were based on Althaus et al., (2007).

Table 2. Standard and alternative broiler diet composition (%)

Components	Standard (%)	Alternative (%)
Corn	40.00	40.00
Wheat	16.38	17.40
Soybean	31.00	28.40
Sunflower seeds	6.00	5.07
Vegetable oil blend	3.00	2.53
Mono-calcium phospate	2.00	2.00
Limestone	0.60	0.60
Mineral and vitamin suplement	0.40	0.40
Salt	0.25	0.25
Lysine	0.18	0.18
Methionine	0.19	0.19
Tomato seeds and peels	0.00	3.00

Environmental impacts associated with chicken life cycle were estimated following ReCiPe characterisation method (Goedkoop, 2009). Based on the default list of impact categories elaborated by Guinée et al., (2001). Four impact categories were selected among the so-called “baseline impact categories”: Climate Change (CC), Ozone Depletion (OD), Acidification Potential (AP) and Eutrophication Potential (EP). Additionally, Water Depletion (WD) category was also considered, since water consumption has an important relevance in Spain, especially in Mediterranean areas and Cumulative Energy Demand (CED) (Jungbluth, et al., 2004) was chosen as an energy flow indicator.

3. Results and discussion

3.1. Life cycle impact assessment results

As previous studies on broiler production have pointed out (Leinonen et al., (2012), Pelletier (2007)) LCA results (Figure 2) showed that the major impacts to the environment are caused in the feed production stage, mainly from the crop cultivation phase, followed by on-farm operations such as heating of broiler housing or manure storage and spreading. Fodder production showed crucial impacts in most categories but especially for climate change (CC), ozone depletion (OZ) and freshwater eutrophication (FE) with a global contribution among 40-50%. In terms of global warming, N_2O emissions appeared most important (33%), followed by CO_2 emissions from burning fossil fuels and transport (31%) and CO_2 emissions from deforestation (26%). Soy used in the poultry feed has a particular influence in the analysed system since it is mainly imported from Brazil, which is associated to impacts linked to long distance transport and deforestation for grain production.

Environmental burdens related to broiler production were found to be the most relevant for AC (80%) and also significant for CC (15%) and FE (33%). This impact was linked to energy consumption, but especially to NH_3, N_2O and CH_4, emmisions derived from manure handling.

Slaughtering and meat processing involved low relative contributions around 10% for most of the impact categories, except for water depletion (36%) due to the important amount of water required for cleaning operations. Product storage and delivery accounted for 15% of the total impact for OZ, and 7% for CC, derived from emisions related to energy consumption from refrigeration but also refrigerant leakages during refrigerated transport.

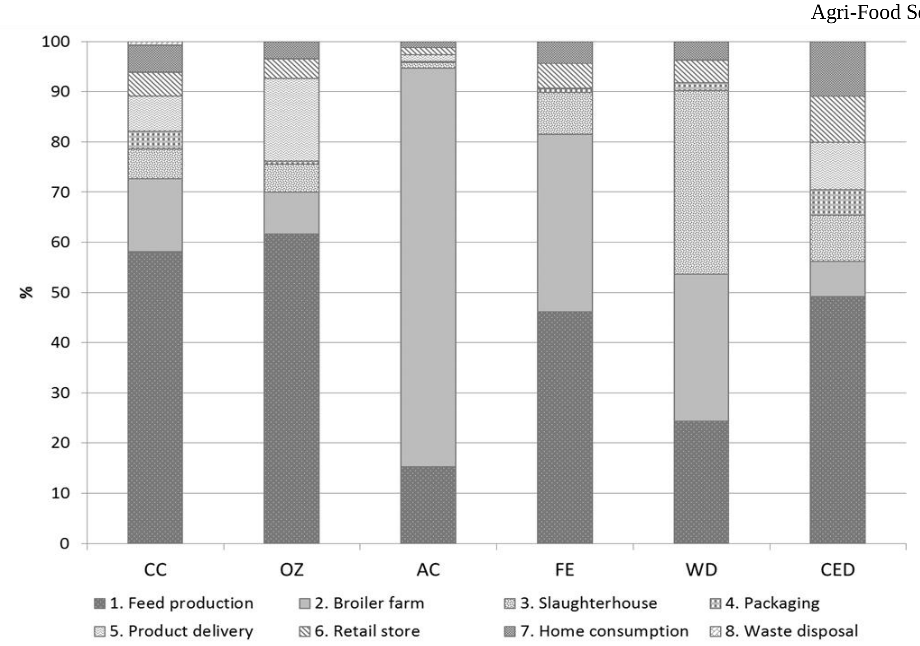


Figure 2. Main contributions from different stages to the environmental impact of the chicken product

In terms of cumulative energy demand, feed production was the dominant stage (around 50%) followed by home consumption (11%) and product delivery (10%), but other stages included in the analysed system represented significant contributions, such as slaughtering (9%), retail store (9%) and broiler production (7%).

Comparability of the results with previous LCA studies of poultry meat may be difficult due to differences in the methodology, especially with regards to the functional unit selected, system boundaries of the study and the method chosen for impact assessment stage. Nevertheless, results expressed per kg of chicken product were found to be of the same magnitude as several previous studies for CC (4.5 kg CO_2-eq/kg prod.) and AC (123.3 kg SO_2 eq/kg prod.) (Williams et al., 2006) (Leinonen et al., 2012) (Katajajuuri et al., 2007) (Seguin et al., 2011). Estimated values for FE (5,5g kg PO_4^{3-} equiv/kg prod.) were similar to those observed by Katajajuuri et al., (2007) but lower than most of literature reviewed, which can be partly attributed to differences in the methodology applied for impact assessment. CED results (46.6MJ/kg prod.) were very similar to shown by Katajajuuri et al., 2007 but in general higher than other studies. This can be explained attending to differences in the scope and functional unit selected for the analysis which often refers to poultry meat at farm gate. The current study ranged more stages of the life cycle of the product (from cradle to grave), which considers some high energy demanding phases, such as home consumption, retail storage and product delivery.

3.2. Improvement options identified

In the first place, since feed production was identified as one of the major contributors to the environmental burdens of the chicken item, the option of processing food industry by-products as valuable ingredients for poultry feed was explored. Tomato peels and seeds were dried and conditioned for its use as feed components (up to 3%) allowing to reduce the amount of imported soy in the formulation.

In relation to water consumption at slaughterhouse two main optimisation actions were identified. Firstly, direct reutilisation of water from washing boxes process to clean installations and transport lorries was pointed out. Secondly, internal water line from meat processing was selected for its potentially re-use in scalding after decontamination through pulsed light and additional clorination. Combining both options it was estimated that 18% of water consumption at this stage may be reduced.

Conventional tray and film were also ecodesigned. Increased use of plastic packaging has led to serious ecological problems due to their total non-biodegradability (Siracusa et al., 2008). Specifically food packaging materials are often contaminated by foodstuff and organic substances, making recycling impracticable and economically not convenient most of the times. As a consequence thousands of plastic materials from food items are either landfilled or incinerated every year, increasing the problem of municipal waste collection and disposal (Kirwan and Strawbrigde, 2003). As a proposal to face this issue, a biodegradable packaging for the chicken item was studied, based on polylactic acid and ethylene vinyl alcohol materials.

Finally, another major stage investigated through the project was product delivery phase, leading to a series of improvement actions identified. Significant reductions related to electrical and diesel consumption were estimated around 10-15% at this phase by means of improved refrigerated storage, distribution routes optimisation and reverse logistics solutions.

3.3. Compared LCA results based on potential improvements

Significant impact reductions were achieved through the improvements identified at different stages of the life cycle, as can be observed in Table 1. In the feed production stage, freshwater eutrophication and global warming effect decreased 11% and 6% respectively by replacing conventional ingredients with tomato by-products in the poultry diet, avoiding the import of grain from long distance.

Water consumption and wastewater generation were minimised about 16% in the slaughterhouse and meat processing stage through recycling and re-use alternatives, by applying innovatives techniques such as pulsed light decontamination.

Conventional plastic tray and film were replaced with biodegradable materials avoiding up to 20% of greenhouse gases emissions associated to life cycle packaging. However biodegradable materials showed an increased impact in freshwater eutrophication linked to the crop cultivation stage required to obtain raw materials for bioplastic production.

Environmental impacts during product delivery stage were also reduced by improving different aspects along the supply chain, among others, implementing modularity at refrigerated storage spaces, optimising delivery routes and promoting eco-driving lessons. Estimated savings in electrical and diesel consumption at this phase lead to a decrease between 13-15% in all the impact categories.

In terms of the whole life cycle of the analysed chicken product combined improvements have resulted in significant reductions mainly in three categories: climate change (-6.7%), freshwater eutrophication (-5.3%) and water depletion (-11.9%).

Table 3. Environmental results per FU for the considered improvement options.

	Poultry Feed		**Slaughtering**		**Packaging**		**Product delivery**		**Combined scenario**	
	Value	**Red.(%)**	**Value**	**Red.(%)**	**Value**	**Red.(%)**	**Value**	**Red.(%)**	**Value**	**Red.(%)**
CC	1.53	-4.8%	0.16	-2.5%	0.14	-20.8%	0.17	-13.1%	2.70	-6.7%
OZ	0.12	-2.2%	0.01	-2.3%	0.01	87.8%	0.03	-15.0%	0.19	-1.5%
AC	11.20	-1.3%	0.85	-3.7%	0.34	-31.6%	0.83	-13.0%	74.45	-0.5%
FE	0.46	-9.3%	0.08	-8.4%	0.04	78.9%	5.3E-04	-13.0%	1.09	-5.3%
WD	4.39	-3.3%	5.73	-15.9%	0.75	-54.4%	0.02	-13.1%	17.88	-11.9%
CED	13.61	-1.8%	2.50	-3.1%	1.89	30.1%	2.32	-13.0%	27.99	-1.4%

CC = Climate change; Units: kg CO_2 eq., OZ = Ozone depletion; Units: mg CFC eq., AC = Terrestrial acidification; Units: g SO_2 eq., FE = Freshwater eutrophication; Units g P eq., WD = Water depletion; Units: l, CED = Cumulative energy demand; Units: MJ

4. Conclusion

Through the ECOALIM project, LCA methodology has been successfully applied in order to identify critical stages and operations along the life cycle of a food product from an environmental point of view, but also for comparative analysis between different technologies and potential improvement options. LCA proved to be a useful tool directly involved in the decision making process when minimising the environmental impact associated to food chains, and additionally a considerable option to promote competitiveness, innovation and sustainability through the eco-design of food products.

According to the results of the present study the strong contribution of the poultry feed production stage on the environmental impact of a chicken product has been pointed out, as previously highlighted by other authors (Alvarenga, 2012) (Leinonen et al., 2012) (Katajajuuri, 2009). Nevertheless, the relevance of other phases of the life cycle should not be underestimated. Environmental improvements at every stage can lead to significant global reductions due to the high volume of consumption of the analysed product.

Acknowledgements: This work was performed with the financial support of the Spanish Ministry of Science and Innovation. We want to thank the partners involved in the project ECOALIM for their help with data collection

5. References

Althaus H.-J., Dinkel F., Stettler C. and F. W. (2007) Life Cycle Inventories of Renewable Materials. Final report ecoinvent Data v2.0 No. 21. Swiss Centre for Life Cycle Inventories, Dübendorf, retrieved from: www.ecoinvent.org.

Alvarenga, R. A. F. D., da Silva Júnior, V. P., & Soares, S. R. (2012). Comparison of the ecological footprint and a life cycle impact assessment method for a case study on Brazilian broiler feed production. Journal of Cleaner Production, 28, 25-32. Elsevier Ltd.

de Vries, M., and I. J. M. de Boer. 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livest. Sci. 28:1–11

EPER (2007). European pollutant emissions registry. Eper-España. http://www.eper-es.com

EMEP-CORINAIR (2007). Emission Inventory Guidebook. European Environment Agency. Copenhagen.

Goedkoop M.J., Heijungs R, Huijbregts M., De Schryver A.;Struijs J.,; Van Zelm R. (2009) ReCiPe 2008, A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level; First edition Report I: Characterisation;

Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, van Oers L, Wegener A, Suh S, Udo de Haes HA (2001) Life cycle assessment – An operational guide to the ISO standards. Centre of Environmental Science, Leiden, The Netherlands

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., Sutter, J. 2007: Life Cycle Inventories of Bioenergy. ecoinvent report No. 17, Swiss Centre for Life Cycle Inventories, Dübendorf, CH.

Jungbluth, N., Frischknecht, R. (2004) Implementation of Life Cycle Impact Assessment Methods. Ecoinvent Report No. 3;

Katajajuuri, J.-M., Grönroos, J. & Usva, K. 2009. Environmental impacts and related improvement options of supply chain of chicken meat. Proceedings of the 6th International Conference on Life Cycle Assessment in the Agri-Food Sector. Zurich, Switzerland, Nemecek, T. & Gaillard, G. (eds.), Agroscope Reckenholz-Tänikon Research Station ART, June 2009. pp. 370-380. ISBN 978-3-905733-10-5.

Kirwan, M. J., & Strawbridge, J.W. (2003). Plastics in food packaging. Food Packaging Technology 174e240.

Leinonen, I., A. G. Williams, J. Wiseman, J. Guy, and I. Kyriazakis. 2012. Predicting the environmental impacts of chicken systems in the United Kingdom through a life cycle assessment: Broiler production systems. Poult. Sci. 91:8–25.

Nemecek T., Heil A., Huguenin O., Meier S., Erzinger S., Blaser S., Dux. D. and Zimmermann A. (2007) Life Cycle Inventories of Agricultural Production Systems. ecoinvent report No. 15, v2.0. Agroscope FAL Reckenholz and FAT Taenikon, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, retrieved from: www.ecoinvent.org.

Pelletier, N. 2008. Environmental performance in the US broiler poultry sector: Life cycle energy use and greenhouse gas, ozone depleting, acidifying and eutrophying emissions. Agric. Syst. 98:67–73

Siracusa, V., Rocculi, P., Romani, S., & Rosa, M. (2008). Biodegradable polymers for food packaging: a review. Trends in Food Science & Technology, 19(12), 634-643. Elsevier Ltd.

Tassou, S. a., De-Lille, G., & Ge, Y. T. (2009). Food transport refrigeration – Approaches to reduce energy consumption and environmental impacts of road transport. Applied Thermal Engineering, 29(8-9), 1467-1477.

Williams, A. G., E. Audsley, and D. L. Sandars. 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. http://www. agrilca.org.

Zufia, J.; Pardo.G. (2011). Market research to assess environmental impacts of consumer stage in food LCA. Life Cycle Management Conference. Berlín.

Seguin, F; van der Werf, H; Bouvarel, I.; Pottiez, E (2011). Environmental analysis of organic broiler production in France and improvement options. Life Cycle Management Conference. Berlin.

Product-Oriented Environmental Management System (POEMS) in the agri-food sector: main results of the EMAF project

Roberta Salomone[1*], Maria Teresa Clasadonte[2], Maria Proto[3], Andrea Raggi[4], Ioannis Arzoumanidis[4], Giuseppe Ioppolo[1], Agata Lo Giudice[2], Ornella Malandrino[3], Agata Matarazzo[2], Luigia Petti[4], Giuseppe Saija[1], Stefania Supino[3], Alessandra Zamagni[4]

[1] University of Messina, Messina, Italy
[2] University of Catania, Catania, Italy
[3] University of Salerno, Salerno, Italy
[4] University G. d'Annunzio, Pescara, Italy
[*] Corresponding author. E-mail: *roberta.salomone@unime.it*

ABSTRACT

In this paper the main results of the Eco-Management for Food Project (co-funded by the Italian Ministry of Education, University and Research - PRIN No. 2008TXFBYT) are presented. Within the EMAF project, a Product-Oriented Environmental Management System (POEMS) framework that is specifically tailored for the agri-food industry, is being designed and implemented. It has a modular structure as it is made up of a set of complementary tools: an Integrated Quality and Environmental Management System; a simplified Life Cycle Assessment methodology; guidelines for product environmental communication. Finally, this paper describes the main results of the POEMS framework implementation to different pilot food companies, in order to verify the effective functioning and to highlight the strong and weak points of the POEMS model and of its individual fundamental elements.

Keywords: POEMS, Integrated Management Systems, Simplified LCA, Environmental Product Labels, Agri-Food Supply Chain

1. Introduction

Product-Oriented Environmental Management System (POEMS) is a new framework designed to bring together traditional environmental management systems and tools oriented towards environmental product performances. It is *"a systematic approach to organizing a firm in such a way that improving the environmental performance of its products across their product life cycles becomes an integrated part of operations and strategy"* (de Bakker et al., 2002). Despite there still being no standard reference and few studies available in literature – mainly in manufacturing industries and only one first attempt (Ardente et al., 2006) in the agri-food sector – a growing number of organisations are experiencing the need to integrate environmental management systems standards with those addressed to the environmental evaluation of products, shifting attention from system/process to product/service.

In this context, the definition of a POEMS framework specifically tailored for the agri-food industry is the core target of the Eco-Management for Food (EMAF) Project (co-funded by the Italian Ministry of Education, University and Research - PRIN 2008TXFBY), that sets out to define, test and disseminate innovative environmental management tools in order to improve the sustainability and competitiveness of agri-food companies. The choice of this particular sector is mainly due to its economic importance in the European Union, as well as the considerable amount of natural resources used and environmental pollutants released by this industry; indeed, the Environmental Impact of Products - EIPRO study (Tukker et al., 2006), conducted by the European Commission, showed that among the products consumed in Europe, food and beverages are the ones which are associated with major environmental impacts, in a life cycle perspective.

2. Methods

Within the EMAF project, the POEMS framework has a modular structure resulting from the integration of complementary environmental management tools: the underlying basis is an Integrated Quality and Environmental Management System (ISO 9001 and ISO 14001/EMAS), while the product orientation is provided by a Simplified Life Cycle Assessment and a suitable environmental product label or declaration chosen following Guidelines for the Environmental Product Communication.

The innovative character of the EMAF project has a double dimension connected to the fact that each environmental management tool included in this study is developed in its methodological structure and then applied in pilot firms, both with a single and an integrated approach, offering to agri-food organisations a "modular" format that refers to each tool separately and to the POEMS in general. In this way, whatever the starting point of the firms is and whatever their targets are, they will find an answer and a strategy via which they may formulate their own route to eco-compatibility. In the following a synthesis of the single environmental management tools of which the POEMS model is made up is presented with a brief description of their methodological structure and the main results of their implementation to different pilot food companies, in order to verify the effective functioning and to highlight the strong and weak points of the POEMS model and of its individual fundamental elements.

3. Results and discussion

3.1. Integrated Quality and Environmental Management System: the structural basis of POEMS

An Integrated Management System (IMS) is based on the combination of separate Management Systems (MSs) in order to plan, realise, control, audit and improve systematically a wide array of company performance, related principally to quality, environment, health and safety. During the last few decades, researchers have discussed IMS in a multitude of theoretical and empirical studies, focusing on different perspectives and addressing important aspects concerning possible strategies, methodologies and degree of the integration process (Salomone, 2008; Bernardo et al., 2009). Strategies refer to the selection and implementation sequence of sub-systems, while methodologies cover the implementation phases and steps; the degree concerns the level of integration that the organisation intends to achieve (Karapetrovic, 2004; Jorgensen et al., 2008).

The multitude of theoretical approaches found in the existing literature on the integration of MSs, lead to the conclusion that there is not a “one size fits all” methodology on which to build an integrated structure. Every integration process depends on the specific characteristics of the organisations involved, particularly in reference to dimension, number of pre-existent MSs, and sector features.

Within the EMAF project it was of primary importance to design a methodology for IMS implementation - which forms the backbone of POEMS -, covering Quality Management System (QMS) and Environmental Management System (EMS), focused on the agri-food sector, and testing its application in a specific pilot firm.

The proposed IMS model has to be seen as a flow of activities, schematized for simplicity in three progressive but different steps of integration, on the basis of the compatibility and complementarity between the requirements of the standards. The IMS model proposed is characterised by an innovative operational value because its approach focuses specifically on the agri-food sector. It has been applied to a pilot company and permits its continuous adaptation in accordance to the specific needs of SMEs.

The model proposes a multi-step progressive approach related to the following phases. The “first level” is “strategic”, identifying the principles, objectives, policies and values useful to the continuous improvement of quality and environmental performance; the “second level” involves aspects linked to the “systemic implementation” of the IMS, by a synergic management of resources and a full analysis of the results achieved in each of the areas considered; the “third level”, however, has a “unifying” nature, aiming at the complete integration and synergies among all the organisational managerial and cultural aspects.

During the integration process in the pilot company some strong points, clustering in internal and external, have emerged (Zeng, et al., 2007). The former are: a focus on a holistic approach and underpinning relationships; the harmonisation of capabilities linked to the early use of formalised MSs; a reduction in unnecessary documents and bureaucracy; improvements in organisational efficiency and effectiveness. The external strong points are principally related to: the worldwide growing diffusion of multiple MSs in the agri-food sector and the need to enhance their synergies; the increasing use of best business practices due to strong competition on the global market; widespread adoption of tools for continuous improvement and benchmarking. The internal weak points, which arose during the implementation process, are: the need for an aptitude to change among employees and management; the difficulties in re-allocation of roles, responsibilities and skills; finally, the need of resources for training, knowledge sharing and dissemination. The lack of market information on IMS approaches pursued in managerial and organisational processes and difficulties in fostering IMS adoption as a tool for creating value on the market are the most important barriers which emerged in the external context.

3.2. Simplified Life Cycle Assessment: the environmental assessment of agri-food products

Life Cycle Assessment (LCA) has been increasingly used to identify and assess the environmental impacts of a variety of goods and services. In the framework of the EMAF project, existing LCA studies and review papers on food supply chains were analysed to report the current state-of-the-art and identify critical issues. More in detail, the purpose of such a review was to identify: a) the main methodological issues in the food sector and how they have been dealt with, b) whether there could be a tendency for some environmental impacts to be more affected than others, and c) whether there were specific stages in products’ life cycles that seem to be more impacting than the others. All the above purposes were related to identifying the information necessary for selecting a simplified tool that could be suitable to be implemented in this field.

Subsequently, a literature review of simplified LCA approaches followed resulting in several papers focusing on a variety of products/sectors (a few ones related directly or indirectly to the food sector). Most papers highlighted that the need to simplify an LCA lies within the time and cost parameters (especially for SMEs) of carrying out a full LCA. In addition, simplified methods were recognised to be useful in the early product design phases, when limited information is available. Finally, these methods were recognised to be helpful in green procurement, for example for identifying the minimum techical specifications based on environmental characteristics. Based on the previous outcomes, a set of criteria were identified for the selection of the most suitable simplified tool among those reviewed: a) ISO-compliance, b) broad focus (a number of impact categories to be considered, not just one, like, e.g. in Carbon Footprint), c) user-friendly interface, d) limited data requirement or adaptability to existing databases, e) relevance to life cycle steps identified in our food LCA review, and f) ease of integration with EPD, POEMS and other communication tools.

Subsequently, a number of decision-making tools for assisting the selection were identified, namely: i) Delphi Method, ii) Analytical Hierarchy Process, iii) Multi-Criteria Decision Making (MCDM) and iv) Rough Sets Theory. It was finally decided to use the MCDM method, after having consulted a number of experts in the domain, according to which a set of weights were assigned to the above criteria by a group of experts. The tools found in the literature are being evaluated by some experts and tool developers themselves through the application of three of the methodologies of the Multi-Attribute Utility Theory family, namely SMART (Barron and Barrett, 1996), Mszros and Rapsk (Meszaros and Rapcsak, 1996) and Entropy Optimisation (Lofti and Fallahnejad, 2010). The results of the simplified LCA will be compared to those of a full LCA. In the meantime, the collection of data (necessary for both types of LCA) was performed at two small winemaking firms in the region of Abruzzo, in Italy. The data collection and the early steps of the tool application demonstrated that performing a simplified LCA may require limited time and resources. Furthermore, simplified tools have clear and easy to understand calculation and visualisation methods and are considered to be suitable for effective communication of the environmental performance of products and services. The user friendliness along with the Life Cycle Thinking orientation are characteristics that simplified LCA tools normally offer, as well. When it comes to opportunities that can make such tools more easily adopted, those can include a proactive approach as regards the strategic management of the environmental variable, a sensitivity of the management to environmental issues and an interest for eco-labelling initiatives on the side of the market.

On the other hand, such tools are characterised by their difficulty in incorporating the methodological differences across firms and sectors. Moreover, a reduced scope and an increased subjectivity are issues that can be considered as weaknesses of simplified LCA tools. As far as external threats are concerned, they are mostly connected to a general lack of environmental awareness by the firms combined with a central focus on short-term problems, mainly due to the pressure by the market. In addition, the fact that a general rule for Small and Medium-sized Enterprises (SMEs) being that environmental management tools are not perceived as an opportunity has to be taken into consideration. In parallel, a tendency for lack of time and/or willingness of the technical staff and the management for data collection was identified. Finally, the fact that environmental issues are often perceived as constraints and source of additional and often unknown (or hidden) costs has to be noted, as well (Masoni et al., 2004). The next steps of the research group include the finalisation of both the full and the simplified LCA implementations for various types of wine in the two firms involved and a comparison of their results in order to assess the robustness of the selected simplified tool.

3.3. Guidelines for product environmental communication: the market orientation

The agri-food production based on more sustainable processes enhances the importance of the relationship of trust between producers and consumers and requires communication tools by which provides useful information, related to the respect of the environmental resources, to consumers. Specific environmental tools may respond to these needs, tools that can orient agri-food firms toward more sustainable production processes and that can attribute to products an objective, recognizable and marketable environmental value (Lo Giudice and Clasadonte, 2010).

Due to the great amount of different environmental communication and labelling systems, a framework of guidelines has been proposed for supporting firms in the choice of the most appropriate environmental communication system, with regards to their productive peculiarities, the environmental impacts, the territorial characteristics and the type of the target market.

The main characteristics of the guidelines can be synthesized as follows: consistency with the provisions of series ISO 14020 and ISO 14063 standards; general character, i.e. all organisations can apply them regardless of size, sector, location; transparency and completeness, to be easily used as a tool for decision making.

The framework (characterised by a clear, credible and transparent language) has a structure based on iterative procedural steps suitable for different kinds of stakeholders whose degree of involvement in the communication process is taken into account.

The proposed guidelines provide the following index:

- Introduction (state of the art, existing labels);
- Principles of environmental communication (terms and definitions, reference standards);
- Goal and scope (voluntary tools of certification, assessment of the business needs, system boundaries, involved stakeholders, markets of references);
- Environmental communication policy (planning of the environmental communication activity, identification of the business tools);
- Environmental communication strategy (involved business resources);
- Measurement of environmental impacts (questionnaires, input-output flow analysis, flowcharts iterative models, decision support systems, identification of indicators of organisations performance, best available techniques, prevention strategies, more appropriate labeling identification);
- Reporting (documentation, logos);
- Environmental Communication (revision policies - audit, cost analysis, potential benefits, description of chosen tool, potential integration with other enterprise management tools, identification of target-audience, final recommendations).

The guidelines have been applied to a pilot company operating in the pasta chain located in eastern Sicily to test their effectiveness and highlight the strong and weak points. The flowcharts and iterative steps of the approach proposed allowed to analyse the firms characteristics and the features of the reference chain, guiding the management to adopt one of the most popular used tool of environmental communication for products: the Environmental Product Declaration (EPD). To this scope, starting with the Product Category Rules (PCR) existing for the area analysed, the assessment of environmental impacts of the production process of the pilot company was made using the methodology of Life Cycle Assessment (LCA), which highlighted the agricultural phase as the most impactful throughout the production cycle (Lo Giudice et al., 2011).

The application of the guidelines, despite the initial lack of knowledge by the company of different communication tools and the difficult involvement of stakeholders within the sector, led to: identify the most appropriate tool for environmental assessment; increasing interest in environmental communication tools, such as trademarks and statements; increase the knowledge of suppliers, distributors and consumers about corporate environmental performance. However, it was found that there are still some open issues of fundamental importance, such as: the lack of knowledge on distributors/consumers of these tools; the limited financial resources in the hands of companies; the limited availability of PCR, in the agri-food sector, relating to the EPD system; the uncertainty on the possible application of the Ecolabel in food products.

3.4. Product-Oriented Management System: the complete framework

Pursuing the goals of EMAF project, a literature review of previous methodological and applicative studies of POEMS was performed in order to identify the most appropriate methodological solutions for the agri-food industry; the information gathered allowed us to define key issues that were then translated in the following POEMS model requirements: a) fundamental structure composed of a management system conforming to ISO 14001 or to the EMAS Regulation, integrated with ISO 9001 and other possible management systems typical of the agri-food sector; b) methodology based on the Deming Cycle, fully exploiting the iterative character of the cycle in order to pursue continuous improvement of both the methodological structure and environmental and product performance; c) product orientation ensured by the integration of a Simplified Life Cycle Assessment methodology suitable for organisations in the agri-food production chain, which can be used to evaluate different cultivation methods, production technologies and alternative materials; d) ability to transform the environmental measures taken into commercial advantages in the best possible way for the organisation, thanks to the use of guidelines that can support firms in their choice of the most suitable form of environmental message, closely linked to the product; e) simplification of certain operational aspects and reduction of “bureaucracy”; f) general character, making it applicable to any type of activity in the agri-food sector, whatever the organisation’s size, nature and position in the agri-food supply chain; g) modular structure, as it is composed of a collection of management tools that can be applied, individually or as an integration of two or more elements, on the basis of organisations’ specific requirements and of the objectives they aim to reach. The POEMS framework is synthetically illustrated in Fig. 1.

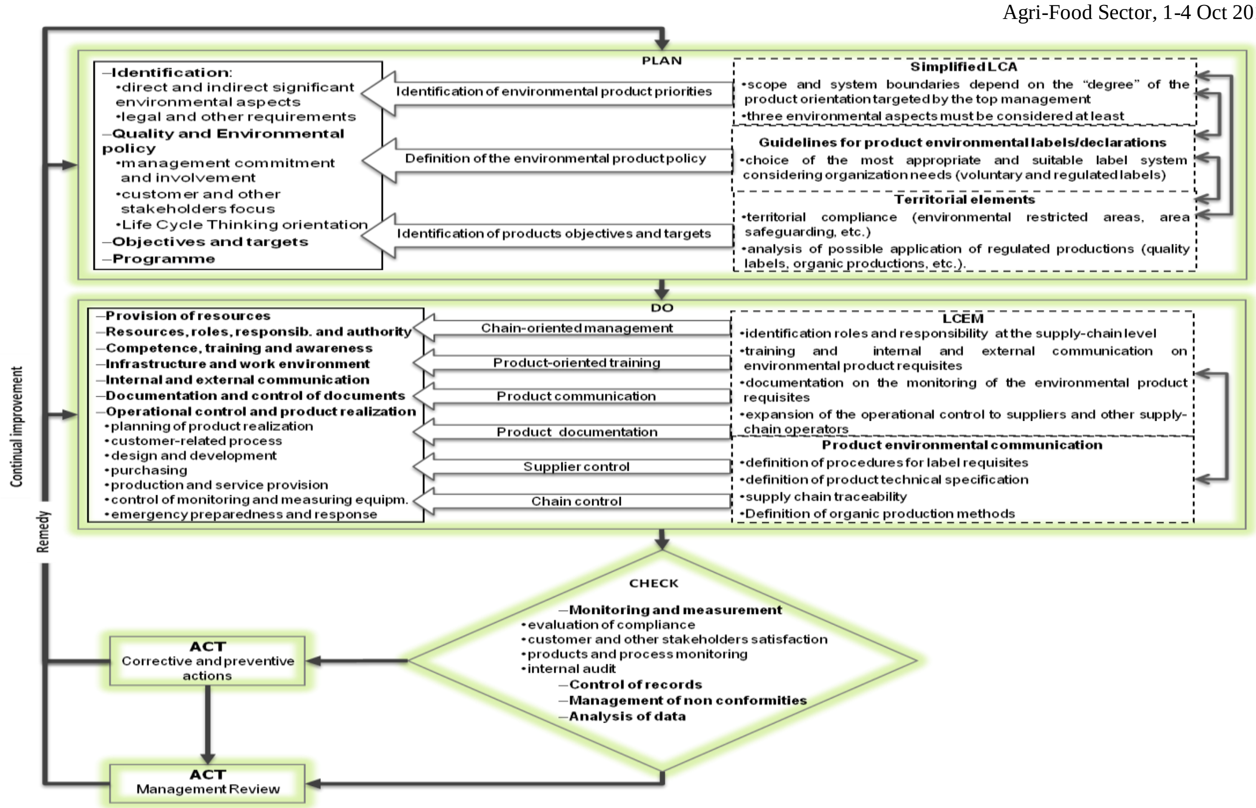


Figure 1. The POEMS framework (EMAF, 2012)

The model was developed from a traditional EMS (ISO14001 standard), integrated with a QMS (ISO 9001 and ISO 9004 standards), and is based on the PDCA cycle. It allows the collection of external customers' and other stakeholders' expectations concerning the product requirements (that is a key issue in ISO 9001 standard but not in the ISO 14001 one). In the PLAN phase, the product-orientation of IMS is guaranteed by the inclusion of a Simplified-LCA (of one or more products), and by the consideration of territorial elements (e.g. land protection; regulated quality brands; organic production) that allows the collection of data and information useful for the evaluation of improvements in the environmental performances of products. This information flow lets to change the initial environmental review so as to take into account the environmental impacts of products and the interaction with the other actors of the supply-chain. In the DO phase, the framework is completed with the inclusion of a Life Cycle Environmental Management strategy in order to improve the decision-making process by deploying a range of information useful to the supply-chain sustainability, the definition of the appropriate product documentation and the preparation of the chosen environmental product label or declaration. Finally, the framework continues with the phases of CHECK and ACT.

In order to verify the effective functioning of the POEMS model, its application in pilot companies, operating in two different agri-food supply chains, has already been started up. The two supply chains were chosen in order to involve firms operating in important Italian sectors, from an economic and/or environmental point of view, but with very different characteristics and problems: the olive oil and the roasted coffee industries. During the POEMS implementation in the pilot companies some strong and weak points have emerged; they can all be summarized in the following main issues: even if the model demonstrated the robustness of its general and iterative character and a reduction of "bureaucracy", the need for a huge quantity of data and the necessity of defining common goals and implementing joint efforts with the other actors of the supply chain, in the real practice, hinder the full implementation of a POEMS. Indeed, firms are reluctant to collect information from other actors and in the agri-food organisations a limited co-operation across the supply chain still persists. This resistance to change should be faced up with proper information and training activities in order to struggle the lack of a supply chain management perspective.

4. Conclusion

The innovative character of the POEMS model is determined by the integration approach of tools that are generally analysed as "independent" tools, while in the EMAF project they are closely related to each other. The adoption of an IMS, made up of a QMS and an EMS, represents a fundamental step in the transition from a conventional to a more sustainable business practice in agri-food organisations. At the same time, it represents only the starting point towards the complex pathway for the improvement of the global performance of agri-food products and processes, in accordance to the environmental sustainability perspective. Such a vision, indeed, requires the adoption of an array of tools, aiming at a POEMS as the final target. The other tools to be integrated are a Simplified LCA and a proper product environmental communication tool. Indeed, the product-orientated approach allowed by a Simplified LCA methodology, specifically suited for agri-food SMEs, and guidelines for supporting firms in the choice of the most appropriate environmental communication system, are deemed as highly necessary for a successful POEMS framework: they may assure the market orientation essential to counter the erroneous firm's conviction that environmental management tools are not a business opportunity. In fact, the applicative phase of EMAF project (that is still in progress) has highlighted that various important factors affecting the application of each environmental management tool are widespread in many organizaton of the agri-food sector; these factors include the lack of market information, the understanding and awareness of environmental issues, the difficulty to consider these tools as instruments for creating value in the market, and the limited co-operation across supply chains. This means that cultural and structural changes and continual improvement are the imperatives that should be properly managed in order to directly connect POEMS and its single components with the challenges of transforming the sustained efforts into effective business opportunities.

5. References

Ardente, F., Beccali, G., Cellura, M., Marvuglia, A., 2006. POEMS: a case study of a winemaking firm in the South of Italy. Environ. Manage.. 38, 350-364.

Barron, F., & Barrett, B., 1996. The efficacy of SMARTER – Simple Multi-Attribute Rating Technique Extended to Ranking. Acta Psychologica 93, 23-36.

Bernardo, M., Casadesus, M., Karapetrovic, S., Heras, I., 2009. How integrated are environmental, quality and other standardized management systems? An empirical study. J. Clean. Prod.17 (8), 742-750.

de Bakker, F.G.A., Fischer, O.A.M., Brack, A.J.P., 2002. Organizing product-oriented environmental management from a firm's perspective. J. Clean. Prod. 10, 455-464.

EMAF, Eco-Management for Food Project, 2012. Available from http://ww2.unime.it/emaf/

Jorgensen, TH, Remmen, A., Mellado, M.D., 2006. Integrated Management Systems – three different levels of integration. J. Clean. Prod.14 (18), 713-722.

Karapetrovic, S., Casadesús, M., 2009. Implementing environmental with other standardized management systems: scope, sequence, time and integration. J. Clean. Prod.17 (5), 533-540.

Lo Giudice, A., Clasadonte, M. T, Matarazzo, A, 2011. LCI preliminary results in the Sicilian durum wheat pasta chain production. J. of Commodity Science. Technology and Quality 50, 65-79.

Lo Giudice, A., Clasadonte, M. T., 2010. The EPD for the agro-food chain products. Calitatea-acces la success: facing the challenges of the future – excellence in business and commodity sciences 116, 472-480.

Lofti, F., Fallahnejad, R., 2010. Imprecise shannons entropy and multi attribute desicion making. Entropy 12(1).

Masoni, P., Sára, B. Scimìa, E. & Raggi, A., 2004. VerdEE – A tool for adoption of Life Cycle Assessment in small- and medium-sized enterprises in Italy, Progress in Industrial Ecology 1 (1-3), 203-228.

Meszaros, C., Rapcsak, T., 1996. On sensitivity analysis for a class of decision systems. Decis.Support Syst.16, 231-240.

Salomone, R., 2008. Integrated management systems: experiences in Italian organisations. J. Clean. Prod. 16 (16), 1786-1806.

Tukker, A., Huppes, G., Guinée, J., Heijungs, R., de Koning, A., van Oers, L., Suh, S., Geerken, T., Van Holderbeke, M., Jansen, B., Nielsen, P., 2006. Environmental Impact of Products (EIPRO). Analysis of the life-cycle environmental impacts related to the final consumption of the EU-25. Summary of the final report. European Commission: DG Environment e DG JRC.

Zeng, S., Shi, J., Lou, G., 2007. A synergetic model for implementing an integrated management system: an empirical study in China. J. Clean. Prod.15 (18), 1760-1767.

An LCA support tool for management of protected horticultural systems

Marta Torrellas[1,*], Assumpció Antón[1], Marc Ruijs[2], Nieves García Victoria[2], Juan Ignacio Montero[1]

[1] IRTA, Ctra. de Cabrils, km 2. 08348 Cabrils, Barcelona, Spain
[2] Wageningen UR Greenhouse Horticulture, PO Box 644, 6700 AP Wageningen, The Netherlands
[*] Corresponding author. E-mail: *marta.torrellas@irta.cat*

ABSTRACT

Life cycle assessment (LCA) is a useful tool for environmental management, but is complex for non-specialists. The objective of this study was to develop an easy-to-use support tool to help stakeholders in horticulture with their decision making as a way of mitigating the environmental impact of protected crops. Using a simplified LCA, three spreadsheets were provided and, as a result, two types of crops (tomato and rose) and two types of structures (plastic multi-tunnel and Venlo glass greenhouse) were evaluated. Results are expressed by functional unit in six midpoint impact categories. With the analysis of three case studies, results showed that this environmental calculator is a useful tool to determine major burdens in greenhouse production systems and evaluate the efficient use of inputs. Simplification of the tool created some difficulties that may be improved with further research, such as the selection of appropriate data sets and characterisation models.

Keywords: simplified LCA, decision support, environmental impacts, system, greenhouse structure

1. Introduction

Life cycle assessment (LCA) is an environmental management tool that can be used for multiple applications, such as evaluating the environmental impacts of a product or service, comparison of environmental performance of different products, ecodesign, ecolabelling and environmental product declarations (EPD). LCA methodology is improving continuously with new characterisation models, databases and guidelines to increase robustness and areas of use. Thus, LCA can be a complex tool for non-specialists and, as a result, its application is usually restricted to professionals in the area such as researchers, consultants and other experts. However, one of the goals of scientific community is to extend research objectives from pure analysis towards application in decision making and the context of policy. With this aim in mind, an effort was made to achieve integration between simulation and decision making to provide tools to simplify a complex system.

Over time, societal concern about environmental problems has increased the demand for reliable information and tools to understand and mitigate environmental damage. Lately, environmental management has changed through increased requirements and decision making that simultaneously considers economic and social systems, as well as ecosystems. One of the answers to satisfy this demand has been the appearance in the market (i.e. websites) of environmental calculators developed as simplified life-cycle management tools to simulate systems and support decision making. Specific calculators are oriented to a variety of professional sectors such as ecodesign, the construction industry, energy activities and waste management. In the food industry, calculators can be found for the environmental evaluation of personal consumption habits (www. uns.ethz.ch), sustainable shopping (http://jocapqua.urv.es/en/credits.html) and industry activities such as the Carbonostics calculator (http://www.carbonostics.com/). Greenhouse gas emissions that affect global warming (carbon footprint) are usually the only factor taken into account, which gives a reduced view of the environmental problem (Finkbeiner, 2009). Recently, simplified contributions for a wide range of users have been released, such as the Cool Farm Tool, a greenhouse gas calculator for crop and livestock production at farm level (Hillier, 2011) and Musa software for water assessment of different agricultural production systems (Amores Barrero et al., 2012). Horticultural environmental calculators focus on open-field systems, but the authors are not aware of any such calculators for protected crops. The aim of this study was to develop an easy-to-use environmental tool to calculate the efficiency of inputs of protected crops and evaluate options to reduce them. The calculator is based on a simplified LCA, gives results for six environmental impact categories and is designed to help a mix of users with decision making. This paper presents the environmental tool that was applied to three production systems. The tool was developed in the context of the EUPHOROS research project and free access is available on the project website in several languages.

2. Methods

This environmental calculator was based on a simplified LCA following the principle of "as simple as possible and as complex as necessary" (Pidd, 1996) and the ISO 14040 standard (ISO-14040, 2006). The tool includes three Excel spreadsheets for three greenhouse production systems representative of current agricultural practices in Europe (Montero et al., 2011). The following scenarios were used as reference situations: a tomato crop in a multi-tunnel greenhouse under Southern European climate conditions, and a tomato crop and a rose crop in Venlo glass greenhouses under Central European climate conditions. Each spreadsheet has four sheets disclosed to users: *Instructions, Input Data, Total Results* and *Detailed Results.* Users can simulate their own production system by following a few easy steps: selection of one scenario, data entry and consultation of results.

The *Input Data* sheet consists of a questionnaire structured around different topics. A link to a fertiliser calculator is available to calculate the amount of total nutrients based on the specific fertiliser doses applied to the crop. These user data are primary data used to calculate results, along with secondary data from international databases. Datasets from Ecoinvent database 2.2 (Ecoinvent, 2010) were used for processes such as the manufacture of greenhouse components (metal, plastic and glass), substrate, pesticides, means of transport and disposal. The average European electricity production mix was the process selected to evaluate the environmental impact of electricity consumption. The LCAFoods database (Nielsen et al., 2003) was used for the manufacture of generic fertilisers N, P_2O_5 and K_2O. The questionnaire provides default data for reference production systems that can be used if users do not have specific data.

The *Total Results* and *Detailed Results* sheets display results in figures and graphs for the user's production system under study and the reference situation by functional unit and impact category. Users can compare their own results with those of the reference situation. The *Total Results* sheet shows the total contribution of the production. The *Detailed Results* sheet presents results broken down by the stage in the production system.

We used an attributional LCA and mass functional units were selected: 1 tonne of classic loose tomatoes for the tomato crops and 1000 stems for the rose crop. The system boundary was from raw material extraction to farm gate, including waste material disposal. The production system was structured in stages to facilitate calculations and the interpretation of results: greenhouse structure, auxiliary equipment, climate control system, fertilisers, pesticides and waste management (Figure 1). The processes considered for the environmental analysis included inputs and outputs in the manufacture of greenhouse components, transport of materials, material disposal and greenhouse management, i.e. water, fertilisers, pesticides, electricity consumption and energy consumption in the case of heating (Torrellas et al., 2012). For this simplified LCA and following the cut-off criteria of the ILCD Handbook (ILCD, 2010), processes with an environmental impact below 5% were omitted when they were considered not relevant to an agricultural production system. The amount of materials in the structure was calculated using formulas developed from a reference production system inventory.

Figure 1. Production system diagram, including stages and processes for a tomato crop in: (1a) a multi-tunnel greenhouse in European Southern climate conditions; (1b) a Venlo glasshouse in European Central climate conditions.

1a) 1b)

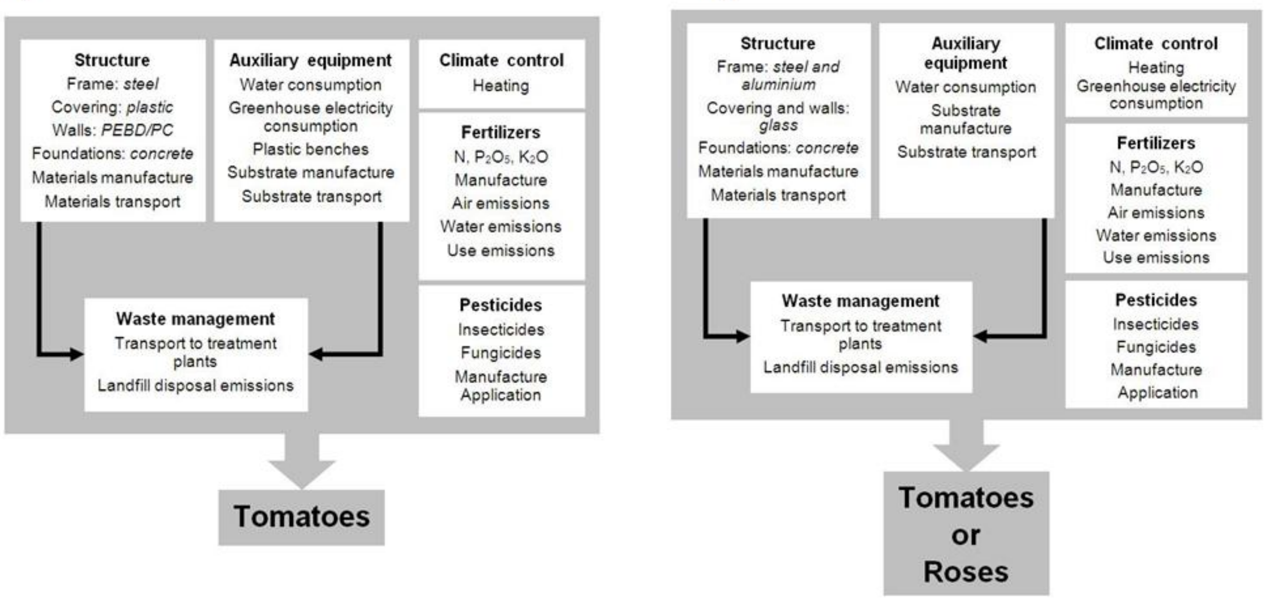


Table 1. Questionnaire summary of input data sheet, for reference and case study scenarios.

Issue	Input	Units	Tomato crop, multi-tunnel greenhouse		Tomato crop, Venlo glasshouse		Rose crop, Venlo glasshouse	
			Case study	Reference	Case study	Reference	Case study	Reference
Crop	Crop	name	Tomato	Tomato	Tomato	Tomato	Roses	Roses
	Yield	produce·m^{-2}	20.0 kg·m^{-2*}	16.5 kg·m^{-2}	56.5 kg·m^{-2}	56.5 kg·m^{-2}	289 stem·m^{-2*}	275 stem·m^{-2}
	Density	p·m^{-2}	1.23	1.23	1.25	1.25	8.5	8.5
	Stems per plant	number·p^{-1}	2	2	2	2		
	Growth period	weeks	52	52	52	52	52	52
Substrate	Type of substrate	name	Perlite	Perlite	Rockwool	Rockwool	Rockwool	Rockwool
	Substrate life span	years	3	3	1	1	1	1
	Bag volume	l	30	30	14	14	6.42	6.42
	Plants per bag	number	3	3	3	3	3	3
Structure data	Number of spans	number	10*	18	25	25	21	21
	Span width	m	6*	8	8	8	9.6	9.6
	Span length	m	60*	135	200	200	200	200
	Roof vents: total greenhouse number	number	10*	36				
	Gutter height	m	4.0*	4.5	6	6	6	6
	Ridge height	m	4.5*	5.8	6.8	6.8	6.76	6.76
	Greenhouse walls	material	LDPE*	PC	Clear glass	Clear glass	Diffuse glass*	Clear glass
	Greenhouse frame life span	years	15	15	15	15	15	15
	Greenhouse roof covering life span	years	3	3	15	15	15	15
	Greenhouse walls life span	years	3*	15	15	15	15	15
Energy consumption	Total greenhouse electricity consumption	kWh·m^{-2}	0.641	0.641	10	10	633	633
Watering	Water consumption	L·m^{-2}	475	475	795	795	902.5	902.5
	Irrigation system open/closed	type	Open	Open	Closed	Closed	Closed	Closed
Fertilisers	N	kg·m^{-2}	0.050*	0.060	0.1688	0.1688	0.1163	0.1163
	P_2O_5	kg·m^{-2}	0.035*	0.038	0.04058	0.04058	0.0276	0.0276
	K_2O	kg·m^{-2}	0.135*	0.117	0.18548	0.18548	0.128	0.128
Pesticides	Fungicides	kg·m^{-2}	0.00285	0.00285	0.0007	0.0007	0.0036	0.0036
	Insecticides	kg·m^{-2}	0.00038	0.00038	0.0003	0.0003	0.0006	0.0006
Heating	Heating	type	No heating	No heating	CHP	CHP	CHP	CHP
	Fuel	none	NO	NO	Natural gas	Natural gas	Natural gas	Natural gas
	Fuel consumption	m^3·m^{-2}	0.00	0.00	42.1*	64.7	101.7	101.7

*Data in case study differing from reference situation

The indicators and impact categories selected for the environmental assessment were: the five midpoint impact categories defined by the CML2001 method v.2.04 (Guinée et al., 2002), namely, abiotic depletion (kg Sb eq), acidification (kg SO_2 eq), eutrophication (kg PO_4^{-3}eq), global warming (kg CO_2 eq) and photochemical oxidation (kg C_2H_4 eq); one energy flow indicator (cumulative energy demand, MJ); and one inventory flow indicator (water use, m^3).

In this study, the environmental calculator was used to analyse three case studies. Each case was a type of production system that was compared with the corresponding reference production system. The main data for each situation were included in Table 1. The first case study was for tomato crop in a smaller multi-tunnel greenhouse with LDPE walls, a higher yield and lower doses of fertilisers. The second case was an energy-saving cultivation method for tomato crop in a Venlo glass greenhouse with 35% reduced heat demand and the same yield. Finally, the third case study was rose crop in a greenhouse with diffuse glass, an anti-reflective coating and a 5% higher yield.

3. Results

Potential environmental impacts are provided on the *Total Results* and *Detailed Results* sheets. Results are expressed by functional unit in six impact categories. In this study, results were for the three production systems, including the reference situation from default data and a case study for each production system (Table 2), described as follows:

3.1. Tomato production in a multi-tunnel greenhouse

Results for the reference situation showed that the structure, fertilisers and auxiliary equipment were major contributors to all impact categories. The structure made the greatest contribution to the impact categories of abiotic depletion (50%), global warming (37%), photochemical oxidation (54%) and cumulative energy demand (50%) due to the high amount of steel and plastic in the frame, covering and floor. Fertilisers were the main burden in acidification (39%), mainly because of ammonia emissions into the air during their application; and eutrophication (56%), because of nitrate emissions to water, since there was an open-loop irrigation system. The auxiliary equipment had significant contributions because of substrate and electricity consumption, between 16% and 39%, depending on the impact category. The climate control system made nil contributions, as there was no heating. Pesticides and waste management stages made contributions below 3% in the total production system.

The case study for a smaller greenhouse, lower fertiliser doses and higher yield showed contributions below the reference situation, between 13% and 37%, depending on the impact category. Structure contributions decreased between 2% and 9% as the amount of metal and plastic was reduced. Nevertheless, relative contributions of structure increased in the total production system. Reduction of fertiliser doses directly reduced the contribution to EUP, as a lower amount of lixiviates reached aquifers. A higher yield made reductions to all impact categories, as a mass functional unit was used.

3.2. Tomato production in a Venlo glass greenhouse

In this production system, a CHP was used for heating and electricity production. Energy allocation of natural gas was used to determine the impact of using natural gas to heat the greenhouse (Torrellas et al., 2012). The climate control system was the main contributor to all the impact categories, between 81% and 97% of the total in the reference situation because of the high natural gas consumption to heat the greenhouse. The structure was the second burden and made contributions between 2.0% and 10% because of metal and glass contributions. Fertilisers made contributions between 0.6% and 8.6% due to emissions during the manufacturing process and ammonia emissions into the air after these fertilisers are applied to the soil. Auxiliary equipment contributions were lower than 1.9% of the total, and those of pesticides and waste management were all around 0%.

With a reduction of 35% of natural gas consumption, climate control stage contributions decreased significantly in all impact categories between 22% and 33%. Nevertheless, the climate control system was still the main burden, with contributions between 75% and 95% of the total.

3.3. Rose production in a Venlo glass greenhouse

The climate control system was the main contributor in the reference situation and in this case study because of natural gas consumption for heating and electricity consumption for lighting. Contributions were between 98% and 99% in the reference situation and the case study. The structure made contributions below 1.1% in both production systems. In the case study with diffuse glass, structural contributions increased compared with the reference situation by between 4% to 16%, depending on the impact category, as extra electricity was needed in the anti-reflective coating process. In this situation, because of the effect of a higher yield, contributions to other impact categories decreased by 4% compared with the reference situation.

Table 2. Contributions to impact categories for: (2a) tomato crop in a multi-tunnel greenhouse, (2b) tomato crop in a Venlo glass greenhouse and (2c) rose crop in a Venlo glass greenhouse. Values are impact category indicators for the reference situation (Ref) and percentage variation versus the reference situation for each case study (CS). Results are by functional unit, tonne tomato for tomato crops and 1000 stems for rose crop.

2a)

Impact category	Total		Structure		Auxiliary equipment		Climate control system	Fertilisers		Pesticides		Waste management	
	Ref	C	Ref	C	Ref	C	Ref	Ref	C	Ref	C	Ref	C
ADP, kg Sb eq	1.3E+00	-14	6.4E-01	-9	4.7E-01	-19	0.0E+00	1.5E-01	-24	1.8E-02	-18	1.1E-02	3
AAP, kg SO_2 eq	9.4E-01	-18	3.3E-01	-6	2.2E-01	-18	0.0E+00	3.7E-01	-30	1.2E-02	-18	5.5E-03	5
EUP, kg PO_4^{-3} eq	5.0E-01	-37	1.3E-01	-2	8.3E-02	-18	0.0E+00	2.8E-01	-59	7.9E-03	-18	1.5E-03	4
GWP, kg CO_2 eq	2.0E+02	-17	7.5E+01	-6	6.2E+01	-18	0.0E+00	6.2E+01	-29	2.3E+00	-18	1.3E+00	0
POP, kg C_2H_4 eq	3.3E-02	-13	1.7E-02	-7	1.0E-02	-18	0.0E+00	4.1E-03	-27	8.7E-04	-18	2.0E-04	4
CED, MJ	3.1E+03	-14	1.6E+03	-8	1.2E+03	-18	0.0E+00	3.0E+02	-24	4.4E+01	-18	2.5E+01	3

2b)

	Total		Climate control system	
Impact categories	Ref	C	Ref	C
ADP, kg Sb eq	1.5E+01	-32	1.4E+01	-33
AAP, kg SO_2 eq	3.3E+00	-24	2.6E+00	-29
EUP, kg PO_4^{-3} eq	8.5E-01	-18	7.0E-01	-22
GWP, kg CO_2 eq	1.9E+03	-31	1.8E+03	-33
POP, kg C_2H_4 eq	2.1E-01	-29	2.0E-01	-32
CED, MJ	3.1E+04	-31	3.0E+04	-33

2c)

	Total		Structure	
Impact category	Ref	C	Ref	C
ADP, kg Sb eq	1.3E+01	-4	6.0E-02	11
AAP, kg SO_2 eq	5.9E+00	-4	5.7E-02	5
EUP, kg PO_4^{-3} eq	3.4E+00	-4	1.8E-02	16
GWP, kg CO_2 eq	1.7E+03	-4	9.6E+00	8
POP, kg C_2H_4 eq	2.6E-01	-4	2.6E-03	4
CED, MJ	3.4E+04	-4	1.4E+02	14

4. Discussion

In this section we discuss the main results achieved in the study, the methodology used and the benefits and drawbacks of the environmental calculator. Additionally, we propose several points that could be improved with future research.

The main objective of this study was achieved with the development of an easy-to-use environmental software tool to evaluate the environmental performance of protected horticultural production systems. We consider this calculator to be a useful contribution for decision making for a mix of users involved in horticulture.

LCA was appropriate to evaluate the potential impacts of protected crops objectively and transparently. The simplification of LCA in the design of this tool was done following international standards and guidelines (ISO-14040, 2006 and ILCD, 2010) to justify the inclusion or exclusion of processes. As a result, the tool includes a justified representation of the most significant processes in a greenhouse production system.

Users can easily simulate their crops and evaluate the effect on the environment of alternatives by reducing inputs and improving waste management practices. The calculator provides approximate but good quality results to compare different scenarios and follow the evolution of cleaner agricultural practices. Different damage to the environment can be studied, as six midpoint impact categories were included. The analysis of these very different structures, multi-tunnel and Venlo greenhouses, was adequately resolved by the development of specific spreadsheets and formulas to calculate their structural materials.

Simplicity is one of the main advantages of this calculator, but it was also the cause of some limitations. Many variables that affect agricultural systems could not be implemented in the calculator, such as geography, climate, soil characteristics, water availability and management practices such as conventional and organic farming. The regional variation of electricity production was not considered and consequently a more precise calculation of emissions is not possible (Torrellas, submitted). Many of these issues could be solved by including more spatial datasets or by designing open-source software. Fertiliser and pesticide results could be more detailed and pesticide toxicity could be included. Therefore, the tool should be able to update the characterisation models with more recent ones such as USEtox (Rosenbaum et al., 2008) and ReCiPe (Goedkoop et al., 2009).

The environmental tool was successfully tested by a group of agriculture support technicians. Nevertheless, the authors believe that future research into the topics discussed above could improve the tool. Of course, the opinions of users will be very helpful so the tool can be adapted to their needs.

5. Conclusion

This environmental calculator is a useful tool to evaluate the potential environmental impacts of protected crops and gives results for several impact categories. The tool is designed for a mix of users to support decision making. A simplified LCA was used to design the calculator. Nevertheless, the tool could be improved by considering other types of agricultural systems, the regional variability in water use assessment and electricity production, and consistent long-term datasets, and by making use of the latest advances in modelling and computing techniques. Users' opinions will be valuable so the tool can be adapted to their needs.

6. References

Amores Barrero, M.J., Ferrer, F., Cabot, O., Duaigües, A., Antón, A., Castells, F. Implementing decision making in water management in agriculture based on environmental indicators. 8th International Conference on LCA in the Agri-Food Sector, Rennes, France, 2-4 October 2012.

Ecoinvent, 2010. Ecoinvent Data v2.2. Swiss Centre for Life Cycle Inventories.

EUPHOROS 2008-2012. Efficient use of inputs in protected horticulture. In:KBBE-2007-12-04 SFPT.

Finkbeiner, M., 2009. Carbon footprinting – opportunities and threats. International Journal LCA 14:91-94.

Goedkoop, M., Heijungs, R., Huijbregts, J., Schryver, A., Struijs, J., Zelm, R., 2009. ReCiPe 2008, A life cyle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G.R.K., de Koning, A., et al., 2002. Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards. Kluwer, Dordrecht, The Netherlands.

Hillier, J., Water, C., Malin, D., Garcia-Suarez, T., Mila-i-Canals, Ll., Smith, P., 2011, A farm-focused calculator for emissions from crop and livestock production. Environmental Modelling & Software 26:1070-1078.

ILCD, 2010. International Reference Life Cycle Data System Handbook. European Commission. Joint Research Centre. Institute for Environment and Sustainability,

ISO-14040, 2006. Environmental management-Life cycle assessment-Principles and framework. International Organisation for Standardisation ISO. Geneva.

Montero, J.I., Antón, A., Torrellas, M., Ruijs, M., Vermeulen, P., 2011. Report on environmental and economic profile of present greenhouse production systems in Europe. European Comission. FP7-KBBE-2007-1. EUPHOROS project. http://www.euphoros.wur.nl/UK

Nielsen, P., Nielsen, A., Weidema, B., Dalgaard, R, Halberg, N., 2003. LCA food data base.

Pidd, M., Five simple principles of modelling. The 1996 Winter Simulation Conference, Coronado, California, USA, 8-11 December 1996.

Rosenbaum, R., Bachmann, T., Gold, LS., Huijbregts, M., Jolliet, O., Juraske, R., Koehler, A., Larsen, H. MacLeod, M., Margni, M., McKone, T., Payet, J., Schuhmacher, M., Meent, D., Hauschild, M. 2008. USEtox- the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. International Journal LCA.

Torrellas, M., Antón, A., Ruijs, M., García, N., Stanghellini, C., Montero, J.I., 2012. Environmental and economic assessment of protected crops in four European scenarios. Journal of Cleaner Production 28:45-55. doi:10.1016/j.jclepro.2011.11.012.

Environmental impacts of food consumption and its reduction potentials

Niels Jungbluth*, René Itten, Salome Schori

ESU-services Ltd., CH-8610 Uster, www.esu-services.ch
* Corresponding author. E-mail: *jungbluth@esu-services.ch*

ABSTRACT

Nutrition accounts for 30% of environmental impacts caused due to the final consumption of Swiss households. Therefore, it is necessary to investigate possibilities for the reduction of these impacts. We developed a general framework for such an analysis. Based on a more detailed analysis of this consumption domain, it is investigated, for which percentage environmental impacts can be reduced by a certain change in consumer behaviour. Finally, the resulting values are used to estimate the potential reduction compared to the total environmental impacts. With a combination of different measures such as less meat and luxury products, no products grown in heated greenhouses and reduction of obesity and wastages, it would be possible to reduce the environmental impacts of nutrition by two thirds and the total household consumption by more than 10%.The most promising single lifestyle change is a vegetarian diet. This general framework has also been used to investigate reduction potentials in the consumption domains of mobility and energy use of households.

Keywords: food consumption, reduction potential, environmentally friendly diet, sustainable life styles

1. Introduction

Nutrition accounts for about 30% of environmental impacts caused due to the final consumption of Swiss households (Figure 1, Jungbluth et al., 2011). This value does not even include meals consumed in restaurants, hospitals and retirement homes. It is thus the most important consumption domain from an environmental point of view. Therefore, it is necessary to investigate and understand the environmental impacts of food consumption and possibilities for the reduction of environmental impacts.

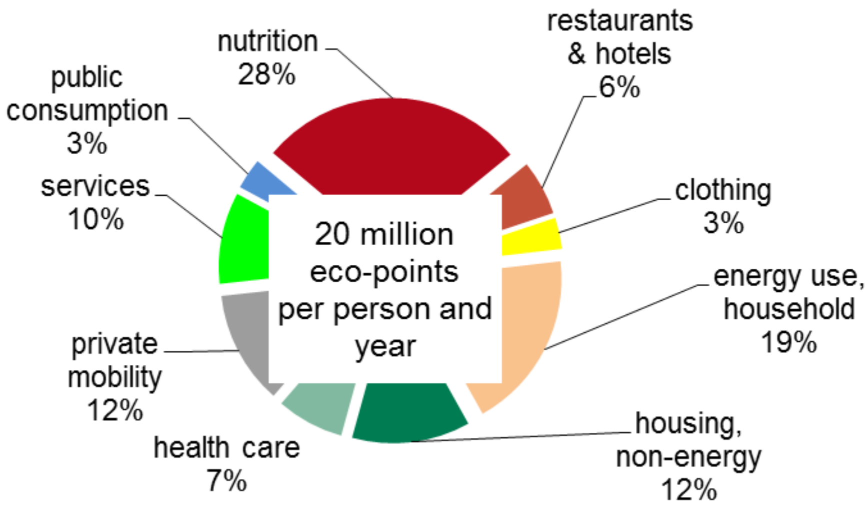


Figure 1. Share of environmental impacts of different household activities evaluated with the ecological scarcity method 2006 (Jungbluth et al., 2011).

2. Methods

Several options for reducing environmental impacts were compared within a general framework. Besides the consumption of food products also reduction potentials for impacts due to energy use in households and private mobility were investigated in order to assess potential impacts of more sustainable lifestyles (Jungbluth et al., 2012a; Jungbluth & Itten 2012).

The ecological scarcity method was used as a key indicator for the impact assessment (Frischknecht et al., 2009), but the results were also compared with respects to greenhouse gas emissions and energy use. Such a single-point indicator, summarizing all relevant environmental impacts, is seen as a necessity for the communication of LCA results to the consumers (Jungbluth et al., 2011a).

In a first step of analysis the share of the environmental impacts related to food consumption was investigated (as shown in Figure 1) with a top-down approach using an environmentally-extend Input-Output-Analysis (EE-IOA) for Switzerland (Jungbluth et al., 2011).

In a second step the consumption sector of nutrition was split up into different categories of consumed products. This calculation is based on food consumption statistics (Schweizerischer Bauernverband 2007) and life cycle assessment (LCA) data (Jungbluth et al., 2012b).

The contributions to the total impact of the different food items for the second step (bottom-up approach) are shown in Figure 2. Meat and fish account for about one quarter of the environmental impacts due to food consumption. Together with milk and eggs, animal products account for nearly half of the environmental impacts. Coffee and alcohol are the most important single products within the category of beverages. This is due to the pesticides and copper applied during the growing of the basic agricultural products. Transports, packages and processing are of minor importance for the overall environmental impacts.

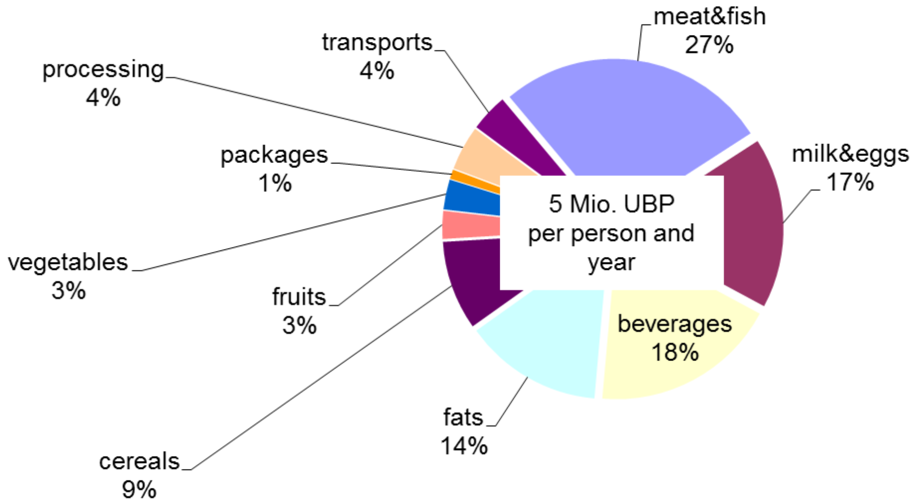


Figure 2. Importance of different product groups in total environmental impacts of nutrition evaluated with the ecological scarcity method 2006 (Jungbluth et al., 2012a; Jungbluth & Itten 2012)

The results for the top-down (share of nutrition in Figure 1) and bottom-up approaches (Figure 2) are compared in Figure 3. The overall differences are small. For some impact categories results differ because of the more general allocation schemes used in the EE-IOA.

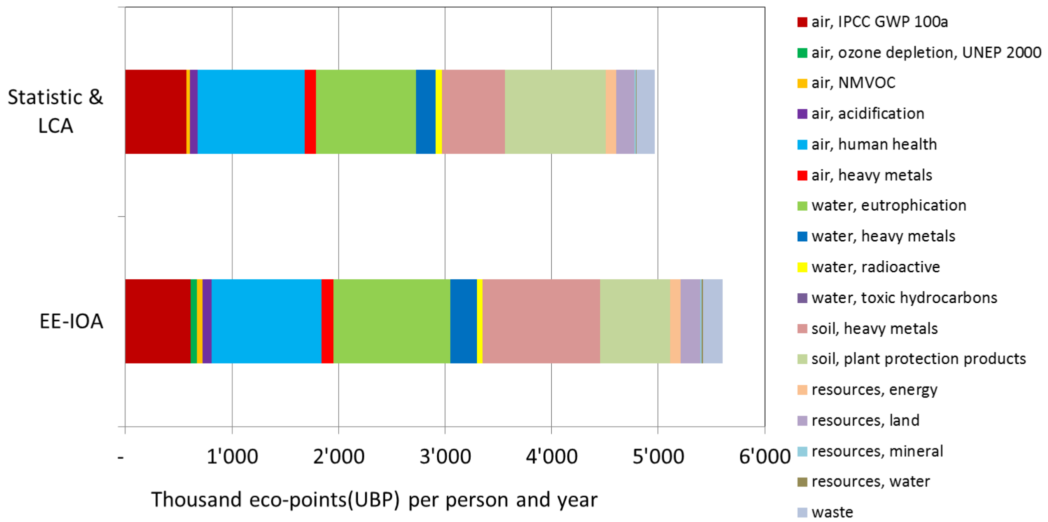


Figure 3. Comparison of top-down and bottom-up approaches according to single impact categories in the ecological scarcity methodology. Total eco-points due to nutrition according to the ecological scarcity method 2006 per person and year.

Based on the detailed analysis of this consumption domain, it was investigated, by which percentage environmental impacts can be reduced due to a certain change in consumer behaviour. In this paper we highlight and compare the reduction of total environmental impacts, if all consumers would:

1. Buy locally
2. Buy seasonally

3. Eat vegetarian
4. Buy organic food
5. Resign on luxury food (coffee, alcohol)
6. Reduce food wastes
7. Reduce obesity to normal weight
8. Combine different changes in a healthy and environmentally friendly diet

Several assumptions were necessary in order to model these scenarios for a potentially sustainable diet.

Ad 1: Buying locally should reduce the environmental impacts due to transportation. However, some restrictions have to be considered. Switzerland is only 50% self-sufficient with respects to food production; the rest has to be imported as long as consumption patterns do not change. Buying exclusively locally would only be an option for all consumers if meat consumption is reduced considerably in parallel (Würtenberger 2003; BWL 2011). Thus, here we assume only that air-transported products are avoided. It was not possible to model the change of environmental impacts due to the variation of production patterns in different countries including a potentially increased share of greenhouse products which might be bought as regional products.

Ad 2: The main aim of buying seasonal fruits and vegetables is reducing environmental impacts due to the production in heated greenhouses. In the calculation this was considered by reducing the amount of fruits and vegetables produced in heated greenhouses by 90%. Not considered is the possible reduction of transport distances if only seasonal products from the region are bought.

Ad 3: A vegetarian diet includes several alimentary changes in order to replace meat products with milk and eggs and other foodstuffs. The assumption that environmental impacts can be reduced by about 30% is based on a literature review (Faist 2000; Jungbluth 2000; Kramer 2000; Leuenberger & Jungbluth 2009; Seemüller 2001; Taylor 2000; Uitdenbogerd et al., 1998).

Ad 4: In the scenario for organic food it is assumed that all products are produced in organic agriculture. For most of the food products we had organic datasets for the calculation at our disposal (Jungbluth et al., 2012b). For some imported food products, e.g. rice, there was no LCI data on organic production available and thus no change has been considered. Furthermore it is considered in the calculation that the Swiss regulations for most organic labels prohibit the production in heated greenhouses and aircraft transport.

Ad 5: Luxury food (sometimes also called stimulants) is defined as food products which are not necessary from a nutritional point of view or which might even be unhealthy. Here we assume that Swiss consumers cease drinking alcohol and coffee. Further issues would be the reduction of sugar containing products such as cake and chocolate and the reduction of fatty snacks. This has not yet been considered in the modelling.

Ad 6: Consumers also throw away food which would have been perfectly fit to eat, e.g. because they buy or cook too much. About 15% of the food products produced undergo this fate (Gustavsson et al., 2011). Here we assume that the consumers don't waste any food. However the wastage in other parts of the life cycle was not altered for this calculation.

Ad 7: Obesity is a serious health problem in many wealthy countries. In 2007 about 37% of the Swiss adults had a body mass index (BMI) higher than 25[2]. Here we assume that food consumption is reduced by all consumers to a level that they do not reach a BMI of over 25. This would lead to a reduction of the average body weight of about 3.7 kg (or 10 kg for overweight people). The calculations for the reduction of food consumed are based on a conference paper (Cordella et al., 2009). Due to lack of data, a general reduction of food consumption has been assumed, not considering that mainly soft drinks, fat and sugar might be responsible for obesity (Zwick & Müller 2012).

Ad 8: For the last scenario an environmentally friendly and healthy diet is assumed. Here meat consumption is reduced to about 2 portions a week. This corresponds to the amount recommended by health specialists. Furthermore different options mentioned before are combined in order to assess a realistic scenario that can be followed by all consumers. The assumptions are based on the previous assessments and a review of relevant literature (Carlsson-Kanyama et al., 2003; Fazeni 2011; Grießhammer et al., 2010; Jungbluth 2000; Kramer 2000; Meier & Christen 2012). This option is also promoted by nutritionists (von Koerber et al., 1999).

A ninth interesting option for impact reduction would be a smoking stop. (Tabaco products also fall in the consumption sector nutrition.) Due to lack of data this scenario could only be investigated qualitatively.

The approach taken in order to assess the reduction potentials is explained here with an example for option 4, which assumes the purchase of organically produced food items (Figure 4). The latter would reduce the total impacts of food consumption by about 15%. The detailed analysis with the ecological scarcity

2 http://www.bag.admin.ch/themen/ernaehrung_bewegung/05207/05218/05232/index.html?lang=de, 12.9.2011

method shows that impacts of organic products are considerably lower with regard to the use of plant protection products. On the other side there are higher impacts due to heavy metal emissions to soil in organic agriculture which is mainly due to the use of copper as a plant protection product. For many other impact categories the average impacts according to the food basket are comparable.

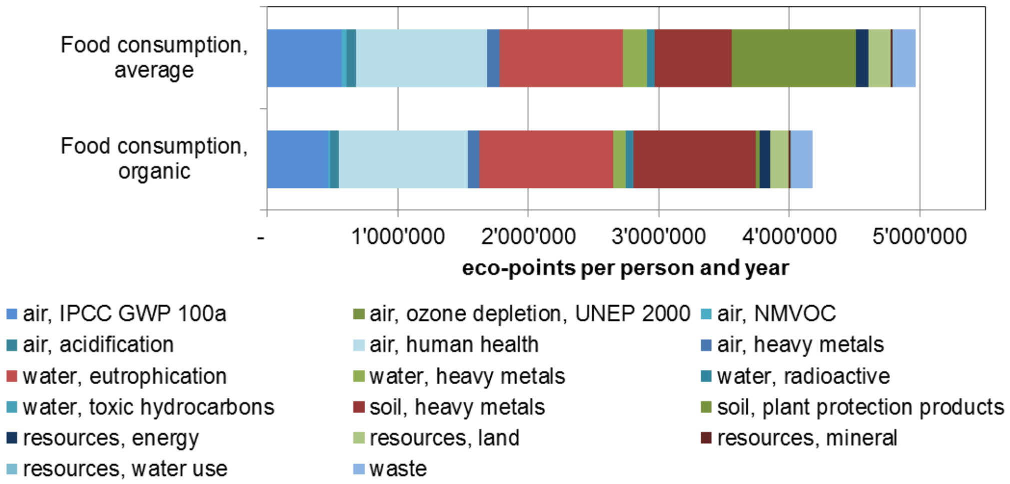


Figure 4. Comparison of the average diet with a diet based on organically grown products with the ecological scarcity method 2006 (Jungbluth et al., 2012a; Jungbluth & Itten 2012)

3. Results

The reduction potentials for the different scenarios described before are shown in Figure 5. The highest potential reduction was calculated for a combination of different measures. Within the healthy and environmentally friendly diet, it is assumed that meat consumption is reduced to two portions of meat a week instead of six. Furthermore, air-transported products are avoided and only seasonal fruits and vegetables are bought. These measures lead to a reduction of the environmental impacts of domestic nutrition by two thirds and total household consumption by more than 10%. The most promising single change in lifestyle is a vegetarian diet. On the other side a change to merely a regional or seasonal choice of products does not show a high potential for reducing the total environmental impacts. The choice for seasonal products is only relevant for fruits and vegetables, which make up a small share of the total environmental impacts. Buying locally is a restricted option in Switzerland due to the insufficient production capacities within the country.

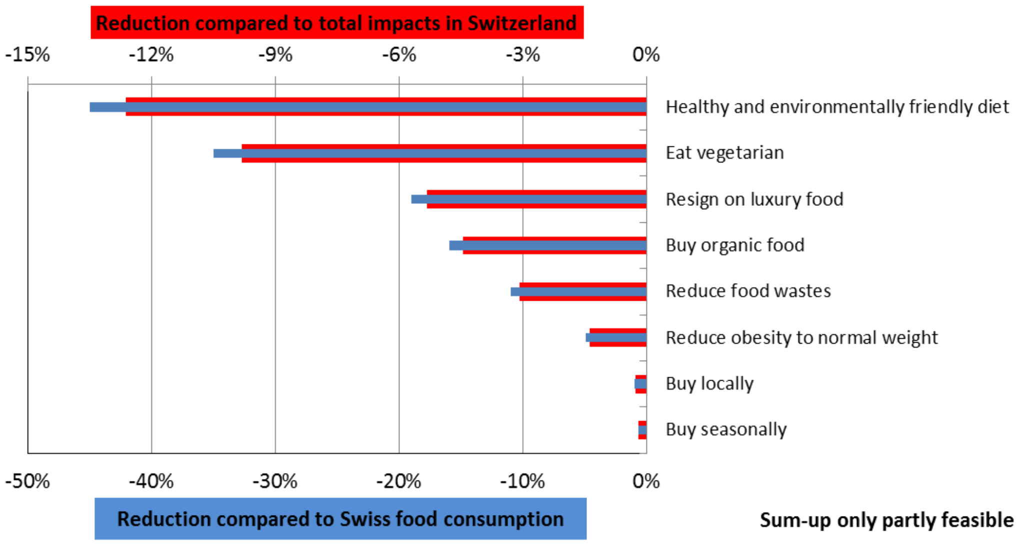


Figure 5. Reduction potentials for the total environmental impacts due to behavioural changes in food consumption evaluated with the ecological scarcity method 2006

4. Discussion

The approach developed in this research project allows a comparison of different options for the reduction of environmental impacts due to food consumption patterns or within other fields of consumption. The research focuses on the options that can be followed up by private food consumers. An additional reduction of environmental impacts is possible if further measures are taken within the production chain. Such options would be for example the reduction of wastes throughout the production chain or the improvement of farming practices from an environmental point of view.

The difficulties arising for single persons from implementing these options for sustainable lifestyles have not been considered for the conclusions in this article nor have rebound effects been taken into account. An evaluation based on carbon footprint or energy demand alone comes to partly different conclusions because in this case impacts from transporting and energy consumption (e.g. heated greenhouses) become more important (Jungbluth et al., 2012a; Wiegmann et al., 2005).

Furthermore also reduction potentials in the consumption domains of mobility and energy use in private households were evaluated, but are not focussed on in this paper.

5. Conclusion

This research project shows that in order to reach a healthy and environmentally friendly consumption pattern several nutritional adjustments should be combined. Nevertheless, the reduction of meat and animal products is the most important issue from an environmental point of view. The second most promising approach is the reduction of luxury food. Considerable reduction of total environmental impacts are possible if consumers would follow these suggestions.

6. References

BWL (2011) Berechnung theoretisches Produktionspotential Fruchtfolgeflächen. Bundesamt für wirtschaftliche Landesversorgung (BWL), Bern, CH, retrieved from: http://www.bwl.admin.ch/themen/01006/index.html?lang=de.

Carlsson-Kanyama A., Pipping Ekström M. and Shanahan H. (2003) Food and life cycle energy inputs: consequences of diet and ways to increase efficiency. In: Ecological Economics(DOI 10.1016/S0921-8009(02)00261-6).

Cordella M., Tugnoli A. and Santarelli F. (2009) An LCA approach to the Environmental Impact of Overweight and Obesity. University of Bologna.

Faist M. (2000) Ressourceneffizienz in der Aktivität Ernähren: Akteurbezogene Stoffflussanalyse. Dissertation Nr. 13884. Eidgenössische Technische Hochschule Zürich, Zürich, retrieved from: e-collection.ethbib.ethz.ch/show?type=diss&nr=13884.

Fazeni K. (2011) Energiebilanzen und Treibhausgasemissionen der österreichischen Landwirtschaft unter Berücksichtigung von Ernährungsgewohnheiten. In proceedings from: Abschlussveranstaltung Gesunde Ernährung und Nachhaltigkeit, 21.06.2011, retrieved from: http://iwr.tuwien.ac.at/wasser/projektseiten/gern/kaminabende.html.

Frischknecht R., Steiner R. and Jungbluth N. (2009) The Ecological Scarcity Method - Eco-Factors 2006: A method for impact assessment in LCA. Federal Office for the Environment FOEN, Zürich und Bern, retrieved from: www.bafu.admin.ch/publikationen/publikation/01031/index.html?lang=en.

Grießhammer R., Brommer E., Gattermann M., Grether S., Krüger M., Teufel J. and Zimmer W. (2010) CO_2-Einsparpotenziale für Verbraucher. Öko-Institut, Berlin, retrieved from: http://www.verbraucherfuersklima.de/cps/rde/xbcr/projektklima/CO2-Einsparpotentiale-Oeko-Institut.pdf.

Gustavsson J., Cederberg C., Sonesson U., van Otterdijk R. and Meybeck A. (2011) Global food losses and food waste: Extent,causes and prevention. Swedish Institute for Food and Biotechnology (SIK) commissioned by Food And Agriculture Organization Of The United Nations, Gothenburg, Sweden, retrieved from: http://www.fao.org/fileadmin/user_upload/ags/publications/GFL_web.pdf.

Jungbluth N. (2000) Umweltfolgen des Nahrungsmittelkonsums: Beurteilung von Produktmerkmalen auf Grundlage einer modularen Ökobilanz. Dissertation Nr. 13499. Eidgenössische Technische Hochschule Zürich, Umweltnatur- und Umweltsozialwissenschaften, dissertation.de, Berlin, D, retrieved from: www.esu-services.ch/address/niels/nahrungsmittelkonsum/.

Jungbluth N., Nathani C., Stucki M. and Leuenberger M. (2011) Environmental impacts of Swiss consumption and production: a combination of input-output analysis with life cycle assessment. Environmental studies no. 1111. ESU-services Ltd. & Rütter+Partner, commissioned by the Swiss Federal Office for the Environment (FOEN), Bern, CH, retrieved from: www.esu-services.ch/projects/ioa/ or www.umwelt-schweiz.ch.

Jungbluth N., Itten R. and Stucki M. (2012a) Umweltbelastungen des privaten Konsums und Reduktionspotenziale. ESU-services Ltd. im Auftrag des BAFU, Uster, CH, retrieved from: http://www.esu-services.ch/projects/lifestyle/.

Jungbluth N. and Itten R. (2012) Umweltbelastungen des Konsums in der Schweiz und in der Stadt Zürich: Grundlagendaten und Reduktionspotenziale. ESU-services GmbH im Auftrag der Stadt Zürich (unpublished), Zürich.

Jungbluth N., Flury K., Büsser S., Stucki M., Schori S., Itten R. and Frischknecht R. (2012b) Life cycle inventory database on demand: EcoSpold LCI database of ESU-services. ESU-services Ltd., Uster, CH, retrieved from: www.esu-services.ch/data/data-on-demand/.

Kramer K. J. (2000) Food matters: On reducing energy use and greenhouse gas emissions from household food consumption. Ph.d. Thesis. Center for Energy and Environmental Studies of the University of Groningen (IVEM RUG), The Netherlands, retrieved from: www.foodmatters.tmfweb.nl.

Leuenberger M. and Jungbluth N. (2009) Ökoprofil von vegetarischen und fleischhaltigen Grossküchenmahlzeiten. ESU-services GmbH im Auftrag des WWF Schweiz, Uster, CH.

Meier T. and Christen O. (2012) Factor gender in an environmental assessment of the consumption of animal- and plant-based foods in Germany. In: Int J LCA, **17**(5), pp. 550-564, 10.1007/s11367-012-0387-x retrieved from: http://www.springerlink.com/content/l415530205u58376/.

Schweizerischer Bauernverband (2007) Statistische Erhebungen und Schätzungen - über Landwirtschaft und Ernährung 2006. Abt. Statistik, Brugg, CH, retrieved from: www.bauernverband.ch.

Seemüller M. (2001) Ökologische bzw. konventionell-integrierte Landbewirtschaftung: Einfluss auf die Ernährungssicherung in Deutschland in Abhängigkeit vom Konsumanteil tierischer Lebensmittel. In: Zeitschrift für Ernährungsökologie, **2**(2), pp. 94-96, retrieved from: www.scientificjournals.com/erno/welcome.htm.

Taylor C. (2000) Ökologische Bewertung von Ernährungsweisen anhand ausgewählter Indikatoren. Inauguraldissertation. Justus-Liebig-Universität Gießen, retrieved from: bibd.uni-giessen.de/ghtm/2000/uni/d000074.htm.

Uitdenbogerd D. E., Brouwer N. M. and Groot-Marcus J. P. (1998) Domestic energy saving potentials for food and textiles - An empirical study. 2. Wageningen Agricultural University, Wageningen.

von Koerber K., Männle T., Leitzmann C., Eisinger M., Watzl B. and Weiss G. (1999) Vollwert-Ernährung: Konzeption einer zeitgemässen Ernährungsweise. 9 Edition. Haug, ISBN 3-7760-1734-1, Heidelberg.

Wiegmann K., Eberle U., Fritsche U. and Hünecke K. (2005) Umweltauswirkungen von Ernährung - Stoffstromanalysen und Szenarien. Öko-Institut, Darmstadt.

Würtenberger L. (2003) Import and Export of Agricultural Land Use. Diplomarbeit Nr. 1/03. Eidgenössische Technische Hochschule, Zürich.

Zwick M. M. and Müller C. (2012) Umweltauswirkungen des Ernährungsverhaltens von normalgewichtigen, übergewichtigen und adipösen Jugendlichen. In: Stuttgarter Beiträge zur Risiko und Nachhaltigkeitsforschung: Juvenile Adipositas aus soziologischer, psychologischer und sozialökologischer Perspektive, Vol. 24 (ed. Zwick M. M.).

Gender and dietary recommendations in an IO-LCA of food consumption in Germany

Toni Meier*, Olaf Christen

Institute of Agricultural and Nutritional Sciences, Martin-Luther-University Halle-Wittenberg, Betty-Heimann-Straße 5, 06120 Halle (Saale), Germany

* Corresponding author. E-mail: *toni.meier@nutrition-impacts.org*

ABSTRACT

Besides technical improvements and a reduction of food losses in the food chain, diet shifts offer practicable opportunities to reduce environmental impacts in the agri-food sector on a low-cost level. Due to their production intensity, different foods of animal or plant origin play a crucial role in the assessment of the environmental impacts of human nutrition and dietary habits. Based on a representative nutrition survey in Germany from the year 2006, a life cycle assessment was conducted to quantify nutrition-related impacts, with a special focus on the socio-demographic factor gender in comparison with different dietary recommendations, dietary styles and the average diet profile 20 years ago. Regarding the analysed scenarios the highest impact changes would be expected from the vegan and the ovo-lacto vegetarian diet. The impact potentials of the recommendations of UGB and D-A-CH range on the 3rd and 4th position, but are still significant. Concerning gender the average female diet is already closer to the recommendations. In comparison to the years 1985-89 all indicators (exception blue water) show lower impacts, mainly derived by changes in the diet. In comparison to that, impact changes due to food losses were lower and mainly contrarian, which could be explained by higher food losses in 2006 compared to 1985-89.

Keywords: Input-output LCA, direct land use change & land use (dLUC, LU), diet shift, nutrition patterns, dietary recommendations

1. Introduction

Human nutrition has a strong effect on environmental impacts. Taking political considerations into account (EC 2011), nutritionally acceptable und environmentally sound measures have to be developed to cope with current agro-ecological challenges. Various studies with a life cycle perspective have identified food supply as one of the main contributors to environmental impacts (Nijdam et al., 2005, Tukker et al., 2011). To facilitate political and economic decisions various life cycle assessments (LCA) have been elaborated: (i) either on a product level basis or (ii) on a diet basis to identify the most polluting food items or to compare dietary choices (Carlsson-Kanyama 1998, Taylor 2000, Peters et al., 2007, Muñoz et al., 2010, Jungbluth et al., 2011, Meier & Christen 2012).

Besides technical solutions (improvements in efficiency during production and processing) and a reduction of food losses, changes in diets and nutrition patterns are also discussed with a view to decreasing environmental impacts of the agri-food sector (Stehfest 2009, Popp et al., 2010).

The first part of the study builds mainly upon Meier & Christen (2012), where the influence of the factor gender in an LCA of food consumption was analysed. Here, we examine these gender-related differences in comparison to nutrition recommendations (D-A-CH & UGB), nutrition styles (ovo-lacto vegetarian, vegan) and the average diet profile 20 years ago (in the years 1985-89).

2. Materials and methods

For the study, representative data sets concerning German food production and consumption were used (BMELV 2009, BML 1991). Exact, subgroup-specific intake data was provided by the both National Nutrition Surveys from the years 1985-89 and 2006 (Kübler et al., 1995, MRI 2008). The environmental impact assessment was based mainly on the input-output tables of SEEA (System of Environmental and Economic Accounting, Schmidt & Osterburg 2011). To consider the impacts of food imports, emissions from direct land use change and land use (dLUC, LU), food processing, trade/transport and packaging, these were complemented by several LCA data sets - e.g. Leip et al., (2010), the Danish LCA Food database (Nielsen et al., 2003), Institute of Applied Ecology (2010) and Mekonnen & Hoekstra (2012) for blue water. Thus, the system boundaries are set cradle-to-store. The functional unit considered on the product level refers to 1 kg consumed product. The reference year in the study is the year 2006. According to ISO 14040/14044 (2006) the four distinct steps of an LCA have been completed.

As regards environmental impact assessment, global warming potential (GWP) was assessed, which included emissions from direct land use change and land use (dLUC, LU), along with five inventory indicators (ammonia emissions, land use, blue water use, phosphorous use and cumulative primary energy demand (CED)).

The following food groups were analysed: milk products (including butter, high-fat milk products like cheese and cream, low-fat milk products like milk and yoghurt), meat products (including pork, beef/veal, poultry, goat/lamb), egg products, fish products, grain products, vegetables, legumes, fruits, nuts & seeds, potato products, vegetal oils & margarine, sugar/sweets.

For the comparison with dietary recommendations and diet styles the following quantifiable food-related dietary profiles were examined (Table 1). In contrast to nutrient-based dietary recommendations (NBDR) there exist food-based dietary recommendations (FBDR). These are more consumer-friendly and could be, if sufficiently determined (ample, consistent and standardised product categories), compared and analysed environmentally.

Table 3. Types of dietary recommendations and diet styles analysed

	Description	Reference
Dietary recommendations	D-A-CH (official recommendations for Germany, Austria and Switzerland)	DGE (2008)
	UGB (alternative recommendations by the Federation for Independent Health Consultation with less meat, but more legumes and vegetables)	UGB (2011)
Dietary styles	Ovo-lacto vegetarian (plant-based diet with egg and milk products, without meat and fish)	USDA, USDHHS (2010)
	Vegan (totally plant-based diet, without meat, milk, fish and egg products and instead more fortified soy-based milk products, more legumes, nuts and seeds	USDA, USDHHS (2010)

As entries concerning alcoholic beverages (beer, wine, spirits) as well as coffee, tea and cocoa do not exist in most of the recommendations and diets, these product groups were omitted in the assessment. Nevertheless, as all recommendations consider the intake of fruits and sugar via soft drinks and juices, we considered this intake pathway, too. Furthermore, grains in beer were reallocated to the corresponding main group ´grain products´. Table 2 gives an overview of the intake amounts analysed based on 2,000 kcal person^{-1} day^{-1}.

Table 4. Intake amounts analysed based on 2,000 kcal person^{-1} day^{-1}

	Intake 1985-89	Intake 2006 mean	Intake 2006 men	Intake 2006 women	D-A-CH	UGB	ovo-lacto vegetarian	vegan
	g p^{-1} d^{-1}							
Butter	20	12	13	11	11	10	8	-
High-fat milk products (cheese, cream)	38	46	42	51	55	75	732 (a)	-
Low-fat milk products (milk, yoghurt)	169	207	191	223	225	375		-
Vegan milk products	-	-	-	-	-	-		732 (a)
Meat products	158	103	121	84	64	40	-	-
Egg products	31	18	18	19	9	9	16	-
Fish products	17	25	25	25	26	25	-	-
Grains	293	278	299	258	362	403	295	295
Vegetables	145	231	192	270	400	500	245	245
Legumes	- (b)	- (b)	- (b)	- (b)	- (b)	52	124	128
Fruits	134	347	276	419	250	200	250	250
Nuts & seeds	2	3	3	4	- (c)	- (c)	21	26
Potato products	108	80	80	80	112	82	107	107
Vegetal oils, margarine	22	15	15	15	24	30	27	34
Sugar	54	70	71	69	32	32	32	32
Sum	1192	1437	1345	1528	1571	1833	1857	1850

(a) in whole milk equivalents
(b) legumes are included in vegetables
(c) D-A-CH & UGB do not have quantifiable recommendations for nuts & seeds

To allow an environmental assessment the intake amounts were converted to consumption amounts. Therefore statistically derived consumption data for the years considered, 2006 and 1985-89, was divided by the corresponding intake amounts. Thus, the conversion could be embedded consistently in official data. The so elaborated product-specific conversion factors (CF = consumption / intake) as well as the environmental impact factors used in the assessment are outlined in Table 3.

Table 5. Conversion and environmental impact factors based on the functional unit

		Butter	High-fat milk products (cheese, cream)	Low-fat milk products (milk, yoghurt)	Vegan milk products (in whole milk equivalents)	Meat products	Egg products	Fish products	Grains	Vegetables	Legumes	Fruits	Nuts & seeds	Potato products	Vegetal oils, margarine	Sugar	Modelled as
Conversion factor 1985-89	= consumption / intake	0.77	0.89	0.89	0.73	0.48	0.69	0.52	0.89	0.68	0.68	0.48	0.15	0.42	0.57	0.29	Data concerning food consumption from BML (1991); data concerning food intake from Kübler et al. (1995)
Conversion factor 2006		0.71	0.73	0.73	0.73	0.46	0.54	0.61	0.96	0.85	0.85	1.04	0.33	0.41	0.28	0.56	Data concerning food consumption from BMELV (2009); data concerning food intake from MRI (2008)
CO_{2e}-emissions	kg / kg	19.7	10.0	2.7	0.7	9.3	3.1	2.4	1.5	0.8	1.1	0.8	0.8	0.5	3.4	1.9	Data concerning domestic agriculture & upstream processes from Schmidt & Osterburg (2011) and Brentrup & Pallière (2008); data concerning fish from Nielsen et al. (2003); data concering fruits partly from Sanjuan et al. (2005); data concerning dLUC, LU from Leip et al. (2010); data concerning processing from BMELV (2009); data concerning trade/transport from DIW (2008) and data concerning packaging from Institute of Applied Ecology (2010). If no data for imported products were available, domestic conditions were assumed.
NH_3-emissions	g / kg	82.5	39.4	9.7	0.2	39.6	19.7	0.2	1.9	0.7	0.6	0.6	0.6	0.3	3.3	1.5	Data concerning domestic agriculture from Schmidt & Osterburg (2011); data concerning fish from Nielsen et al. (2003); data concerning processing from BMELV (2009); data concerning trade/transport from DIW (2008) and data concerning packaging from Institute of Applied Ecology (2010). If no data for imported products were available, domestic conditions were assumed.
Land use	m^2 / kg	20.7	9.9	2.4	0.7	11.0	3.8	0.2	1.8	0.5	2.1	0.9	2.9	0.3	4.1	1.2	Data concerning domestic agriculture from Schmidt & Osterburg (2011); data concerning packaging from Institute of Applied Ecology (2010). Data for imported products from FAO Stat. (2012).
Water use (blue)	l / kg	86.0	44.3	15.6	6.4	36.7	8.8	14.7	5.2	27.4	8.8	87.6	1425.3	5.7	15.7	9.5	Data concerning domestic agriculture from Schmidt & Osterburg (2011) and Mekonnen & Hoekstra (2010); data concerning processing from BLE (2007) and data concerning packaging from Institute of Applied Ecology (2010). Data for imported products from Mekonnen & Hoekstra (2010).
Phosphorous use	g / kg	70.0	33.5	8.2	1.3	33.0	10.3	4.4	5.3	2.7	3.7	1.4	1.4	0.9	9.7	3.6	Data concerning domestic agriculture from Schmidt & Osterburg (2011). P use during processing was not examined.
Cumulative energy demand (CED)	MJ / kg	52.7	33.0	12.5	7.4	44.5	20.2	13.1	11.2	10.2	7.7	11.1	11.1	5.0	19.4	18.0	Data concerning domestic agriculture & upstream processes from Schmidt & Osterburg (2011) and Brentrup & Pallière (2008); data concerning fish from Nielsen et al. (2003); data concering fruits partly from Sanjuan et al. (2005); data concerning processing from BMELV (2009); data concerning trade/transport from DIW (2008) and data concerning packaging from Institute of Applied Ecology (2010). If no data for imported products were available, domestic conditions were assumed.

3. Results

For all indicators the results show strong variation between the genders. Even if the physiologically different consumption amounts among men and women are levelled out on the basis of 2,000 kcal $person^{-1}$ day^{-1}, men still show a higher impact in terms of GWP (CO_{2e} +8%), ammonia emissions (+14%), land use (+11%), P use (+10%) and CED (+2%). In contrast, women demonstrate a higher water demand (+18%; Table 4 and Fig. 1). These differences are primarily caused by a higher share of meat products and butter in the usual diet of men as well as more fruit, vegetables and nuts & seeds in the typical diet of women.

Table 6. Environmental impacts of food consumption in the years 1985-89 and 2006 (incl. genders) in Germany as well as of several dietary recommendations & dietary styles based on 2,000 kcal person^{-1} day^{-1}

		1985-89 mean	2006 mean	2006 men	2006 women	D-A-CH	UGB	ovo-lacto vegetarian	vegan
CO_{2e} emissions	t person^{-1} year^{-1}	2.3	**2.0**	2.1	1.9	1.8	1.8	1.5	0.9
NH_3 emissions	kg person^{-1} year^{-1}	7.7	**6.4**	6.8	6.0	5.1	4.6	3.9	0.7
Land use	m² person^{-1} year^{-1}	2,429	**2,059**	2,170	1,946	1,749	1,693	1,514	1,019
Water use (blue)	m³ person^{-1} year^{-1}	26.8	**27.1**	24.7	29.5	19.6	19.5	51.1	57.0
Phosphorous use	kg person^{-1} year^{-1}	7.4	**6.3**	6.7	6.0	5.6	5.4	4.5	2.3
Cumulative energy demand	MJ person^{-1} year^{-1}	13.0	**11.6**	11.7	11.4	10.7	11.1	10.1	8.4

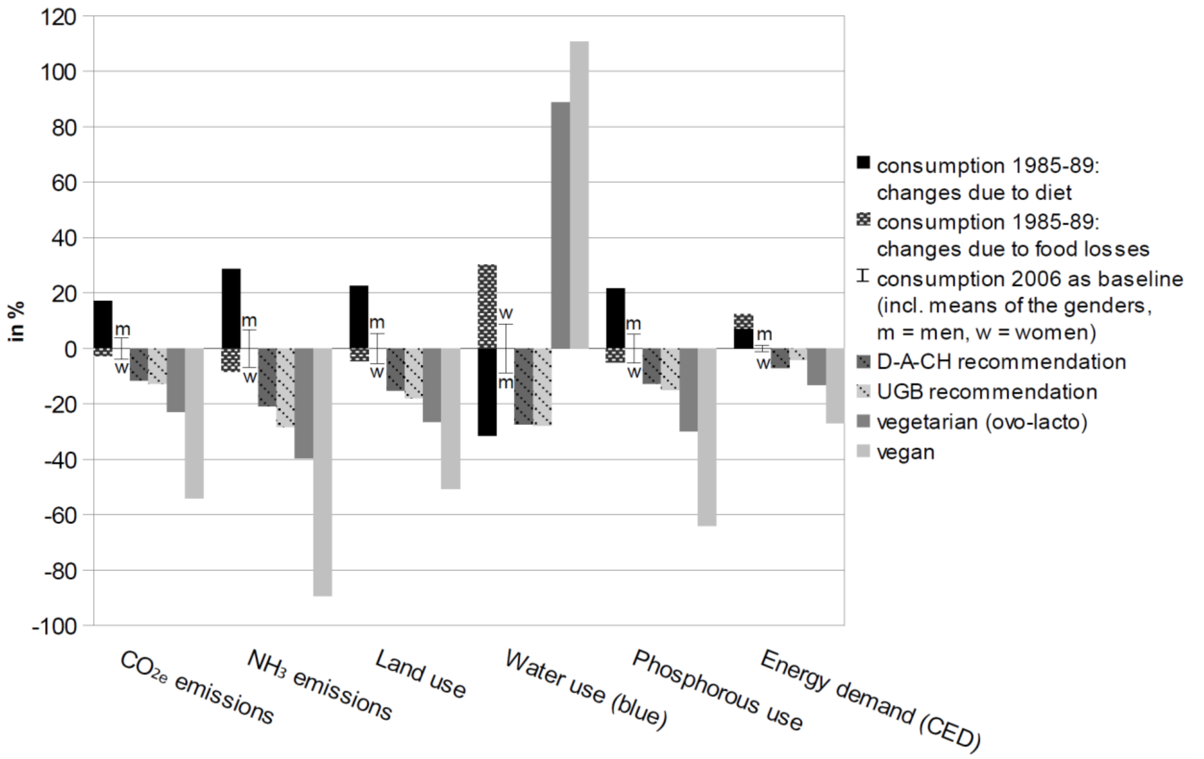


Figure 6 Environmental impacts of food consumption in 2006 in Germany as baseline scenario in comparison to 1985-89, genders and dietary scenarios based on 2,000 kcal person^{-1} day $^{-1}$

In comparison to the dietary recommendations and the dietary styles, which can be mainly characterised by an increasing share of legumes, nuts and vegetables in the profiles instead of meat, butter, egg and fish products as well as fruits (D-A-CH > UGB > vegetarian > vegan), both genders could reduce the impacts of their diets if they were to be more in line with the recommendations or diet styles. With the exception of blue water, the reduction potentials for men are twice as high as women's. In other words, the average female diet is already closer to the recommendations. Nevertheless, women's average diet corresponds to higher blue water use, mainly caused by higher consumption of fruits as well as of nuts & seeds, which are often produced in water-scarce areas in foreign countries. According to FAO trade statistics (FAO 2011), fruits imported in 2006 into Germany were mainly produced in Spain and Italy, whereas nuts & seeds were mainly imported from China, USA, Turkey and Iran. Related to the average intake in 2006 the strongest reduction potentials were determined for the vegan (-27% CED – -89% NH_3) and the ovo-lacto vegetarian diets (-13% CED – -40% NH_3), with the exception of blue water (vegan: +110%, vegetarian: +89%). Here we have to bear in mind that for the recommendations (D-A-CH, UGB) quantifiable intake amounts for nuts & seeds do not exist, although an increased intake in these scenarios would be probable.

However, in comparison with the environmental impacts caused by average nutrition in the years 1985-89, almost all indicators, with the exception of blue water, show a reduced impact. Due to different diets and different conversion factors (and therefore food losses) in 2006 and 1985-89, the observed differences could be caused either by variations in the average diet or varying food losses (Fig. 1).

For the reductions observed the main driver has been a shift in diets, with the exception of blue water. Here, mainly caused by an increased intake of fruits, blue water also increased accordingly. But this rise was almost compensated by less food losses in 2006 (for fruits). For the other indicators (GHG, NH_3, land use, P use), which are more driven by animal products, increased food losses partially countervailed gains achieved through shifting diets. Beneficial reductions by both means (diet shift and less food losses) have been observed only for the cumulative energy demand (CED).

Concerning sensitivity analysis, a perturbation analysis was performed with a variation of ± 25% of the input parameters.

4. Discussion

Taking different reference years, countries and system boundaries into consideration our results are comparable to other studies (e.g. Taylor 2000, Peters et al., 2007, Muñoz et al., 2010, Jungbluth et al., 2011). Nevertheless the following limitations of our study should be mentioned:

- system boundaries cradle-to-store (not cradle-to-grave)
- attributional approach, although data for fish was generated by a consequential approach
- for the scenario analysis (comparison with recommendations and diet styles) a consequential approach would be more appropriate to also analyse rebound effects (e.g. market effects)
- for GHG emissions, NH_3 emissions, P use and CED, if no separate data for imported products was available then these were modelled as domestically produced
- although different intake and production data were used, for 1985-89 the same production conditions (and therefore production efficiencies), import shares and import countries were assumed
- nuts & seeds were omitted in the scenario analysis of the recommendations (D-A-CH, UGB)
- due to the ongoing discussion about water in LCAs, we only considered blue water

With regard to the diet styles analysed it should be noted that a vegan diet, and to a lesser extent a vegetarian one, could provide an insufficient supply of essential nutrients.

5. Conclusion

The study shows that within one society distinct diet profiles of men and women with markedly different environmental impacts are already established. Nevertheless, with regard to dietary recommendations and alternative diet styles (vegetarian, vegan) men, and to a lesser extent women as well, could achieve significant environmental benefits (with the exception of blue water use). In comparison to the years 1985-89, all indicators showed reduced environmental impacts, but with distinct contributions of the main drivers (diet shifts and food losses). Further research should also consider health impact assessments to ensure that alterations in diet profiles due to environmental constraints do not lead to disadvantageous public health effects. Particular attention should be paid here to potentially undernourished subgroups (such as toddlers, children, the elderly, sick people, pregnant women etc.).

6. References

BLE 2007. Energie,- und Wasserverbrauch des Ernährungsgewerbes 2006. Meldungen aus der Ernährungswirtschaftmeldeverordnung (EWMV), unveröffentlicht. Bundesanstalt für Landwirtschaft und Ernährung, Bonn.

BMELV 2009. Statistisches Jahrbuch über Ernährung, Landwirtschaft und Forsten in der Bundesrepublik Deutschland. 53. Jahrgang, Bundesministerium für Ernährung, Landwirtschaft und Verbraucherschutz, Bonn/Berlin.

BML 1991. Statistisches Jahrbuch über Ernährung, Landwirtschaft und Forsten der Bundesrepublik Deutschland 1991. Bundesminister für Landwirtschaft, Ernährung und Forsten, Bonn.

Brentrup, F., Pallière, C. 2008. GHG emissions and energy efficiency in European nitrogen fertiliser production and use. International Fertiliser Society, York.

DGE 2008. Vollwertig essen und trinken nach den 10 Regeln der DGE. Deutsche Gesellschaft für Ernährung e.V., aid Infodienst, Bonn, p. 7-25.

DIW 2008. Verkehr in Zahlen. Deutsches Institut für Wirtschaftsforschung. 37. Jg. Dt. Verkehrs-Verlag, Hamburg, p. 247.

EC 2011. A roadmap for moving to a competitive low carbon economy in 2050. European Commission, Brussels.

FAO Stat 2011. Detailed trade matrix for Germany 2006. Food and Agriculture Organization of the UN, Rome. http://faostat.fao.org. Accessed 25 November 2011.

FAO Stat 2012. Production, crops and yields. Statistical Division of the Food and Agriculture Organization of the UN, Rome. http://faostat.fao.org. Accessed 21 January 2012.

Institute of Applied Ecology 2010. GEMIS 4.6 — Global Emissions Model of Integrated Systems. Institut für angewandte Ökologie, Freiburg.

ISO 2006. Environmental Management — Life Cycle Assessment — Principles and framework. (ISO 14040:2006). International Organization for Standardization, Paris.

Jungbluth, N., Nathani, C., Stucki, M., Leuenberger, M. 2011. Environmental Impacts of Swiss Consumption and Production. A combination of input-output analysis with life cycle assessment. Federal Office for the Environment, Bern. Environmental studies no. 1111: pp. 171.

Kübler, W., Anders, H.J., Heeschen, W. 1995. Ergebnisse der Nationalen Verzehrsstudie (1985-1988) über die Lebensmittel- und Nährstoffaufnahme in der Bundesrepublik Deutschland. Band XI, VERA- Schriftenreihe, Gießen.

Leip, A.,Weiss, F.,Wassenaar, T., Perez, I., Fellmann, T., Loudjani, P., Tubiello, F., Grandgirard, D., Monni, S., Biala, K. 2010. Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS) – final report. European Commission, Joint Research Centre.

Meier, T., Christen, O. 2012. Gender as a factor in an environmental assessment of the consumption of animal and plant-based foods in Germany. in: Int J Life Cycle Assess 17 (5), p. 550-564.

Mekonnen, M.M., Hoekstra, A.Y. (2010): The green, blue and grey water footprint of crops and derived crop products, Value of Water Research Report Series No. 47, UNESCO-IHE, Delft, the Netherlands.

MRI 2008. Nationale Verzehrsstudie II — Ergebnisbericht, Teil 2 — Die bundesweite Befragung zur Ernährung von Jugendlichen und Erwachsenen. Max Rubner Institute, Karlsruhe, pp. 174–234.

Muñoz, I., Milà i Canals, L., Fernández-Alba, A.R. 2010. Life cycle assessment of the average Spanish diet including human excretion. in: Int J Life Cycle Assess, p. 794-805.

Nielsen, P.H., Nielsen, A.M., Weidema, B.P., Dalgaard, R., Halberg, N. 2003. LCA food data base. www.lcafood.dk. Accessed 9 September 2011.

Nijdam D.S., Wilting, H.C. Goedkoop, M.J, Madsen, J. 2005. Environmental load from Dutch private consumption—how much damage takes place abroad? in: J Ind Ecol 9 (1–2): p. 147-168.

Peters, C.J., Wilkins, J.L., Fick, G.W. (2007): Testing a complete-diet model for estimating the land resource requirements of food consumption and agricultural carrying capacity: The New York State example. In: Renew Agr Food Syst 22 (02). p. 145–154.

Popp, A., Lotze-Campen, H., Bodirsky, B. 2010. Food consumption, diet shifts and associated non-CO_2 greenhouse gases from agricultural production. in: Global Environ Chang 20 (3), p. 451-462.

Sanjuan, N., Ubeda, L., Clemente, G. Mulet, A., Girona, F. 2005. LCA of integrated orange production in the Comunidad Valenciana (Spain). In: IJARGE 4 (2): p. 163.

Schmidt, T., Osterburg, B. 2011. Berichtsmoduls ‘Landwirtschaft und Umwelt’ in den umweltökonomischen Gesamtrechnungen II. Tabellenband. Von Thünen-Institut (vTI). Braunschweig, p. 29.

Stehfest, E., Bouwman, L., Vuuren, D.P., Elzen, M.G.J., Eickhout, B., Kabat, P. 2009. Climate benefits of changing diet. in: Climatic Change 95 (1-2), p. 83-102.

Taylor C. 2000: Ökologische Bewertung von Ernährungsweisen anhand ausgewählter Indikatoren. Dissertation. Justus-Liebig-Universität. Gießen, pp. 179.

Tukker, A., Goldbohm, R.A., de Koning, A., Verheijden, M., Kleijn, R., Wolf, O. 2011. Environmental impacts of changes to healthier diets in Europe. in: Ecol Econ 70 (10), p. 1776-1788.

UGB 2011. Recommendations for a whole food nutrition. Verband für Unabhängige Gesundheitsberatung e.V. (UGB) - Federation for independent health consultation, Wettenberg/Gießen.

USDA, USDHHS 2010. Dietary Guidelines for Americans 2010. 7th Edition. U.S. Department of Agriculture, U.S. Department of Health and Human Services, Washington, DC, p. 81-82

Comparing environmental impacts of end-of-life treatments of food waste

Mischa Zschokke*, Thomas Kägi, Fredy Dinkel

Carbotech AG, environmental consulting, 4002 Basel, Switzerland
Corresponding author. E-mail: *m.zschokke@carbotech.ch*

The goal of this study was to compare the environmental burden of landfilling food waste with three alternative biowaste treatments "composting", "anaerobic digestion" and "municipal solid waste incineration". The life cycle inventories for anaerobic digestion and composting includes a new approach to account for benefits in using biowaste as a fertiliser substitute. Depending on the impact assessment method used, the ranking of the different treatment methods tend to vary. But taking into account the range of uncertainty the three examined treatment methods show comparable environmental impacts. Landfilling of food waste in contrast results in a much higher environmental impact compared to the other three treatment methods. As all investigated treatment methods show similar results, the decision on which technique to use can be based on other factors such as economics, available infrastructure or even on the composition and nature of the food waste because different methods are favourable for treatment.

Keywords: biowaste, benefits of biowaste, incineration, composting, anaerobic digestion

1. Introduction

Every year huge quantities of edible food end up in landfills worldwide (e.g. 7 million tonnes in the United Kingdom and 34 million tonnes in the United States (Eunomia 2006, WRAP 2007a and 2007b, EPA 2011). Roughly one third originates from producers/supply chain, one third from retail and the final third from regular households (Sibrián et al., 2006, Parfitt et al., 2010,).

In addition to the costs for disposal, these landfills generate large amounts of greenhouse gases. Landfill gas emissions are one of the largest anthropogenic sources of methane especially because of food waste (Adhikari 2006). In the United States food waste now represents the single largest component of municipal solid waste reaching landfills and incinerators, and generates more than 16 percent of all methane emissions in that country (EPA 2012). Not only could the direct emissions from landfills be decreased by reducing the amount of landfilled food waste but the use of alternate methods for treatment of food waste could further reduce the environmental impact.

2. Methods

2.1 Goal and Scope

The goal of this study was to compare the environmental burden of landfilling food waste with three alternative biowaste treatments "composting", "anaerobic digestion" and "municipal solid waste incineration" (MSWI) as described in Dinkel et al., (2011).

The functional unit used in the presented study is 1 kg of treated food waste. The applied inventory methodology is derived from the ecoinvent version 2.2 guidelines (Frischknecht R. and Jungbluth N., 2007). Data for the investigated methods of treatment are based on existing ecoinvent version 2.2 processes and were extended and updated with new values in the following fields:

- emissions from anaerobic digestion: updated values for N_2O, CO_2, CH_4 and NH_3 in the digestion process and from spreading digestate
- emissions from composting: updated values for N_2O, CO_2, CH_4 and NH_3
- TCDD-2,3,7,8-emissions in municipal solid waste incineration were adjusted to account for the current regulatory values
- the heating value of biowaste was adapted according to own calculations to be suitable to model incineration in municipal solid waste incineration plants

2.2 Inventory data

In particular, the emissions for composting and anaerobic digestion were updated by field measurements and generally show lower values than previously reported. The methane emissions in the current version of ecoinvent are overestimated by about 5 times.

The used values for the biological treatment methods are summarised in the following tables:

Table 7. Composting – converted to CO_2-equivalents per kg food waste

Emissions [g] / kg	Transport / Pre-treatment	Average	Biological Process	Average	Total
	[g]	in [g] CO_2eq	[g]	in [g] CO_2eq	in [g] CO_2eq
CH_4, biogenic	0.01-0.1	1.25	0.5-1.5	25.00	26.25
CO_2, biogenic			260.00		
CO_2, fossil	4 - 13	10.00	2 - 10	7.80	17.80
N_2O		0.00	max 0.05	14.90	14.90
Total		11		48	**59**

Table 8. Anaerobic digestion – converted to CO_2-equivalents per kg food waste

Emissions [g] / kg	Pre-storage	Aver-age	AD[a] Process	Aver-age	Storage/ Post-comp.	Aver-age	CHP[b]	Aver-age	Gas condi-tioning	Aver-age	Total
	[g]	in [g] CO_2eq	[g]	in [g] CO_2eq	[g]	in [g] CO_2eq	[g]	in [g] CO_2eq	[g]	in [g] CO_2eq	in [g] CO_2eq
CH_4, bio-genic	<= 0.1	1.25	0.5-0.8	15.00	1-2.5	37.50	0.5-1.5	25.00	0.1-1.5	12.50	78.75
CO_2, bio-genic			260.00								
CO_2, fossil	4 - 13	10.00	2.60	2.60		2.60					15.20
N_2O	0-0.010	2.98	0-0.010	14.90		14.90					32.78
Total		14		32		55		25		12	**126**

[a] anaerobic digestion; [b] combined heat and power generation

The life cycle inventories for anaerobic digestion and composting includes a new approach to account for benefits in using biowaste as a fertiliser substitute. Studies comparing different technologies to utilise bio-waste normally only take into account the benefits for energy and nutrients. These studies usually show that digestion or incineration is ecologically favourable to composting. In this study we used an approach pro-posed by Fuchs and Schleiss (2008) making a substitution with peat and straw in order to include the value of soil structure on applying compost or digestate. The effect of the new approach proposed on specific re-sults can be considerable as shown in Figs 1 and 2.

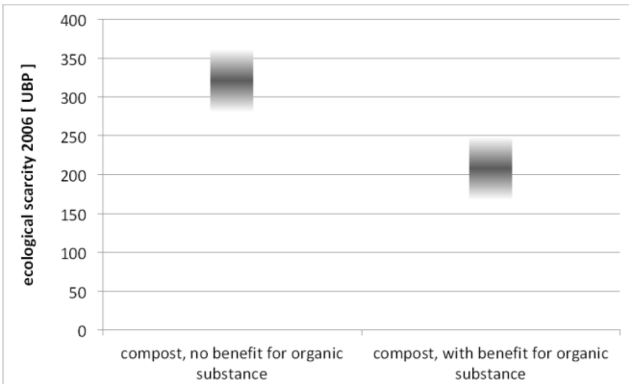


Figure 1. Effect of accounting organic substance in composting, using ecological scarcity method 2006

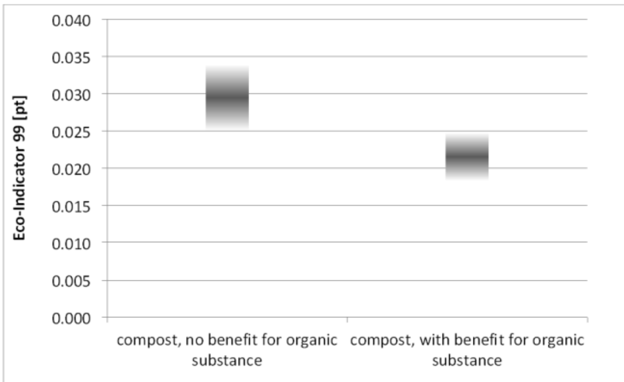


Figure 2. Effect of accounting organic substance in composting, using Eco-indicator 99 (H/A) Total

To assess the impacts of landfilling, several sources from the United States of America, from the United Kingdom, from the FAO (Food And Agriculture Organization Of The United Nations) and from Switzerland were used. The life cycle inventory calculations are based on ecoinvent 2.2 (ecoinvent 2010) and were up-dated with current values from literature (EPA 2009 and 2011, Gustavsson et al., 2011, WRAP 2007). It is assumed that the emitted landfill gas (LFG) consists of about 50 percent methane and about 50 percent car-bon dioxide as well as a small amount of non–methane organic compounds. For the presented comparison no capturing of landfill gas is assumed, all emissions go to the atmosphere. Leaching of certain compounds from landfill such as heavy metals is considered. Due to lack of data and the large margin of error with ecotoxicity data, the resulting uncertainties are fairly large.

2.3 Impact assessment

The LCA was performed using the software EMIS 5.7 (Environmental Management and Information System) developed by Carbotech AG and SimaPro 7.3.3 by PRé Consultants.

To compare the different treatment processes, the systems were expanded using an avoided burden and basket of benefits approach (Dinkel et al., 2009). Different environmental impacts were calculated and to evaluate the impacts the methods Eco-indicator 99 (Goedkoop and Spriensma, 2001) and ecological scarcity 2006 (Frischknecht, 2009) were used. Several sensitivity analyses were made to determine the robustness of the impact methods. Specific midpoint indicators such as global warming potential (IPCC 2007) are shown separately.

Inclusion of ReCiPe as a substitute method for Eco-indicator 99 was evaluated but had to be dismissed because of irregularities in the assessment of phosphorus emissions and the valuation of heavy metals in soil.

3. Results

The results presented are shown for Eco-indicator 99 and IPCC 2007, the ecological scarcity method 2006 is not displayed, as the outcomes are comparable to Eco-indicator 99.

Depending on the impact assessment method used, the ranking of the different treatment methods tend to vary. But taking into account the range of uncertainty the three examined treatment methods show comparable environmental impacts. As shown in the following figures, the landfilling of food waste in contrast results in a much higher environmental impact compared to the other three treatment methods.

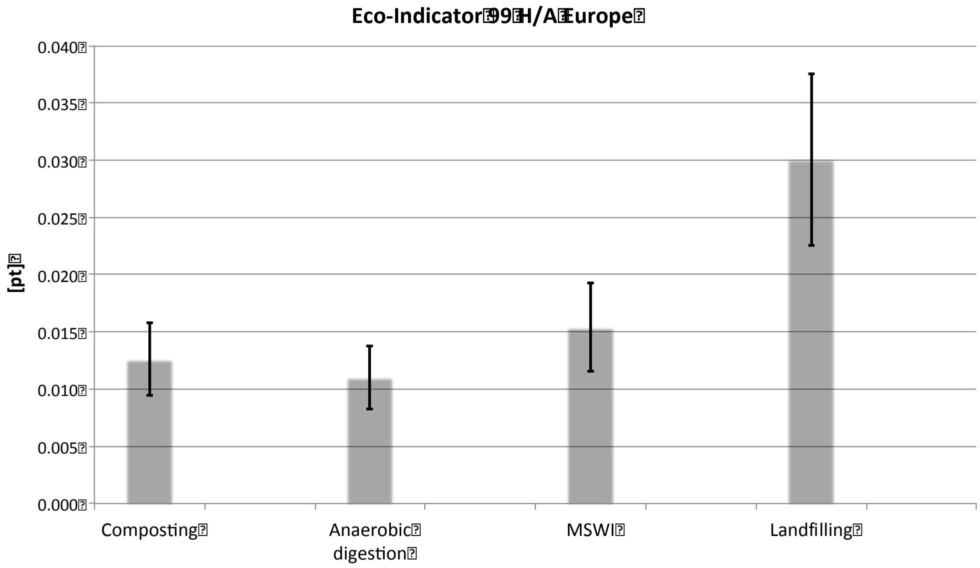


Figure 3. Eco-indicator 99 (H/A) Total (system modelled according to basket of benefits approach)

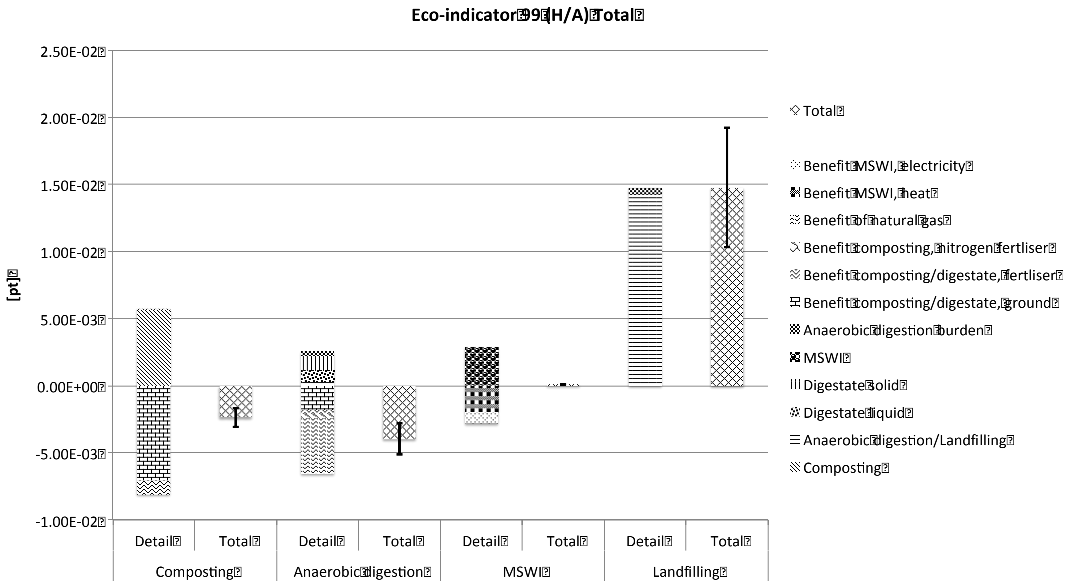


Figure 4. Eco-indicator 99 (H/A) Total, detail (system modelled according to avoided burden approach)

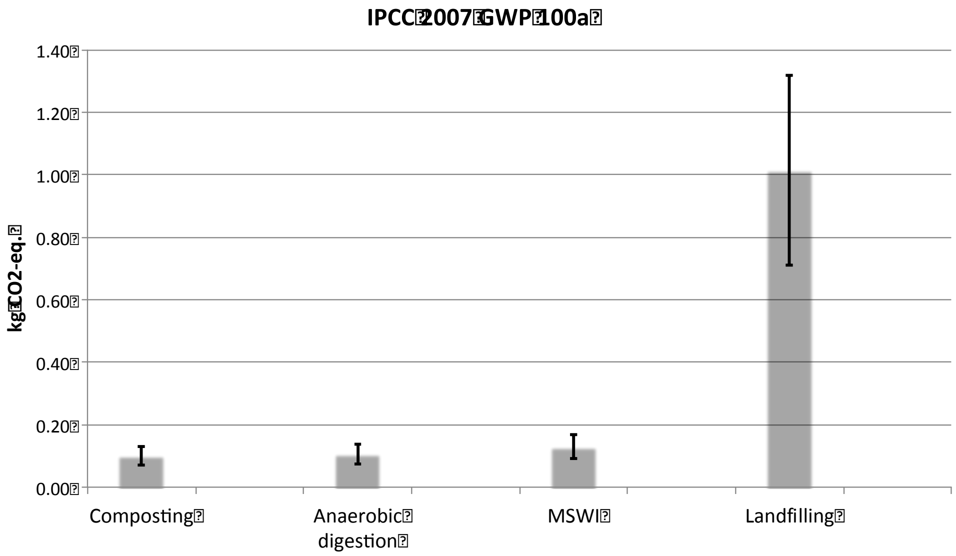


Figure 5. IPCC 2007 100a (system modelled according to basket of benefits approach)

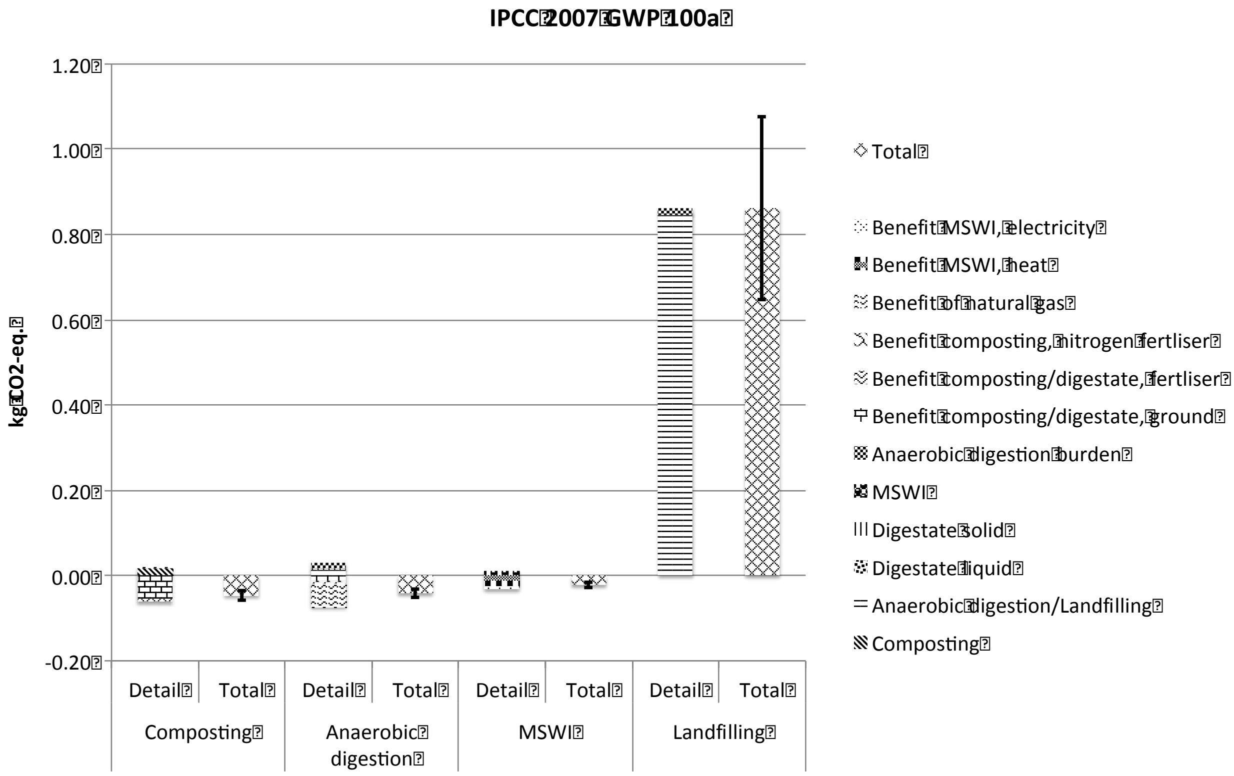


Figure 6. IPCC 2007 100a, detail (system modelled according to avoided burden approach)

Compared to Fig. 4 (Eco-indicator 99), Figs. 5 and 6 clearly show that the main burden with landfilling originates from greenhouse gas emissions, essentially from methane.

4. Discussion

Our calculations have confirmed that all three treatment methods display comparable environmental impacts. Landfilling always has significantly higher environmental burdens regardless of which impact assessment method is used. In this study we did not include efforts to capture and use emissions from landfills. Such programmes (e.g. the U.S. EPA's Landfill Methane Outreach Program (LMOP)) could considerably reduce the environmental impacts from landfills and help to use landfill gas as energy resource. We estimate that the possible reductions will nonetheless not be able to place landfilling ahead of the other three methods investigated.

The sheer amount of food waste still going to landfills basically makes any treatment method favourable to simple landfilling: in the United States 35 Million tonnes of food were sent to landfill in 2010, responsible for the emission of approximately 20 to 40 Tg of CO_2eq into the atmosphere.

As all investigated treatment methods show similar results, the decision on which technique to use can be based on other factors such as economics, available infrastructure or even on the composition and nature of the food waste because different methods are favourable for treatment.

5. Conclusions

This study shows the enormous emission reduction potential if food waste is not landfilled but otherwise treated. Naturally it would be even better to reduce the amount of food going to waste as all presented methods are only end of pipe solutions and the environmental impact of food production itself is normally much higher (usually by a factor of 2 to 20) than the impact of the landfill or any other treatment method. As long as we still lose about one third of the produced food along the chain (Gustavsson et al., 2011), we still have lots of room for improvement.

6. References

Adhikari B. K., October 2006, Waste Managment Research, vol. 24 no. 5 421-433

Dinkel et al., 2009, Ökobilanz zur Grüngutverwertung in Basel, Amt für Umwelt und Energie, Basel-Stadt

Dinkel et al., 2011, Ökobilanzen zur Biomasseverwertung, Forschungs- und Entwicklungsprogramm Biomasse und Holzenergie, Schwerpunkt Biomasse ohne Holzenergie, Bundesamt für Energie BFE ecoinvent, 2010, ecoinvent 2010, version 2.2, Swiss Center for Life Cycle Inventories, May 2010

EPA 2009, December 2010, Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Facts and Figures for 2009, United States Environmental Protection Agency, Solid Waste and Emergency Response (5306P) Washington, DC 20460

EPA 2011, November 2011, Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Facts and Figures for 2010, United States Environmental Protection Agency, Solid Waste and Emergency Response (5306P) Washington, DC 20460

EPA 2012, Inventory of U.S. greenhouse gas emissions and sinks: 1990 – 2010, April 15, 2012, U.S. Environmental Protection Agency, 1200 Pennsylvania Ave., N.W. Washington, DC 20460 U.S.A.

Eunomia (2006), "A Changing Climate for Energy from Waste?", Friends of the Earth, 26-28 Underwood Street, London

Frischknecht et al., (2009) Methode der ökologischen Knappheit - Ökofaktoren 2006. Bundesamt für Umwelt (BAFU)/ÖBU.

Frischknecht R. and Jungbluth N., ecoinvent report No.1, Overview and Methodology, 2007

Fuchs J., Schleiss K., 2008, Ökologische Bewertung der organischen Substanz, Zusatzmodul zu Forschungsprojekt Auswirkungen von Komposten und von Gärgut auf die Umwelt, Bodenfruchtbarkeit, sowie die Pflanzengesundheit, im Auftrag des BAFU, Bern

Goedkoop and Spriensma, The Eco-indicator 99 – a damage oriented method for LCIA, Methodology Report, Third edition, 22 June 2001, PRé Consultants B.V., Amersfoot http://www.pre-sustainability.com/content/reports

Gustavsson et al., 2011, Global food losses and food waste, Swedish Institute for Food and Biotechnology (SIK) Gothenburg, Sweden

IPCC, 2007. Climate Change 2007: The Scientific Basis. In: Solomon, S. et al., (eds.), Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). IPCC, Intergovernmental Panel on Climate Change, Cambridge University Press, The Edinburgh, Building Shaftesbury Road, Cambridge, UK.

Parfitt et al., 2010, Food waste within food supply chains: quantification and potential for change to 2050, Phil. Trans. R. Soc. B 2010 365, 3065-3081

Schleiss K. und Jungbluth N., 2006, Verwertung der biogenen Abfälle in der Stadt Zürich, Bericht mit besonderer Berücksichtigung der biogenen Abfälle aus Haushalten, ERZ Entsorgung + Recycling Zürich, Hagenholzstrasse 110, Postfach, 8050 Zürich, LCA not published

Sibrián et al., 2006, Estimating household and institutional food wastage and losses in the context of measuring food deprivation and food excess in the total population. Statistics Division, Working Paper Series no: ESS/ESSA/001e, Rome, Italy: FAO.

WRAP, 2007a, "Understanding Food Waste", WRAP The Old Academy, 21 Horse Fair, Banbury OX16 0AH, England

WRAP, 2007b, "New WRAP Research Reveals Extent of Food Waste in the UK", WRAP The Old Academy, 21 Horse Fair, Banbury OX16 0AH, England

High nutritional quality is not associated with low greenhouse gas emissions in self-selected diets of French adults

Florent Vieux[1], Louis-Georges Soler[2], Djilali Touazi[2], Nicole Darmon[1,*]

[1] UMR 1260 INRA, 1062 INSERM, 'Aix-Marseille Univ Nutrition, Obésité et Risque Thrombotique', 13385 Marseille, France
[2] Institut National de Recherche Agronomique UR 1303 ALISS- 65 Boulevard de Brandebourg - 94205 Ivry sur Seine, France
* Corresponding author. E-mail: *nicole.darmon@univ-amu.fr*

ABSTRACT

Food consumption contributes approximately 15-30% of the total greenhouse gas emissions (GHGE) in the developed countries. The aim of the present work was to analyse the relationship between the nutritional quality of self-selected diets and their associated greenhouse-gas emissions (GHGE). Each adult of the INCA2 national dietary survey (n=1918) was aggregated in one of four increasing nutritional quality group and GHGE (in g CO_2e/d) of his/her diet was estimated. High-quality diets contained more plant-based foods, notably fruit and vegetables, and less sweets and salted snacks than low-quality diets. Expressed per 100Kcal or 100g consumed, the highest GHGE was recorded for meat, fish and, eggs food group and the lower for starchy foods. After adjustment for energy intake, high-quality diets had significantly higher GHGE (+4% and +14% in men and women respectively) than low quality diets. This suggests that environmental and nutritional objectives are not necessarily aligned.

Keywords: diet, food, nutrient recommendations, energy density

1. Introduction

Changing food consumption patterns is often considered as an important driver of climate change and a way of reducing the environmental impact of the food sector, which contributes approximately 15-30% of the total greenhouse gas emissions (GHGE) in the developed countries (Garnett, 2008;Kim and Neff, 2009;Kling and Hough, 2010;Tukker et al., 2006). In particular, changing the diets (Carlsson-Kanyama et al., 2003;Coley et al., 1998) through a reduction in meat consumption in high-income countries (associated with a reasonable increase in low-income countries) has been proposed as a good way to reduce the GHGE related to the food sector, whilst simultaneously improving the people's health (McMichael et al., 2007). However, meat, fish and dairy products are unique sources of specific and essential nutrients, and the reduction of their consumption raises a number of nutritional challenges (Millward and Garnett, 2010).

Sustainable diets have been defined by the Food and Agriculture Organization (FAO) as "diets protective and respectful of biodiversity and ecosystems, culturally acceptable, accessible, economically fair and affordable; nutritionally adequate, safe and healthy; while optimizing natural and human resources" (FAO, 2010). Accordingly, the FAO recommends to give due consideration to sustainability when developing food-based dietary guidelines and policies, and acknowledges the need for studies demonstrating the synergies between the different dimensions of sustainability (FAO, 2010). The aim of the present study was therefore to analyse in detail the relationship between the nutritional quality of self-selected diets and their associated greenhouse-gas emissions. To account for the actual diversity of food consumption patterns in France, data from the latest dietary survey conducted among a representative sample of the French adult population were used (AFSSA, 2009). Based on the previously published GHGE of a selection of some highly consumed foods in this population (Vieux et al., 2012), the daily GHGE of each diet was estimated and correlated with the consumption of food-groups and with indicators of nutritional quality, such as the Mean Adequacy Ratio (MAR). Then, to avoid *a priori* assumptions about the food content of high and low nutritional quality diets, a way of classifying them that only relied on their energy density and their nutrient contents was specifically developed for this study, and the daily GHGE of diets of increasing nutritional quality according to this classification were compared.

2. Methods

2.1. Population sample and dietary data

The dietary data used in the present study were derived from the 7-d food records of a nationally representative random sample of adults (n=2624; age > 18 years) participating in the INCA 2 cross-sectional dietary survey ('*Enquête Individuelle et Nationale sur les Consommations Alimentaires*', ('*Individual and National Survey on Food Consumption'*) conducted in 2006-2007 by ANSES (French agency for food, environmental and occupational health safety) (AFSSA, 2009). After the exclusion of under-reporters using standard procedures, the present analysis was conducted on a final sample of 1,918 adults (776 men and 1,142 women). All of the food items declared as consumed by the participants during the survey (n=1314 foods and beverages, including water) were listed in a survey-associated food database giving the nutritional composition of each food item. Total diet weight, total energy intakes, intake of food groups and nutrient intakes were calculated on a daily basis for each participant, based on the list of foods and beverages he/she recorded, and the energy and nutrient content of the items consumed.

2.2. Three indicators of nutritional quality

The Mean Adequacy Ratio (MAR), the Mean Excess Ratio (MER) and the dietary Energy Density (ED) were used as nutritional quality indicators and were estimated without taking into account the nutrients from alcoholic beverages.

The MAR was used as an indicator of good nutritional quality, as it has been repeatedly shown to be positively associated with other indices of dietary quality (Cox et al., 1997;Dubois et al., 2000;Krebs-Smith et al., 1987;Torheim et al., 2004) and with health indicators (Ferland and O'Brien, 2003;Keller et al., 1997). In the present study, the MAR was calculated for the diet of each individual, as the mean percentage of French Recommended Dietary Allowance (Martin, 2001) for 20 key nutrients (namely proteins, fibre, retinol-eq, thiamin, riboflavin, niacin, vitamin B6, folates, vitamin B12, ascorbic acid, vitamin E, vitamin D, calcium, potassium, iron, magnesium, zinc, copper, iodine and selenium).

We developed the MER by analogy with the MAR, and used it as an indicator of bad nutritional quality. The MER was calculated for each diet as the mean daily percent of maximal recommended values (MRV) for three harmful nutrients, namely saturated fatty acids (SFA), sodium and free sugars. The term "free sugars" refers to added sugars plus sugars naturally present in honey, syrups and fruit juices (Joint WHO/FAO expert consultation, 2003).

Dietary ED was used as an indicator of bad nutritional quality because diets with a low energy density have been shown to have a good overall nutritional quality (Ledikwe et al., 2006;Schroder et al., 2008) and because decreasing the energy density of the diet is recommended by several public health authorities to prevent obesity and obesity-associated disease conditions (WHO, ;World Cancer Research Fund International/ Association Institute of Cancer Research, 2007). Dietary ED (in kcal/100g of diet) was calculated by dividing the energy intake by diet weight of each individual. As proposed by Ledikwe et al., (Ledikwe et al., 2005), items typically consumed as beverages, such as milk, juices, and soft drinks, were excluded of the calculation of energy density.

2.3. Four classes of nutritional quality

A method for classifying individuals based on the nutritional quality of their diets was specifically developed for this study. The three indicators of nutritional quality described above were calculated for each diet. Individuals were then ranked according to the values taken by each indicator compared to its observed median in the populations of men and women separately. A class 1 nutritional quality diet was defined as a diet complying with the three following nutritional objectives: having a MAR above the median, a MER below the median and a dietary ED below the median. Diets complying with only 2, 1 or 0 of these objectives were allocated to class 2, class 3 and class 4 nutritional quality, respectively.

2.4. Estimation of diet-related GHGE

As described elsewhere (Supkova et al., 2011;Vieux et al., 2012), the estimation of diet-related GHGE was based on a selection of 73 widely-consumed food items for which a series of assumptions were made. We assumed that the selected food items were all obtained through the conventional and most frequent production and distribution processes in France. The food-related GHGE values covered the stages of agricultural production, processing, packaging and transportation to retail outlets but the stages that occur after purchase (transportation from store to home, storage, preparation and cooking at home, management of end-of-life phases) were not recovered due to a lack of data. Data were expressed as g CO_2 equivalent per 100g of edible portion (g CO_2e /100g). As previously described (Vieux et al., 2012), a Monte-Carlo simulation was run in order to introduce variability of a GHGE food item and a weighting factor was calculated for each representative food item selected within each category to allow us to estimate the GHGE associated with the food category.

2.5. Statistical analysis

The relations between diet-related GHGE and other dietary variables (energy, weight, MAR, MER, DE, food group intakes) were tested using both simple and partial (adjustment for age, gender and energy intakes) Pearson's correlation coefficients. The food group intakes and the diet-related GHGE were estimated among the 4 nutritional classes. Then, comparisons of means among the 4 classes and tests for linear trends were performed using regression analysis for sample survey data for men and women separately. In additional analyses, diet-related GHGE were adjusted for energy intake. An alpha-level of 0.05 was used to determine statistical significance. Statistical analyses were performed using SAS software version 9.2 (SAS institute, Cary, NC).

3. Results

3.1. Correlation between diet-related GHGE and nutritional quality indicators

In simple regression analyses, the MAR ($R = 0.67$, $p < 0.0001$), the MER ($R = 0.80$, $p < 0.0001$), dietary ED ($R = 0.34$, $p < 0.0001$) and diet-related GHGE ($R = 0.79$, $p < 0.0001$) were each positively and significantly correlated with energy intakes. As expected, after energy adjustment, dietary ED was positively correlated with MER and negatively with MAR; higher MAR scores were associated with lower MER scores. After energy-adjustment, diet-related GHGE was positively correlated with MAR and negatively with dietary ED, but no correlation was observed with the MER (data not shown).

3.2. GHGE of food groups and effect of their consumption on total diet-related GHGE

Whatever the calculation basis (per 100g or per 100 kcal of food consumed) the highest GHGE was recorded for the Meat, Fish, poultry and eggs (MFPE) group and the lowest for starchy foods group (Figure 1). Among the MFPE group, meat had the highest GHGE, which was more than 10 times higher, on a weight basis, than that of fruit and vegetables (1387g vs. 121g CO_2e/100g respectively, data not shown). The second lowest GHGE value (after that of starchy foods), was observed for fruit and vegetables when calculated on a weight basis, but for sweets and salted snacks when calculated on a calorie basis. When expressed per 100 kcal, the GHGE of fruit and vegetables was similar to that of dairy products.

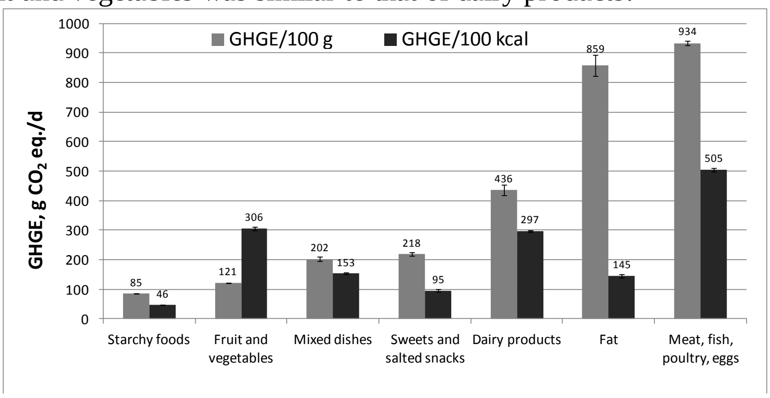


Figure 1. Greenhouse gas emissions (GHGE) related to the consumption of each food group, expressed per 100g and per 100kcal of foods as consumed by adults (n=1918) participating in the INCA2 survey. Values are means. Bars represent the 95% confidence interval.

After adjustment for age, sex and energy intake, a higher consumption of sweets and salted snacks, and of mixed dishes and starchy foods was associated with a lower diet-related GHGE (Figure 2). In contrast, increasing the intake of the other food groups, including that of fruit and vegetables, increased diet-related GHGE. The strongest positive association was seen for the MFPE group (and within that group, for the meat category, data not shown).

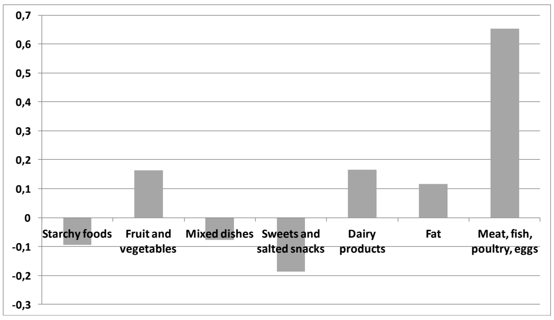


Figure 2. Partial (age, sex and energy-adjusted) Pearson correlations between diet-related greenhouse gas emissions (GHGE, in g CO_2e/d) and the consumption (in g/d) of each food group by adults (n=1918) participating in the INCA2 survey. All coefficients are significantly different from 0 (p< 0.001).

3.3. Food consumption in the four classes of nutritional quality

For both sexes, individuals with high quality diets had higher food intakes and their diets contained significantly more fruit and vegetables and less sweets and salted snacks than low quality diets (data not shown). For both sexes, high quality diets contained significantly more fish and less delicatessen (data not shown) than low quality diets, but the quantity of meat did not differ between nutritional quality classes (class 1 diets contained 69 and 51 g/d of meat, in men and women respectively, data not shown). For women, high quality diets also contained more poultry and eggs than low quality diets, so that the total intake of the MFPE group increased with increasing nutritional quality for them.

3.4. Diet related GHGE in the four classes of nutritional quality

The crude and adjusted values of daily diet-related GHGE in the four classes of nutritional quality are shown Figure 4. Without adjustment (panel A), GHGE was not significantly different between the four classes for men (p=0.27) and it was greater in the highest nutritional quality class for women (p=0.0021). After adjustment for energy intakes (panel B), diets with a high nutritional quality tend to be associated with higher GHGE values than diets with a lower nutritional quality (p<0.0001 whatever the gender).

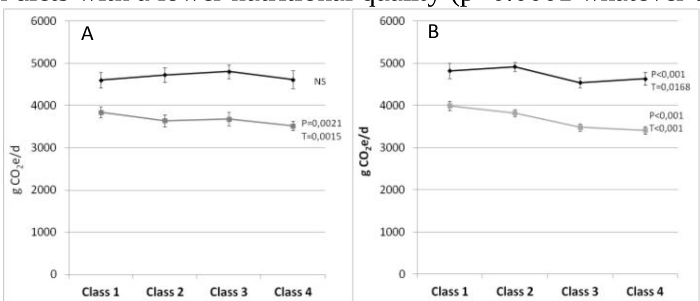


Figure 4. Greenhouse gas emissions (GHGE, in g CO_2e/d) associated with the diets of adults participating in the INCA2 survey, according to the nutritional quality of their diets. Crude values (A) and values adjusted for total energy intakes (B). Values are means. Bars represent the 95% confidence interval. P=global p-value, T= test for linear trend p-value. NS=non significant.

4. Discussion

The present study showed that, at a given level of energy intake, diet-related GHGE tend to be positively associated with nutritional quality: i) The more nutrient-dense diets (high MAR) had a high GHGE whereas the more energy-dense diets (high ED) had a low GHGE; ii), the consumption of sweets and salted snacks was negatively associated with diet-related GHGE whereas the consumption of fruit and vegetables was positively associated with it; iii) when diets were classified according to their overall nutritional quality, high-quality diets tend to have the highest GHGE, although they contained more plant-based products than low-quality diets.

Compared with other international studies, our approach was original in two ways: firstly because we analysed diets spontaneously consumed by individuals (and could therefore observe a wide and "natural" variety of realistic food choices), and secondly because nutritional quality was introduced in our analyses and was defined by nutrient-based indicators instead of preconceived views on the food composition of balanced diets. In contrast, previous studies on the environmental impact of food consumption were based either on stereotyped meals (Reijnders and Soret, 2003) and diets (Baroni et al., 2007;Carlsson-Kanyama et al., 2003;Carlsson-Kanyama and Gonzalez, 2009) or on the comparison between average and theoretical diets (Eshel and Martin, 2006;Macdiarmid et al., 2011;Risku-Norja et al., 2008;Wallén et al., 2004). Only one of them (Macdiarmid et al., 2011) precisely controlled the nutrient content of the theoretical diets designed, and the conclusion was that "it is possible to create a realistic and affordable diet that meets dietary requirements for health and a 25% reduction in GHGE". However the "realism" of such a diet was doubtful because it was based on arbitrary decisions on which changes are culturally and socially acceptable by people, in particular as regards reducing the consumption of meat and dairy products.

Altogether, our results therefore seem to contradict the widely-accepted view that diets that are good for health are necessarily good for the planet. This notion has progressively emerged, based on the fact that plant-based products have lower environmental impact than animal products, and on the belief that vegetarian diets are necessarily healthy. However, the present results showed that plant-based products may have a similar GHGE than animal products when expressed on a per calorie basis (for instance fruit and vegetable and dairy products in the present study, Figure 1).

This study has limitations. First, diet-related GHGE was estimated based on a limited number of items. However those foods were the most frequently consumed in the studied population, and our estimate of the daily GHGE was of similar magnitude to that estimated in studies conducted in other European populations (Coley et al., 1998;FAO, 2010). Secondly, we used GHGE as the sole environmental criterion. We did not consider the entire life cycle of the food products (only up to retail outlets), and we focused only on conventional production and distribution processes (organic and local production and/or distribution were not considered). In future studies, other environmental criteria, such as water and land use or biodiversity, must also be considered, as well as the impact of alternative production and distribution schemes, and of consumer behaviour (transport, storage, cooking…). Thirdly, the method used to classify diets according to their nutritional quality was not previously published. However, our aim was to classify existing diets based only on their nutrient contents and, to our knowledge, there is no published approach allowing such classification. It should be noted that our method identified diets rich in fruit and vegetables with moderate amounts of a variety of animal products and limited amounts of sweets and salted snacks as being of the highest nutritional quality, which is in accordance with the basic principles of dietary guidelines (USDA, 2011;WHO Regional Office for Europe, 1998).

5. Conclusion

In the present study, the healthiness of diets, whether estimated by a high intake of fruit and vegetables, a low intake of sweets and salted snacks, a high nutrient density, a low energy density, or a more comprehensive definition of nutritional quality (e.g. belonging to class1) was associated with a slightly but significantly higher carbon impact. This suggests that environmental and nutritional objectives are not necessarily aligned. The compatibility of those two dimensions of sustainability should be further examined, using more comprehensive and detailed indicators of the environmental impact of food consumption.

6. References

AFSSA. Summary of the report of the 2006/2007 Individual and National Survey on Food Consumption 2 (INCA 2). http://www.afssa.fr/Documents/PASER-Sy-INCA2EN.pdf.2009.

Baroni L, Cenci L, Tettamanti M, Berati M. Evaluating the environmental impact of various dietary patterns combined with different food production systems. Eur J Clin Nutr. 2007;61(2):279-86.

Carlsson-Kanyama A, Pipping Ekström M, Shanahan H. Food and life cycle energy inputs: consequences of diet and ways to increase efficiency. Ecological Economics. 2003;44(2-3):293-307.

Carlsson-Kanyama A, Gonzalez AD. Potential contributions of food consumption patterns to climate change. Am J Clin Nutr. 2009;89(5):1704S-9S.

Coley AD, Goodliffe E, Macdiarmid J. The embodied energy of food: the role of diet. Energy Policy. 1998;26(6):455-9.

Cox DR, Skinner JD, Carruth BR, Moran J, 3rd, Houck KS. A Food Variety Index for Toddlers (VIT): development and application. J Am Diet Assoc. 1997;97(12):1382-6; quiz 7-8.

Dubois L, Girard M, Bergeron N. The choice of a diet quality indicator to evaluate the nutritional health of populations. Public Health Nutr. 2000;3(3):357-65.

Eshel G, Martin P. Diet, energy, and golbal warming. Earth interactions. 2006;10(9).

FAO. Definition of sustainable diets. International scientif symposium "Biodiversity and sustainable diets United againts hunger". FAO Headquarters, Rome2010.

Ferland S, O'Brien HT. Maternal dietary intake and pregnancy outcome. J Reprod Med. 2003;48(2):86-94.

Garnett T. Cooking up a storm: Food, greenhouse gas emissions and our changing climate. Food Climate Research Network, Centre for Environmental Strategy, University of Surrey. 2008.

Joint WHO/FAO expert consultation. Diet, nutrition and the prevention of chronic diseases. World Health Organ Tech Rep Ser. 2003;916:i-viii, 1-149, backcover.

Keller HH, Ostbye T, Bright-See E. Predictors of dietary intake in Ontario seniors. Can J Public Health. 1997;88(5):305-9.

Kim B, Neff R. Measurement and communication of greenhouse gas emissions from U.S. food consumption via carbon calculators. Ecological Economics. 2009;69(1):186-96.

Kling MM, Hough IJ. The American Carbon Footprint: Understanding your food's impact on climate change. Brighter Planet, Inc. 2010.

Krebs-Smith SM, Smiciklas-Wright H, Guthrie HA, Krebs-Smith J. The effects of variety in food choices on dietary quality. J Am Diet Assoc. 1987;87(7):897-903.

Ledikwe JH, Blanck HM, Khan LK, Serdula MK, Seymour JD, Tohill BC, et al., Dietary energy density determined by eight calculation methods in a nationally representative United States population. J Nutr. 2005;135(2):273-8.

Ledikwe JH, Blanck HM, Khan LK, Serdula MK, Seymour JD, Tohill BC, et al., Low-energy-density diets are associated with high diet quality in adults in the United States. J Am Diet Assoc. 2006;106(8):1172-80.

Macdiarmid J, Kyle J, Horgan G, Loe J, Fyfe C, Johnstone A, et al., Livewell: a balance of healthy and sustainable food choices. WWF-UK and the Rowett Institute http://assetswwforguk/downloads/livewell_report_correctedpdf2011.

Martin A. The "apports nutritionnels conseilles (ANC)" for the French population. Reprod Nutr Dev. 2001;41(2):119-28.

McMichael AJ, Powles JW, Butler CD, Uauy R. Food, livestock production, energy, climate change, and health. Lancet. 2007;370(9594):1253-63.

Millward DJ, Garnett T. Plenary Lecture 3: Food and the planet: nutritional dilemmas of greenhouse gas emission reductions through reduced intakes of meat and dairy foods. Proc Nutr Soc. 2010;69(1):103-18.

Reijnders L, Soret S. Quantification of the environmental impact of different dietary protein choices. Am J Clin Nutr. 2003;78(3 Suppl):664S-8S.

Risku-Norja H, Hietala R, Virtanen H, Ketomaki H, Helenius J. Localisation of primary food production in Finland: production potential and environmental impacts of food consumption patterns. Agric Food Sci. 2008;17:127-45.

Schroder H, Vila J, Marrugat J, Covas MI. Low energy density diets are associated with favorable nutrient intake profile and adequacy in free-living elderly men and women. J Nutr. 2008;138(8):1476-81.

Supkova M, Darmon N, Vieux F, Touazi D, Redlingshoefer B, Russel M. Etude de cas. Impact carbone de régimes alimentaires différenciés selon leur qualité nutritionnelle: une étude basée sur des données Françaises. French (Carbon impact of food and nutritional quality of food choices in France. A case study). INRA ADEME; 2011.

Torheim LE, Ouattara F, Diarra MM, Thiam FD, Barikmo I, Hatloy A, et al., Nutrient adequacy and dietary diversity in rural Mali: association and determinants. Eur J Clin Nutr. 2004;58(4):594-604.

Tukker A, Huppes G, Guinee J, Heijungs R, de Koning A, van Oers L, et al., Environmental Impact of Products (EIPRO): Analysis of the Life Cycle Environmental Impacts Related to the Total Final Consumption of the EU 25. European Commission Technical Report2006.

USDA (United States Department of Agriculture). http://www.choosemyplate.gov/. 2011.

Vieux F, Darmon N, Touazi D, Soler LG. Greenhouse gas emissions of self-selected individual diets in France: changing the diet structure or consuming less? Ecological Economics. 2012;75:91-101.

Wallén A, Brandt N, Wennersten R. Does the Swedish consumer's choice of food influence greenhouse gas emissions? Environmental Science and Policy. 2004;7(6):525-35.

WHO. Diet, nutrition and the prevention of excess weight gain and obesity. Report of a joint WHO/FAO expert consultation WHO, Geneva 2003; Technical report, series 916.

WHO Regional Office for Europe. CINDI dietary guide. 1998;http://www.euro.who.int/__data/assets/pdf_file/0010/119926/E70041.pdf.

World Cancer Research Fund International/ Association Institute of Cancer Research. Food, Nutrition, Physical Activity, and the Prevention of Cancer: a Global Perspective. http://wwwwcrforg/. 2007.

Application of new UNEP/SETAC life cycle initiative methods for land use impact assessment. Land use impacts of margarine

Llorenç Milà i Canals*, Giles Rigarlsford, Sarah Sim

Safety and Environmental Assurance Centre, Unilever R&D, Colworth Science Park, Sharnbrook, Bedfordshire, MK44 1LQ, UK
* Corresponding author. E-mail: *Llorenc.Mila-i-Canals@Unilever.com*

ABSTRACT
New characterisation factors (CF) for land use and land use change (LUC) impacts relating to biodiversity and ecosystem services developed recently within the UNEP/SETAC Life Cycle initiative have been applied to a case study of margarine (Milà i Canals et al., 2012). The new land use impact assessment methods applied help to identify hotspots in the life cycle of margarines, with different proportions and sources of vegetable oils. The specific impacts of each vegetable oil are determined mainly by the yield (land occupation), but also by the type of agriculture (annual vs. permanent crops) and the sourcing location (sensitivity of biomes and occurrence of land use change). Outstanding key challenges to assess land use impacts in LCA include the quantification of LUC and its allocation to specific crops / products; determination of sourcing regions for globalised supply chains; and the choice of reference situation to assess land use impacts.

Keywords: land use impacts, margarine, Life Cycle Impact Assessment, biodiversity, ecosystem services

1. Introduction

New characterisation factors (CF) for the impacts from land use (LU, also called occupation) and land use change (LUC, also called transformation) on biodiversity and ecosystem services have recently been published under the UNEP/SETAC Life Cycle initiative (Koellner et al., 2012a); this paper assesses the application of these new CFs to LU- and LUC-related impact categories in a case study of margarine (Milà i Canals et al., 2012). The specific goals of the case study are to describe and estimate the size of the environmental impacts associated with the cradle-to-gate production of margarine and to understand the ease of use of the new CF. This case study uses as its basis a recently published study on margarine (Nilsson et al., 2010).

2. Methods

The methodological approach in this case study is a descriptive (attributional) LCA (Nilsson et al., 2010). The functional unit of the study is 500g of packaged margarine used as a spread in the UK and Germany representing a low fat and high fat content margarine product respectively. Further description of how the occupation flows were quantified is provided in Milà i Canals et al., (2012), as well as the adaptation of land use flows in the background databases. The quantification of land transformation flows in the relevant countries for the main crops used in the margarine recipes is further explained in section 2.1 below. The land occupation and transformation flows identified were then characterised with the characterisation factors (CF) recommended by the Life Cycle Initiative project, as follows:

- For the Biodiversity Damage Potential (BDP) the approach and CF offered by de Baan et al., (2012) were used. Average world CF were used for those biomes not covered in de Baan et al., (2012).
- For Climate Regulation Potential (CRP), the approach suggested by Müller-Wenk and Brandão (2010) was used.
- For Biotic Production Potential (BPP), the approach and CF offered by Brandão and Milà i Canals (2012) were used.
- For impacts on Ecosystem Services, other than CRP and BPP, the approach and CF proposed by Saad et al., (2012) were used. These include impacts on: Freshwater Regulation Potential (FWRP); Erosion Regulation Potential (ERP); and Water Purification Potential (WPP) assessed here by two indicators related to Physico-Chemical Filtration (WPP-PCF) and Mechanical Filtration (WPP-MF). The results on FWRP and ERP are not discussed in this paper.

Specific assumptions in the application of each impact category are discussed in Milà i Canals et al., (2012).

2.1. Land transformation linked to agricultural stages

As shown in Figure 1, a three step approach was used to determine whether a crop grown in a specific country was potentially related to any land transformation (LUC) in that country, and what transformations were involved if any. A 20 year time period was considered as this is often recommended for the allocation of impacts of land use change (see e.g. Koellner et al., 2012a; Flynn et al., 2012). Because in this case the average LUC in the whole country rather than a specific area (e.g. plantation) was assessed, there was no need for allocating it to the first 20 years of land use (as suggested in BSI, 2008; Koellner et al., 2012a; Flynn et al., 2012). In order to smooth out short-term fluctuations in land use, 5-year averages were used. FAOSTAT (2011) data were used to perform this analysis.

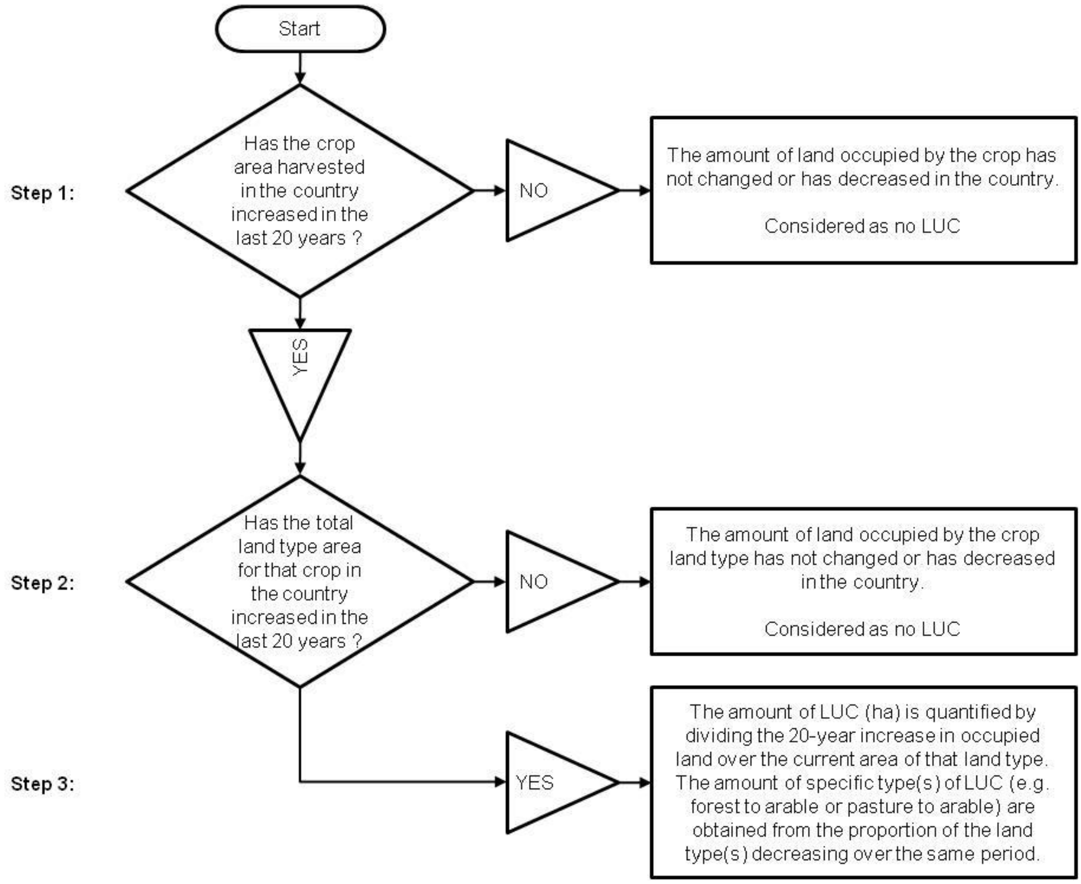


Figure 1. Decision tree to determine the existence and magnitude of land use change for the studied crops in the sourcing countries (Milà i Canals et al., 2012).

This approach is consistent with the recommended approach in the PAS 2050 (BSI, 2011) where the country of production is known but the previous land use is not known (section 5.6.2, point b). BSI (2011) suggest excluding indirect land use changes (ILUC); however, it must be noted that this approach actually does include ILUC occurring within the producing country. ILUC caused by displacement of crops to be grown in other countries, on the other hand, is not included. This is a limitation of an attributional-focused approach; see e.g. Kløverpris et al., (2007) or Brandão (2012) for approaches to deal with ILUC.

3. Results

Figures 2a-g provide the contributions per functional unit of the different ingredients and life cycle stages of the two margarines for the seven land use-related impact categories. For most impact categories (except FWRP, not shown here) the UK margarine with a 38% fat content shows larger total impacts than the German margarine with the higher 70% fat content. This is because the UK margarine contains a higher proportion of oils from low-yielding crops such as sunflower (yield: 1.5 t/ha). Sunflower growing dominates the impact results for the UK margarine (where it represents about 25% of the ingredients) and it also has a significant contribution towards the impacts of the German margarine in which it is only 3.5% of the ingredients. In comparison the impacts for rapeseed are generally lower even though it represents 36% of the German margarine. This can be explained by the higher yield for rapeseed (4.2 t/ha) compared to sunflower. Palm oil, which makes up ca. 26% of the German recipe, has a relatively low contribution to all the land use impact categories, even though significant LUC of 435 m^2/ ha*year of tropical ecosystems with high biodiversity and carbon values was attributed to this crop. This can be explained by the lower impacts associated to permanent agricultural systems such as plantations for most impact categories relative to annual crops.

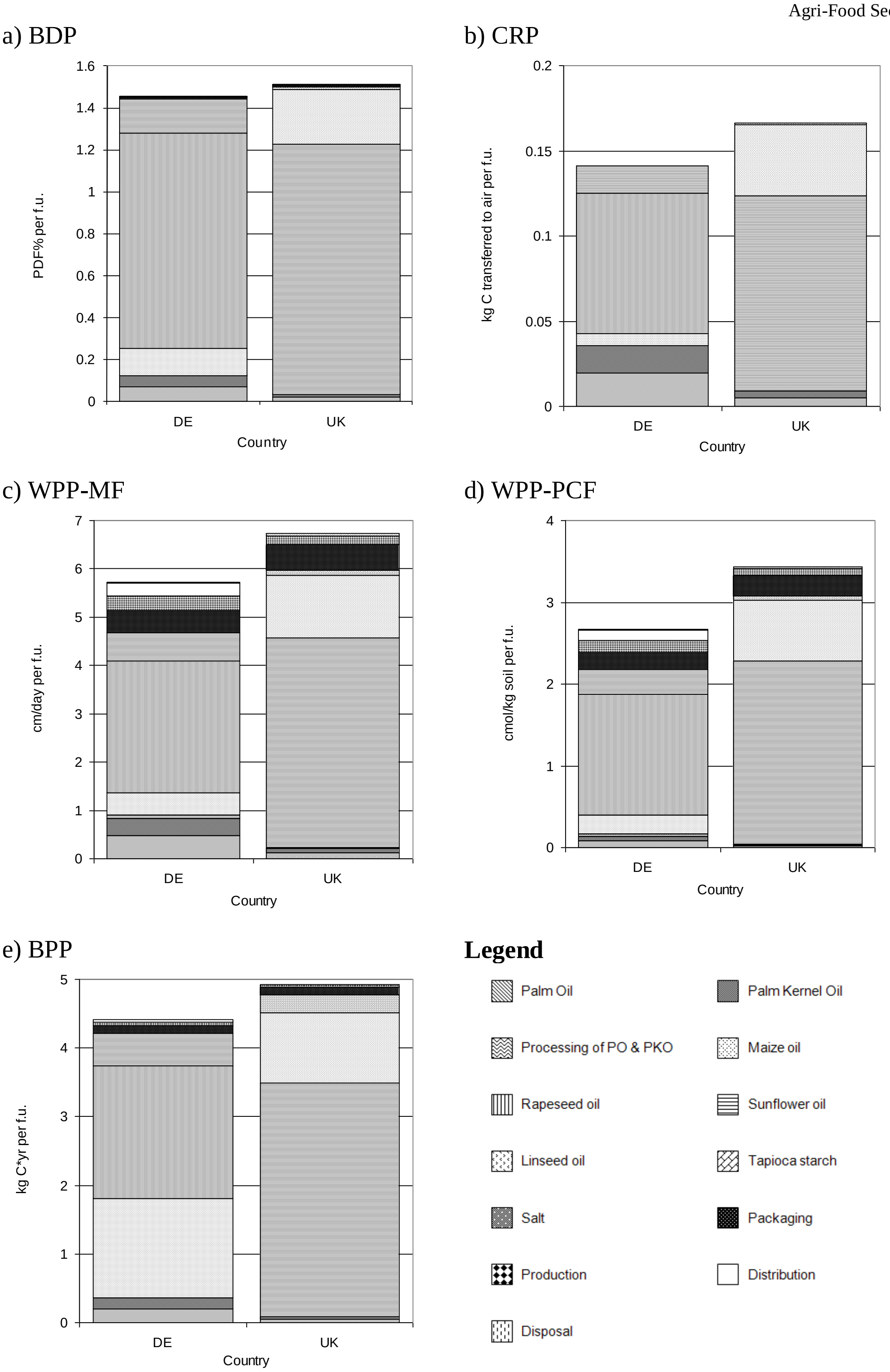


Figure 2. Contributions of the different ingredients and life cycle stages to the land use impact categories: a) BDP, Biodiversity Damage Potential; b) CRP, Climate Regulation Potential; c) WPP-MF, Water Purification Potential through Mechanical Filtration; d) WPP-PCF, Water Purification Potential through Physico-Chemical Filtration; e) BPP, Biotic Production Potential. All impacts expressed per functional unit (f.u.): 500g tub of margarine to be used as spread (Milà i Canals et al., 2012).

In conclusion, the type of oil, not only the total fat content, determines the overall impact and this is mainly due to the differing yields of the various oil crops and hence the land occupation level. In addition, the production system of the various oil crops (annual vs. permanent crops) and the sourcing region (biome) have a significant effect on the contributions to the impact categories, as explained below.

The impacts on biodiversity (BDP) and climate regulation (CRP) (Figures 2a-b) are the largest for the UK margarine because of the contribution from low oil-yielding crops (sunflower; linseed). This is in spite of the fact that the total fat amount of these two oils in the UK recipe is about half the amount of rapeseed oil and palm oil in the German recipe.

The water purification potential (WPP) impact categories (WPP-MF and WPP-PCF, Figures 2c-d) deserve some special attention because they indicate a very significant contribution from land transformation flows linked to non-agricultural or 'sealed' land use (e.g. industrial buildings; transport infrastructure) associated with the packaging's life cycle, product manufacturing and road distribution. This is because the CF for land transformation to sealed land flows are 3-5 orders of magnitude higher than for land transformation to agricultural land (Saad et al., 2012).

Finally, the impact profile for biotic production (BPP, Figure 2e) follows a very similar pattern to the WPP impact profile, but with smaller contributions from palm oil due to the same soil organic carbon being considered for forests and permanent crops (Brandão and Milà i Canals, 2012). In BPP there are also relevant contributions associated with the packaging component of the product due to the occupation of sealed land rather than transformation flows as in the case of the two WPP impacts (see above).

4. Discussion

In terms of the five new impact categories evaluated in this paper, the results of the case study suggest that their impact profiles are largely similar. This is because all of them are actually determined to a large extent by land occupation. The land use types distinguished so far (mainly at the first level of classification as suggested by Koellner et al., 2012b) are useful in highlighting the likely hotspots in the life cycle. In this case study based on a food product most impacts were dominated by the agricultural stages, but the importance of non-agricultural or 'sealed' land uses was shown for some impact categories. A clear need identified for further refinement is in the description of the types of agricultural production. Perennial crops, such as oil palm plantations, are likely to have significantly different impacts on several impact categories when compared to annual arable crops. Therefore, it is not sufficient to provide CF only for "agricultural land" at the first level of land use classification.

Spatial differentiation at the level of biomes has shown to be relevant in this case study. It remains to be studied whether finer levels of bio-geographical differentiation would provide more informational value to such studies but this would need to be balanced against its practical feasibility as discussed below.

Land transformation (land use change, LUC) flows do not have a very significant effect on the impact results, except for those impact categories where CF for transformation flows are significantly larger than for the corresponding land occupation flows. This is due to several factors, including the allocation of LUC over 20 years following the transformation; use of the same reference (potential natural vegetation); and limited (and uncertain) regeneration times and modelling for LUC impacts. A calculation procedure has been suggested to estimate direct land use change from FAO statistics, which is consistent and easy to use for any crop in the world. As noted above, though, this approach ignores indirect LUC caused by displaced production of crops outside the country being studied. The fact that LUC has such a small effect on the final results is somehow contrary to current opinion and policy, which places a significant focus on LUC-derived impacts (e.g. on GHG emissions and biodiversity loss from deforestation). This limitation of the UNEP/SETAC LCI framework for modelling impacts from land use change is related to the assumption of maintaining the land quality in terms of an idealistic potential quality which may never be reached again in reality. In this sense, the results of the impact assessment need to be interpreted as a view of the differences in biodiversity or ecosystem services that are being maintained with respect to an ideal or theoretical potential rather than a description of the actual change in land quality. In this context the new land use impact categories inform of potential impacts on biodiversity and ecosystem services as well as the "opportunity cost" impacts of not letting land regenerate.

The challenges of using the new CF relate to the existing information in LCI databases and in product supply chains. Significant time had to be invested in updating background processes with relevant land use information, even though no attempt was made to provide spatial differentiation information to the background data because at the level at which such data are aggregated it would have been impossible. This illustrates the need for greater consistency or standardisation in how LU/LUC flows are considered in LCI databases. In the case of information describing the product's supply chain, this is not always available. For margarine, this is exemplified by the fact that vegetable oils are often traded as commodity products. This means that the origin of the oils may not always be known, particularly at a country level, thus illustrating a potential disparity or complexity in the level of detail required for the application of these new CF and the information currently available within companies and supply chains.

5. Conclusion

One key learning from this study is that occupation seems to be the key driver for land use impacts, more than transformation. Here, the effects of differing crop yields are the main driver for occupation and hence tend to dominate the results. Furthermore as crop yield data are associated with large variability, it will also be a source of variability / uncertainty for land use impacts. Considering the driver role of land occupation, the conclusion of Nilsson et al., (2010) that margarine has smaller environmental impacts than butter remains valid, since butter production required double the land occupation of margarine. However, it would be premature to suggest that we expect occupation to be always the key driver for impacts, and more case studies should be carried out in order to test this.

The conceptual approach followed to assess land use impacts relates to a theoretical potential reference (i.e. original biome prior to man-made intervention) and therefore it tends to focus the impact assessment away from the obvious sources of actual impacts such as recent land use change. In some decision contexts it may be helpful to explore the use of alternative ecological reference points such as a current reference state (option 3 suggested in Koellner et al., 2012a) or the alternative most probable land use.

In addition, large uncertainties remain around the assessment of impacts from land use change (LUC). These arise already in the inventory stage, in the allocation of LUC to specific crops (e.g. over a certain “amortisation” period for land) and in the inclusion or exclusion of indirect LUC from the analysis. Also in the impact assessment stage there is a large uncertainty related to the relaxation times considered to quantify the CF.

More applications are thus needed that test the limits and potential flaws of the methods and theoretical approaches used in order to gain confidence in the helpfulness of the results as support for various types of decision making. Specifically, research is recommended into allocation of LUC, relaxation times for LUC derived impacts, and effects of different references on the results.

6. References

de Baan, L., Alkemade, R., Koellner, T. 2012. Land use impacts on biodiversity in LCA: a global approach. Int. J. Life Cycle Assess. on-line first DOI:10.1007/s11367-012-0412-0.

Brandão, M. 2012. Food, feed, fuel, timber or carbon sink? Towards sustainable land use - a consequential life-cycle approach. PhD Thesis, University of Surrey, Guildford, UK.

Brandão, M., Milà i Canals, L. 2012. Global characterisation factors to assess land use impacts on Biotic Production. Int. J. Life Cycle Assess. on-line first DOI:10.1007/s11367-012-0381-3.

BSI. 2011. PAS2050:2011 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services, British Standards Institution.

FAOSTAT. 2011. Crop production data 1988-2008 from http://faostat.fao.org/site/567/default.aspx#ancor Accessed 18 March 2011.

Flynn, H.C., Milà i Canals, L., Keller, E., King, H., Sim, S., Hastings, A., Wang, S., Smith, P. 2012. Quantifying global greenhouse gas emissions from land use change for crop production. Global Change Biol. 18 (5) 1622–1635.

Kløverpris, J., Wenzel, H., Nielsen, P.H. 2007. Life cycle inventory modelling of land use induced by crop consumption. Part 1: Conceptual analysis and methodological proposal. Int. J. Life Cycle Assess. 13(1) 13-21.

Koellner, T., de Baan, L., Brandão, M., Milà i Canals, L., Civit, B., Margni, M., Saad, R., Maia de Souza, D., Beck, T. Müller-Wenk, R. 2012a. UNEP-SETAC Guideline on Global Land Use Impact Assessment on Biodiversity and Ecosystem Services in LCA. Int. J. Life Cycle Assess. On-line first.

Koellner, T., de Baan, L., Brandão, M., Milà i Canals, L., Civit, B., Goedkoop, M., Margni, M., Weidema, B., Wittstock, B., Beck, T., Müller-Wenk, R. 2012b. Principles for Life Cycle Inventories of land use on a global scale. Int. J. Life Cycle Assess. On-line first DOI:10.1007/s11367-012-0392-0

Milà i Canals, L., Rigarlsford, G., Sim, S. 2012. Land use impact assessment of Margarine. Int. J. Life Cycle Assess. On-line first DOI:10.1007/s11367-012-0380-4.

Müller-Wenk, R., Brandão, M. 2010. Climatic impact of land use in LCA—carbon transfers between vegetation/soil and air. Int. J. Life Cycle Assess. 15 172-182.

Nilsson, K., Flysjö, A., Davis, J., Sim, S., Unger, N., Bell, S. 2010. Comparative life cycle assessment of margarine and butter consumed in the UK, Germany and France. Int. J. Life Cycle Assess. 15:916-926.

Saad, R., Koellner, T., Margni, M. 2012. Land use impacts on freshwater regulation, erosion regulation and water purification: a spatial approach for a global scale. Int. J. Life Cycle Assess. On-line first.

Development of an integrated indicator for land use based on the economic value of ecosystem services

Viêt Cao[1,2,*], Manuele Margni[2], Basil D. Favis[3], Louise Deschênes[1]

[1] CIRAIG, Department of Chemical Engineering, Polytechnique Montréal, P.O. Box 6079, Station Centre-ville, Montréal, Québec H3C 3A7, Canada[1]
[2] CIRAIG, Department of Mathematical and Industrial Engineering, Polytechnique Montréal, P.O. Box 6079, Station Centre-ville, Montréal, Québec H3C 3A7, Canada
[3] CRÉPEC, Department of Chemical Engineering, Polytechnique Montréal, P.O. Box 6079, Station Centre-ville, Montréal, Québec H3C 3A7, Canada
[*] Corresponding author. E-mail: *viet.cao@polymtl.ca*

ABSTRACT

Soils are one of Earth's essential natural resources, supporting nearly all terrestrial life. In life cycle impact assessment (LCIA), potential impacts due to land use are calculated as the product of surface occupied (or transformed), occupation (or transformation) time and a parameter describing the land quality loss (ΔQ) (Mila iCanals *et al*., 2007). In current LCIA methodologies, the only operational endpoint level indicator for the land quality loss is solely related to terrestrial biodiversity (PDF.m^2.year, PDF being Potential Disappeared Fraction of species) and is not representative of all impacts that originate from land use as shown by a recent project named LULCIA (2008-2012), conducted under the aegis of the UNEP/SETAC Life Cycle Initiative (Koellner *et al*., 2012; de Baan *et al*., 2012; Saad *et al*., 2011; Brandão and Mila iCanals, 2012; Müller-Wenk and Brandão, 2010).

This project expanded the scope of land use assessment, going beyond the biodiversity assessment. This method relates land use to six additional indicators: biotic production potential (BPP), erosion regulation potential (ERP), fresh water regulation potential (FWRP), (mechanical and physico-chemical) water purification potential (WPP) and carbon sequestration potential (CSP), which represent provision and regulation services from ecosystems, as defined in the Millennium Ecosystem Assessment (MEA) (2005). Although the LCIA methodology becomes more comprehensive for relevant pathways linked to land use, this development potentially reduces the capacity of LCA as a decision support system, providing seven midpoints for the land use impact category alone.

This project aims to develop a new method to value the reduction of ecosystem services provided to human society. The method consists in converting the above mentioned midpoint indicators in monetary terms, using economic valuation of the reduction of a given ecosystem service. BPP is estimated with productivity loss while CSP thanks to carbon social cost: less sequestration by soils is equivalent to emission. The other regulation services are estimated through current compensation costs, as they are considered essential (conservative approach).

This method is applied on a case study, the comparative LCA of bio-based polymers. Results show that impact scores are not only influenced by the bio-geographical variability of systems under study (e.g. crop yield in the inventory flow, the land location for the impact characterisation), but also by the socio-economical availability and typology of the compensation systems taken into account. Uncertainties and economic valuation assumptions are further discussed in the paper.

Overall this work shows the feasibility to translate all the midpoints indicators proposed by the LULCIA land use impact assessment framework into economic values, bringing a new level of interpretation for the decision maker. The converted indicators can be summed into an integrated indicator expressing potential impacts and they must be interpreted as the loss of natural (capital of) ecosystem services. It also potentially allows LCA to assess other impacts related to land use, such as aesthetics and recreational aspects (as they are conceptualised as cultural services in the MEA framework).

Keywords: land use, integrated indicator, ecosystem services, economic valuation, impact assessment

1. Introduction

In life cycle impact assessment (LCIA), midpoint potential impacts due to land use are calculated as the product of surface occupied (or transformed), occupation (or transformation) time and a parameter describing the land quality loss (ΔQ) (Mila i Canals et al., 2007). So far, this latter is solely related to terrestrial biodiversity (PDF.m^2.year, PDF being Potential Disappeared Fraction of species), which is certainly not representative of all impacts caused by human interventions, that originate from land use as shown by a recent project named LULCIA (2008-2012), conducted under the aegis of the UNEP/SETAC Life Cycle Initiative (Koellner et al., 2012; de Baan et al., 2012; Saad et al., 2011; Brandão and Mila iCanals, 2012; Müller-Wenk and Brandão, 2010) and also partly assessed by the European Commission JRC ILCD handbook (2010).

Soil quality is both an important and difficult ecosystem component to assess, because of the variety of both soil quality definitions and approaches possible (Garrigues et al., 2011). The “quest” for a synthetic indicator is even more difficult with both the different intended meaning and public possible.

This method relates land use to six new indicators in addition to biodiversity: biotic production (BPP), erosion regulation (ERP), fresh water regulation (FWRP), mechanical and physicochemical water purification (MWPP and PCWPP) and carbon sequestration (CSP) potentials, which represent provision and regulation ecosystem services, as defined in the Millennium Ecosystem Assessment (2005). Ecosystem service is a relative new concept, which makes the link between ecological functions and the service they provide to by humans. Bridging environmental, economic sciences and decision-making policies, they have become a very dynamic area of research, particularly in ecological economics (National Research Council, 2004).

On one hand, the LCIA methodology becomes more comprehensive in regards to relevant pathways linked to land use, but on the other hand, this development can potentially reduce the capacity of LCA to be used as a decision support system, as it can increase up to seven the number of land use indicators required.

This project aims therefore to develop a method to convert impact indicators in monetary value using economic valuation as a common thread to normalise (and possibly aggregate) the new midpoint indicators to a single area of protection representing ecosystem services loss.

2. Methods

2.1. Economic valuation: conversion from functional indicators to loss of natural capital of ecosystem services

The method consists of converting each midpoint i from physical to monetary units:

$$CF'_i = CF_i \cdot MF \cdot (1\text{-}AC)$$

with CF' being the caracterisation factor expressing the loss of natural capital of ecosystem services in $/(m^2.year), CF the regular characterisation factor in physical unit/(m^2.year), MF the monetary factor in $/physical unit and AC the adaptation capacity (dimensionless). The monetary factor describes the cost of the compensation technology while (1-AC) describes the fraction of the ecosystem service that cannot be compensated by humans and thus leads to the exposure to the potential impact. AC is determined by linear correlations with the country gross national product (GDP)..If a country has the capacity to adapt, the value of AC is 1, and there is no impact (Boulay et al., 2011).

The monetary factor values for BPP and CSP are respectively estimated with productivity loss using FAO data (FAOSTAT, 2012) and social cost of carbon (Ackerman and Stanton, 2010). The monetary factors for the other regulation services are estimated through current and potential compensation costs, as they are considered essential and to be replaced in the very short term. It is assumed that current productions systems and their operating mode (including their relationship with ecosystem, ecosystem services and benefits) are sought to be kept in their current state (conservative approach).

Mechanical and physicochemical water purification potentials (WPPs) correspond to the natural equivalent of primary and secondary & tertiary water treatment, respectively. Current world water qualities from Boulay et al., (2011) are extrapolated without this natural filtration to identify the potential compensation technology required. The corresponding costs are calculated with the Water Treatment Estimation Routine (WaTER) from US-EPA (1999). The economic values for FWRP are estimated using urban water supply prices (UNESCO, 2009) and the values for ERP are based on the World Overview of Conservative Approaches and Technologies database (Centre for Development and Environment et al., 2012).

The values calculated aim to estimate and to represent the natural capital provided by ecosystem services by assessing the estimated cost if humans had to produce them by themselves. Corresponding to the "use value" in the total economic value theory (Freeman, 2003), it is a *low-estimate* of the ecosystem services value.

2.2. Spatial considerations and final converted characterisation factor (CFs') calculation

a. Since LCA assesses spatially-global product systems, physical midpoint CFs were developed in the LULCIA project at the world scale. Because land use is by nature a local impact category, spatial variability prevails and this is why they are also regionalised. Each midpoint has a different spatial variability (or resolution, see table 1). In the same way, spatial-dependant economic values were used for the monetary factors and the local availability of the compensation systems is taken into account when possible.

b. Each of the six midpoints was developed for different land type uses (or land covers, representing human activities): grasslands, forests, permanent crops, farmlands, fallow grounds, artificial, urban areas, water related areas and "others". A harmonisation was made to create common land covers and geographic information system (GIS) software was used to design common biogeographical areas (see table 1).

c. Finally, all the converted midpoints indicators (in 2.1) of the same land cover were summed into a single indicator standing for the potential damage costs due to the loss of natural capital of ecosystem services that originates from the considered land use cover.

Table 1 – Land use midpoints (UNEP/SETAC Life Cycle Initiative) and spatial details.

Midpoint	Author	Spatial resolution	Delineation main drivers	Source
ERP MWPP PCWPP FWRP	Saad (2010)	36 lifezones	Annual precipitation, annual biotemperature[1], potential evapotranspiration	Holdridge lifezones (Holdridge, 1947)
BPP	Brandão and Mila iCanals (2011	13 climate zones	Mean annual temperature & precipitation, potential evapotranspiration, soil types	IPCC climate zones (IPCC, 2009)
CSP	Müller-Wenk and Brandão (2010)	14 biomes or 867 ecozones	Several, including species endemism & richness, rarity of habitat type	WWF biomes (Olson et al., 2001)

[1]: in this classification, an adjusted temperature considering that temperatures below freezing or above 30°C do not contribute to life proliferation.

2.3. Case study: bio-based polymers

The converted characterisation factors CFs' were applied on a case study: the comparison of different production locations of bio-based polymers, namely polylactic acid (PLA), polyhydroxy-alkanoate (PHA), thermoplastic starch (TPS) and green polyethylene. Scenarii were designed according different feedstocks and production locations. Since the comparative LCA is not the purpose of the present paper, only the CFs' associated with all those scenarii are presented.

3. Results

3.1. Monetary and characterisation factors

Monetary factors (Figure 1 for the BPP) were calculated according to economic data, mostly available at the country level. Economic valuation assumptions will be discussed. The choice of compensation systems depends indeed of the performances required (increasing with the severity of the impacts modelled in the midpoint level) and their real availability (described with the adaptation capacity): a technology may not be available or affordable in a given country.

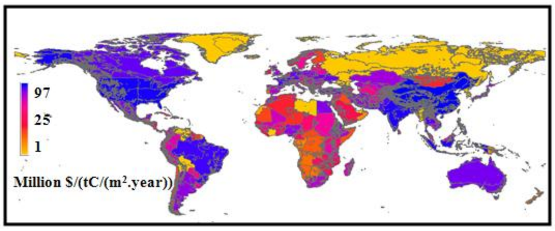


Figure 1. Map of the monetary factors for the Biotic Production Potential (BPP) into monetary units representing the loss of natural capital of ecosystem services (FAO data, country level).

Multiplied with the physical midpoints and the exposure factor (1–AC), they generate the converted characterisation factors CFs' for all the nine land covers developed in 2.2.b, at a global level. Biogeographic and economic boundaries were combined together to draw new frontiers for the ecosystem services (2.2.c). For instance, a map for an agricultural land use (or cover) is represented in Fig. 2.

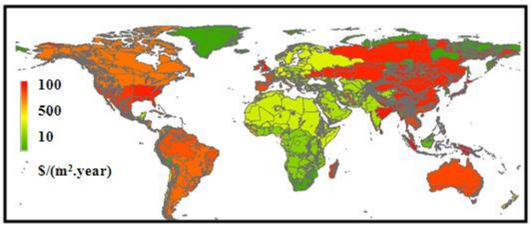


Figure 2: Map of the Biotic Production Potential (BPP) endpoint indicators for the land cover "permanent crops" (IPCC climate zones and FAO country frontiers intersected).

3.2. Case study

The converted characterisation factors CFs' can be used to characterise land use inventory flows into an integrated indicator representing ecosystem services loss in monetary unit ($) (Figure 3). Please note that this assessment step assumes an equal weighting among the different midpoints. Results show that potential impacts are specific to the types of ecosystem service, the location and the type of land cover. This latter affects indeed the physical midpoint (e.g. intensive harvesting degrades more than regular harvesting) and the type of ecosystem service damaged, while location affects them because of their spatial variability. While the physical midpoint assesses the potential effects on ecosystem services, the converted midpoint goes a step further, as it takes into account the local and actual need of the ecosystem services. For example, ecosystem services related to water filtration prevales in Australia, because of both its local scarcity and relative important need.

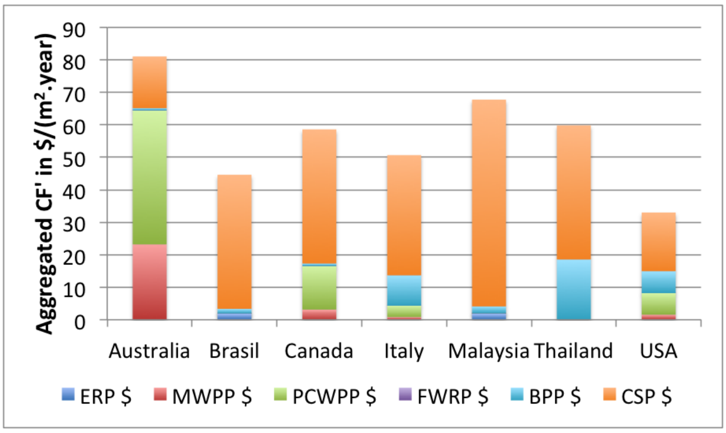


Figure 3. Graph of converted and aggregated land use CFs' for different biopolymer production locations (monetarisation normalisation, equal weighting) for an agricultural land cover.

4. Discussion

The presented framework consists to estimate the *direct* benefits (e.g. use value) provided by the ecosystem services through compensation systems: what would be today's ecosystem benefit lost by the use and degradation of the land. It therefore relies on the assumption/virtual situation that the conservative approach by *local* compensation is always possible and implemented. This estimated value does not capture more elaborate alternatives such as importation, substitution (of biotic products for example), migration (abandon of the ecosystem).

As for using compensation of ecosystem services, it should be highlighted that only the cost of technological systems is assessed by this methodology, this is not consequential life cycle (impact) assessment to the extent that potential impact related to the compensation systems assessed are not taken into account. The new area of protection represents indeed the ecosystem services loss, which can also be interpreted as a (natural) cost to society. Consequential LCA still depends on the study goal and scope. It remains the choice of the practitioner to include or not the complete inventory of the compensation systems, so that all the different impacts on all the categories are assessed.

The developed methodology uses international coherent databases, also recognized to be regularly updated. This lets some place to evolution and adjustment, as economic valuation of ecosystems is still in its early developments and that cost values change with time. An example is the social cost of carbon, which value highly depends on the chosen temporal perspective, itself related to the different perspectives defined in LCA (Hofstetter, 1998). Moreover, this methodology can potentially be applied to future (if developed) midpoints inside, but also outside the land use impact category.

5. Conclusion

The development of the conversion factors allows to express all the midpoints proposed by the LULCIA approach to economic values, bringing a whole new level of interpretation as natural ecosystem services loss. Their economic valuation may potentially allow future LCAs to assess other impacts related to land use, such as aesthetics and recreational aspects (not assessed so far, even outside the LCA).

Moreover, this work allows the aggregation several midpoint indicators into a single indicator related to the area of protection ecosystem service loss, which facilitates decision-making.

6. References

Ackerman, F., Stanton, E. A., 2010. The Social Cost of Carbon - A Report for the Economics for Equity and the Environment Network. Stockholm Environment Institute, Stockholm.

Brandão, M., Mila iCanals L., 2012. Global characterisation factors to assess land use impacts on biotic production. The International Journal of Life Cycle Assessment. 1-10.

Boulay, A.-M., Bouchard, C., Bulle, C., Deschênes, L., Margni, M., 2011. Categorizing water for LCA inventory. The International Journal of Life Cycle Assessment. 16, 639-651.

de Baan, L., Alkemade, R., Koellner, T., 2012. Land use impacts on biodiversity in LCA: a global approach. The International Journal of Life Cycle Assessment. 1-15.

CDE, FAO, ISRIC, GEF, UNCCD, World Bank, 2011. World Overview of Conservation Approaches and Technologies (WOCAT). http://www.wocat.net/en/knowledge-base.html. Accessed 30 July 2011.

European Commission Joint Research Centre, 2010. Analysis of existing Environmental Impact Assessment methodologies for use in LCA. Institute for Environment and Sustainability, Luxembourg.

FAO, 2012. FAOSTAT. Accessed 15 July 2012.

Freeman, A.M., 2003. The Measurement of Environmental and Resource Values. Theory and Methods, second ed. Resources for the Future Press, Washington.

Hofstetter, P., 1998. Perspectives in life cycle impact assessment: a structured approach to combine models of the technosphere, ecosphere and valuesphere, ETH Zurich (Swiss Federal Institute of Technology).

Holdridge, L.R., 1947. Determination of World Plant Formations From Simple Climatic Data. Science. 105, 367-368.

Garrigues, E., Corson, M.S., Angers, D.A., van der Werf, H.M.G., Walter, C., 2012. Soil quality in Life Cycle Assessment: Towards development of an indicator. Ecological Indicators. 18, 434-442.

IPCC, 2009. Agriculture, Forestry and Other Land Use. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. National Greenhouse Gas Inventories Programme & Institute for Global Environmental Strategies, in: The Intergovernmental Panel on Climate Change (IPCC), Hayama (Japan).

Koellner, T., de Baan L., Beck, T., Brandão, M., Civit, B., Goedkoop, M., et al., 2012. Principles for life cycle inventories of land use on a global scale. The International Journal of Life Cycle Assessment. 1-13. Accessed 30 April 2012.

LULCIA, 2008-2012. Operational Characterization Factors for Land use Impacts on Biodiversity and Ecosystem Services in the Life-Cycle Impact Assessment. http://www.pes.uni-bayreuth.de/en/research/projects/LULCIA/

Mila iCanals L.M., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R.F., Gaillard, G., et al., 2007. Key elements in a framework for land use impact assessment within LCA. The International Journal of Life Cycle Assessment. 12, 5-15.

Millennium Ecosystem Assessment, 2005. Our Human Planet, Island Press, Washington.

Müller-Wenk R., Brandão M., 2010. Climatic impact of land use in LCA—carbon transfers between vegetation/soil and air. The International Journal of Life Cycle Assessment. 15, 172-82.

National Research Council, 2004. Valuing Ecosystem Services: Toward Better Environmental Decision-Making, National Academy Press, Washington.

Olson, D. M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., et al., 2001. Terrestrial Ecoregions of the World: A New Map of Life on Earth. BioScience. 51, 933-938.

Saad, R., Margni, M., Koellner, T., Wittstock, B., Deschênes, L., 2011. Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. The International Journal of Life Cycle Assessment. 16(3), 198-211.

UNESCO, 2009. Water in a changing World. United Nations World Water Development Report, UNESCO, Paris.

US EPA, 1999. Water Treatment Estimation Routine (WaTER). www.usbr.gov/pmts/water/media/spreadsheets/WaTER.xls. Accessed 15 August 2011.

Land requirements for food in the Netherlands, a historical analysis over 200 years

Sanderine Nonhebel [1,*], Jan Hessels Miedema [1], Thomas Kastner[2]

[1] Centre for Energy and Environmental Sciences, University of Groningen. Groningen, The Netherlands
[2] Institute for Social Ecology, Alpen Adria University, Vienna, Austria.
[*] Corresponding author E-mail: *S.Nonhebel@rug.nl*

ABSTRACT

The production of food puts a large claim on land. The land required for food depends on the menu and on the agricultural production (yields per hectare). Both factors change over time. We combined data on food consumption over 200 years in the Netherlands with data on agricultural yields over the same period and determined a time series on land requirements for food.
Large changes in the consumption patterns took place. In 1800 the largest share of the kcal in the food (90%) were obtained from milk, wheat, rye, buckwheat, barley and potatoes. 200 years later milk, wheat and potatoes only contributed for 40% to the menu, and pork, sugar and vegetable oils accounted for 40%. Crop yields went up with a factor 4-8. Combination of both made that in 1800 1.4 ha was needed to feed a person, while 200 years later only 0.2 hectare was needed.

Keywords: food consumption patterns, crop yields, land use, time series

1. Introduction

The production of food puts a large claim on land. On a global scale 30% of the land is used for the production of food. Earlier research showed that large differences in land claim existed between various food items and different menus required different land use. Luxurious menus with a lot of meat tend to require 5 times as much land as the menus mainly based on staple food like rice and potatoes. (Gerbens-Leenes and Nonhebel, 2002). The change of menu is strongly related to economic development. With increasing welfare menus change from diets mainly based on staples to diets including animal products, vegetable oils, vegetables and drinks.

The land requirements for food are inversely related to crop yields, when yields double land requirements for food halve. Crop yields per hectare are affected by changes in technology like the use of higher yielding crop varieties, application of (mineral) fertilisers, use of biocides against pests and diseases etc.

Over the last 200 years large changes both in menu as in agricultural technology have taken place in the Netherlands. In this paper we study the overall effect of changes in menu and changes in production techniques on the land required for food in the Netherlands. This historical analysis provides knowledge on how the need for land can change over time, insights can be of use when studying future global food supply.

2. Methodology

We constructed a historical database of food availability per person over 200 years using various sources. Next we collected yield data for most important food crops grown in the Netherlands. Finally we combined food availability data with the crop yields to determine the land required for food using the methodology developed by Kastner and Nonhebel (2010).

2.1 Food consumption patterns

Data on food availability were obtained from different sources, using different food categories and different units covering different timespans. All food units (kg, litres etc) were converted into kcal, and food categories were adjusted (Miedema, 2011). Fig. 1 shows the result. Around the first (1914-1918) and second (1940-1945) world war a large decline in food availability is shown. It is likely that in these years the availability of food deviated from the non-war years, but during wars the collection of statistical data is in general not the first priority of a government, so the quality of the collection of the data can also be a reason for deviation.

The total food availability increased in 200 years from 2500 kcal per person per day to over 3200 kcal. Next to this, changes in the pattern are obvious: around 1800 over 40% of the food originated from livestock (with dairy contributing virtually the whole share), this drops to 20% in 1900 and increases again to 35% in 2000 (Figure 1). In 1800, rye and barley were the cereals most consumed, while in 2000 wheat was the most important cereal. While the consumption of relevant quantities of sugar only started around 1880, in 2000 15% of the calories in the menu were obtained from sugar. Further from 1950 onwards animal fats in the diets were replaced by oils from the oilseeds.

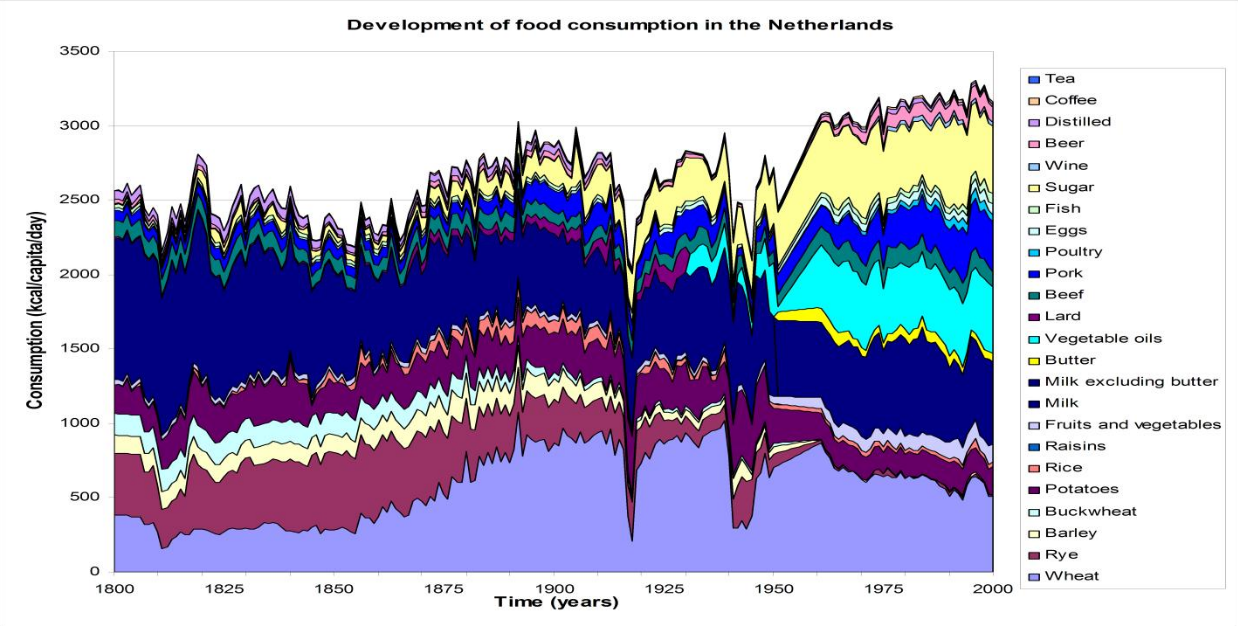


Figure 1 Food availability in the Netherlands (kcal per person per day) from 1800 onwards (source: Knibbe 2007; Miedema 2011).

2.2 Changes in technology (yields)

For the major food crops used in The Netherlands yield data were available for 1855 to 2010. Fig. 2 shows the results. Large differences in yield improvements exist: the yield of rapeseed doubled in 150 years, while the yield of wheat increased 5 times. Buckwheat disappeared from the agricultural system in the Netherlands, only up to 1940 yield data were available. Between 1855 and 1900 not much change in yield was observed. We assumed that in the years before (1800-1855) the yields were also constant. The increase in yields after the Second World War is due to the increased use of fertilisers in The Netherlands: this increased 6 times from 100 mln kg to 600 mln kg.

For the production of milk grass is required. Yields for grasslands are very difficult to obtain, since grass is not harvested and weighted, but cows consume it directly in the fields. Presently yields of grass are in the order of 10 t/ha (Aarts et al., 2005). These yields imply fertilised grasslands. We assumed the development in yields of grasslands followed the same pattern as wheat. This implies that grassland yields were around 1 t/ha in the nineteenth century and increased rapidly after the second world war.

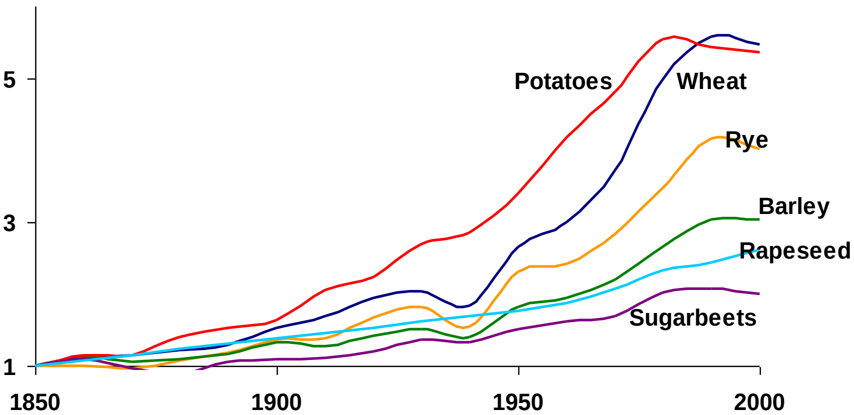


Figure 2. Relative developments in crop yields in the Netherlands in the last 150 year (sources: Smits and van der Bie, 2003, Sneller 1943.)

2.3 Conversion from kcal into area

To determine the land required for food, the food units (kg or litres) have to be converted into area of land. In principle this is done using the yields. Higher yields imply lower land requirements. Some of the food products like sugar and vegetable oils etc. are processed agricultural crops for these products a conversion factor is introduced identifying how much sugar beet is required for the production of 1 kg of sugar. The appendix shows the conversion factors used.

In this analysis we assumed that conversion factors remained constant over the timespan studied. This is not the case in the actual situation. However, in this study we are interested in the interplay between changes in consumption and changes in agricultural production. When we would also introduce changes in conversion techniques the overall picture becomes very complicated and too difficult to analyse. With respect to animal products we applied the same methodology. We assumed that an animal product is the result from a conversion of an agricultural product (wheat into eggs, grass into milk).

3 Results

Fig. 3 shows the land requirements for food per person in the Netherlands over the last 200 years as calculated with the methodology described above. In 1800 about 1.4 hectare per person was needed for the production of food, two thirds of this was for the production of dairy. In the following two centuries, the area needed declined fast to 0.15 ha per person in 2000 (Figure 3). From 1800 to 1900 the decline was caused by the reduced consumption of animal products, from 1900 onwards the consumption of animal origin products increased again but due to use of mineral fertilisers the production per hectare increased, leading to overall decline of the land required for food per person.

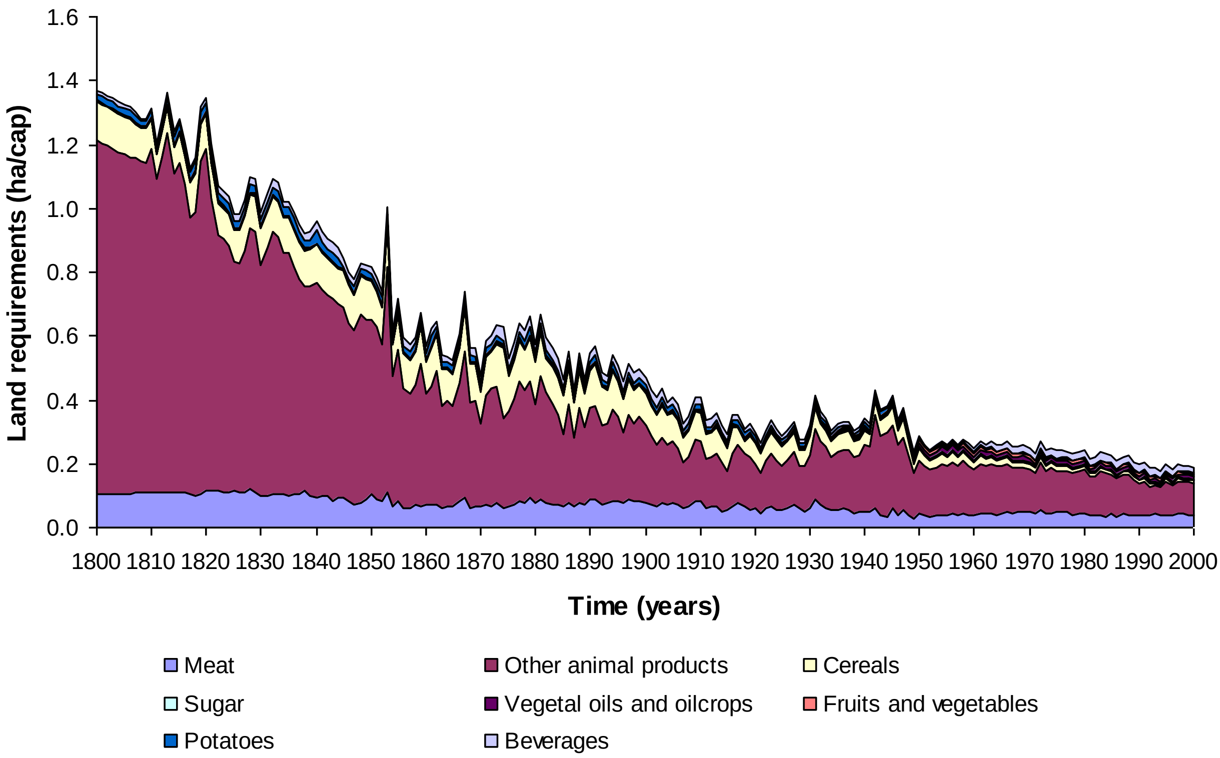


Figure 3 Development of the per capita land requirements in the Netherlands and the distribution over various consumption categories.

4 Discussion

The analysis in this paper uses statistical data over 200 years. The quality of the data used cannot be expected to be constant. Crop yield data from 200 years ago are collected in a different way than present data, the same holds for the data on food availability. In this analysis we assume that all the food was produced in the Netherlands, in the actual situation a part of the food is imported from somewhere else where other yields are obtained. And finally we assume that the conversion factors remain constant over time, which is actually not the case since technological improvements make that presently more oil/sugar etc can be obtained from a crop than 200 years ago.

This implies that the results presented should not be interpreted on face value. We did not calculate the actual land required for food in 1800. For the determination of that value we should have included imports, technological improvements etc and we should have had insights in the reliability of the data. The value of the analysis presented here is in the developments over time and the interrelation in food consumption patterns and agricultural production.

This analysis provides some interesting insights. In the first place it is striking that 200 years ago dairy played such a vital role in the Dutch consumption pattern. This is in contrast with the patterns observed on a global scale: namely that the consumption of animal origin products only starts when a certain level of welfare is reached. The large consumption of milk in the early 19th century is also mentioned in more qualitative food consumption studies (Jobse-van Putten, 1995) they mention that Dutch households consumed a milk based meal (mainly porridge) 14 times a week (so 2 times a day!).

With respect to the production: the crop yields remained more or less constant in the 19th century and after the second world war huge yield improvements were obtained. Crop yields went up with a factor 4-8. But differences between crops were recognised. Some crops disappeared from the agricultural system like buckwheat and others became of minor importance (rye).

The consequence of all these changes is a fast decline of land required for food per person from 1.4 to 0.2 ha per person. This value is in accordance with other studies on land requirements for food in the Netherlands (Gerbens-Leenes et al., 2002; Gerbens-Leenes and Nonhebel, 2002). They determined the land required for food in The Netherlands in 1990, in a far more detailed way: including imports, present day technologies, far more detailed information on the consumption pattern etc.

In the studied period land requirements for food per person was reduced by a factor of 8. However, in the same timespan the population increased from 2 to 17 million people, so the total amount of land needed to feed the Dutch population remained actually constant.

In a recent study on global changes in land requirements for food in the last 50 years, this decline in land per person is found in most regions in the world (Kastner et al., 2012). The magnitude of the decline is smaller than calculated here. For a part this can be explained by the high crop yields that are obtained in the intensive Dutch agricultural system. The yields in the Netherlands tend to be the highest in the world; with the same menu the land requirements are smallest. Another important feature is the high animal product consumption in the Netherlands 200 years ago. The land requirements for meat are determined by the yields of the crops used for feed. Roughly 4 kg of feed are required for 1 kg of meat. A menu mainly based on staple foods requires less land than a menu with meat. For the Dutch situation we observe a decline in the consumption of animal products over the time studied, in combination with the increased yields this leads to the huge decline in land required for food. This decline is typical for the Dutch situation and cannot be generalised to the rest of the world.

5 Conclusions

In 200 years large changes in the Dutch consumption patterns can be recognized. The share of dairy, rye and buckwheat declined and the share of wheat, meat, sugar and vegetable oils increased. The agricultural production system also changed a lot: after the Second World War an increase of the productivity with a factor 4 was obtained. The combination of the change in diets and the change in agricultural productivity led to a reduction of the land required to feed a person from 1.4 ha in 1800 to 0.2 ha in 2000.

6 References

Aarts, H. F. M., Daatselaar, C. H. G., Holshof, G., 2005. Bemesting en opbrengst van productiegrasland in Nederland. Plant Research International B.V. Wageningen.

Gerbens-Leenes, P.W., Nonhebel, S., 2002. Consumption patterns and their effects on land required for food. Ecological Economics 42, 185–199.

Gerbens-Leenes, P.W., Nonhebel, S., Ivens, W.P.M.F., 2002. A method to determine land requirements relating to food consumption patterns. Agriculture, Ecosystems & Environment 90, 47–58.

Jobse-van Putten, J., 1995. Eenvoudig maar voedzaam: cultuurgeschiedenis van de dagelijkse maaltijd in Nederland. Nijmegen: SUN.

Kastner, T., & Nonhebel, S., 2010. Changes in land requirements for food in the Philippines: A historical analysis. Land Use Policy 27, 853-863

Kastner, T., Ibarrola Rivas, M.J., Koch, W., Nonhebel, S., 2012. Global changes in diets and the consequences for land requirements for food. Proc. Natl. Acad. Sci. USA 109, 6868-6872

Knibbe, M. T., 2007. De hoofdelijke beschikbaarheid van voedsel en de levensstandaard in Nederland 1807 – 1913. Tijdschrift voor sociale en economische geschiedenis, 4, 71-107.

Miedema, H.J., 2011. Developments in the Dutch diets: dynamics on two centuries of land requirements for food in the Netherlands. Master Thesis, University of Groningen. EES 2011-125T.

Smits, J. P., & van der Bie R. J., 2003. Tweehonderd jaar statistiek in tijdreeksen 1800–1999. Voorburg / Heerlen: Centraal Bureau voor de Statistiek.

Sneller, Z. W., 1943. Geschiedenis van den Nederlandschen landbouw 1795 – 1940. Groningen: J.B. Wolters Uitgevers-Maatschappij N.V.

7 Appendix

Conversion factors used (source: Miedema, 2011)

Food products and calorific value		Primary products and calorific value		Conversion factor
	kcal/kg		kcal/kg	Calorific value food product / Calorific value primary product
Wheat	3340	Wheat	3340	1.00
Rye	3190	Rye	3190	1.00
Barley	3320	Barley	3320	1.00
Buckwheat	3300	Buckwheat	3300	1.00
Potatoes	670	Potatoes	670	1.00
Rice	3600	Rice	3600	1.00
Raisins	400	Grapes	530	0.75
Fruits and vegetables	200	Fruits and vegetables	200	1.00
Milk	620	Milk	620	1.00
Milk excluding butter	620	Milk	620	1.00
Butter	7170	Milk	620	11.56
Vegetable oils	8840	Rapeseed	4940	1.79
Margarine	7200	Rapeseed	4940	1.46
Diet-margarine	3630	Rapeseed	4940	0.73
Beef	2250	Wheat	3340	4.00
Pork	2360	Wheat	3340	4.00
Poultry	1220	Wheat	3340	3.00
Eggs	1480	Wheat	3340	1.25
Fish	1050	Wheat	3340	1.00
Sugar	4000	Sugarbeets	700	5.71
Wine	750	Grapes	530	1.42
Beer	400	Barley	3320	0.12
Distilled	1980	Barley	3320	0.60
Coffee	560	Coffee	560	1.00
Tea	400	Tea	400	1.00

Organic farming without fossil fuels – LCA of energy self-sufficiency

Cecilia Sundberg[1,*], Marie Kimming[1], Åke Nordberg[1,2], Andras Baky[2], Per-Anders Hansson[1]

[1] Department of Energy and Technology, Swedish University of Agricultural Sciences
[2] JTI – Swedish Institute for Agricultural and Environmental Engineering
[*] Corresponding author. E-mail: *cecilia.sundberg@slu.se*

ABSTRACT

One principle of organic farming is the use of renewable resources, yet it depends on fossil energy. Systems for supplying organic agriculture – one arable farm and a system for milk production - with heat, power and fuel generated from biomass produced on its own land, and the consequences of such production were investigated. Biomass energy systems included power, heat and ethanol from straw and biogas generating power, heat and fuel from ley, manure and straw. For the arable farm, straw or ley from the crop rotation was sufficient for energy supply for farm activities. For milk production, biogas from manure could supply about two thirds of the energy demand of the farm and there was straw to supply the remainder. GHG emissions were reduced in all scenarios compared to the fossil reference. The GHG results were sensitive to assumptions on soil carbon initial content and turnover.

Keywords: biogas, biomass, CHP, renewable energy, straw

1. Introduction

Even though organic farms aim to lower the environmental impact from food production and rely mainly on renewable, locally available resources, they still depend on fossil fuels for energy supply to production processes. Organic farms could increase their credibility as a sustainable alternative if renewable resources were used for producing energy for the farm. The agricultural sector is also considered to have the largest potential to contribute to a higher share of renewable energy in the EU (EEA 2006).

The technical development of systems for energy generation based on biomass has progressed rapidly over the last few years and the number of small-scale applications suitable for farm use has increased. Dairy farms have access to readily available manure which can be used for biogas production. Biogas can be used for heat and power production or is upgraded to vehicle fuel. Hydrolysis of cellulosic substrates is the next generation of ethanol production, projected to replace the much debated production of ethanol from food crops such as sugar or starch products.

This paper summarizes the findings our research on how to supply organic agriculture with energy produced on its own land, and the environmental consequences of such production. The purpose of this study was to assess the self-sufficiency potential, greenhouse gas emissions and energy balance of crop production (described in detail in Kimming et al., 2011) and milk production (Kimming et al., 2012) in a renewable energy supply systems mainly based on bioenergy, compared with systems based on fossil fuels.

2. Method and scenario description

2.1. LCA approach and functional unit

Since the goal was to investigate the impact of changing to a new energy supply system, consequential LCA was used for these studies. The substitution method was used to avoid allocation.

The functional unit (FU) used was 1 kg ECM (energy-corrected milk) at the farm-gate for the milk study. For the arable farm, the FU was the total supply of energy (heat, electricity and vehicle fuel for the 200 ha organic farm for 1 year. The impact categories were energy balance and global warming potential (GWP100).

2.2. Agricultural production systems

The milk farm was assumed to have 100 dairy cows, with an average of 84 animals being milked every day. Data on milk production are presented in Table 1. The farm was assumed self-sufficient in organically produced forage (Table 2).

Table 1. Input data to milk study

Factor	Value
Milk production per cow	8000 kg ECM per year
Replacement rateText	25%
Meat produced per kg ECM	0.0057 kg
Rapeseed oil produced per kg ECM	0.028 kg
GHG emissions for substituted meat production	22 kg CO_2-eq.
GHG emissions for substituted rapeseed oil production	0.8 kg CO_2-eq.

Table 2. Feed production

Product	Yield[a] (kg/ha yr)	Area (ha)
Ley	4230	120
Grazing	4000	70
Spring barley	2440	54
Winter wheat	3228	40
Broad beans	2026	54
Rapeseed cake	1693	40

[a] [a]Yields for barley and wheat are given at 86% dry matter content (DM), rapeseed 91% and broad beans 85% DM. Ley yield is in total solids (TS), with losses during harvest and ensiling subtracted

The 200 ha arable farm had a 7-year crop rotation (Table 3) that included ley twice, to be used as green manure (field beans, oats, ley, winter rapeseed, winter wheat, ley, rye). In the reference scenario, ley and straw were ploughed back into the soil.

Table 3. Arable crop rotation

Crop	Yield[a] (kg/ha yr)
Field beans	2400
Oats	3200
Ley	6000
Winter rapeseed	2000
Winter wheat	3500
Ley	6000
Rye	3200

[a]Crops dried to 86% DM except ley, which is in kg DM

2.3 System boundaries

An overview of the production systems with energy self-supply is shown in Figure 1. The system boundary was set at the farm-gate for products produced on the farm (exception: biomass which is transported to fuel production facilities and back is included). Feedstock for energy carrier production is forwarded to conversion facilities (“Conversion processes”-box in Figure 1). Residual products from the conversion processes, such as ashes and digestion residues, were assumed to be returned to the fields as fertilisers. By-products from the conversion processes were accounted for (“Substituted production”-box in Figure 1). The system was compared with a reference system, in which the milk production process was supplied with energy produced with fossil fuels. Mining, extraction, refining, distribution and consumption of the fossil fuels were included.

2.4 Energy demand

The annual energy demand for the milk farm was 300 GJ electricity (0.14 MJ/kg milk), 115 GJ for grain drying and 95 GJ for hating of buildings. Annual tractor fuel demand was 460 GJ.

In the arable farm, heat was supplied to the residential building (dimensioned capacity 7.4 kW) the hot water system (1.2 kW), the workshop (1.7 kW) and the grain dryer (227 kW). The total annual tractor fuel demand was 414 GJ, electricity demand was 51 GJ and heat demand 290 GJ.

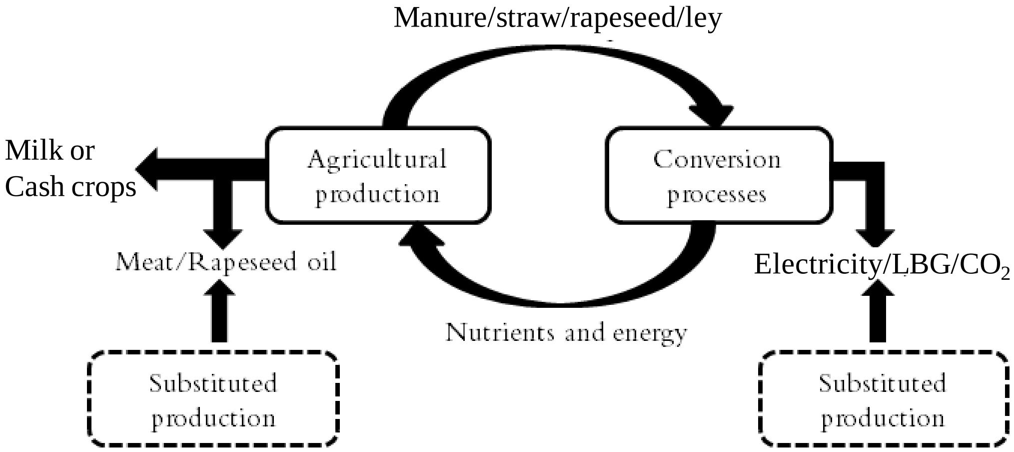


Figure 1. Schematic figure of the energy and material flows on the arable and milk farms respectively. Substituted production refers to the impact on the external market of reduced or increased supply of goods from the farm to the respective market. LBG = liquid biogas.

2.5 Renewable energy supply systems

In the milk production system biogas from manure was the main energy supply in all scenarios. In Scenario M1, biogas produced from manure and cut straw was assumed to cover the entire energy demand. A fraction of the gas was fed to a combined heat and power (CHP) system (gas engine) with electric output 14 kW. The rest was pumped via low pressure pipelines to a cryogenic upgrading facility for liquid biogas (LBG) production. LBG was assumed to be delivered in trucks to the farm, where it was pumped to the tractors via a pumping station in which the LBG first vaporized.

In Scenario M2a**,** the manure on the farm was utilised to produce biogas, assumed to be combusted in a CHP system (gas engine) with electric output 13 kW. The rapeseed oil was assumed to be used to produce rapeseed methyl ester (RME) in a small-scale production unit at the farm. The tractors ran on RME with minor modification of the original diesel engines, such as replacing components with others of more resistant materials (Ahlgren et al., 2010).

In Scenario M2b, RME was assumed to be produced in the same way as in Scenario 2a, but used in a combustion engine for CHP production, with electric output 13 kW. Tractors are assumed to run on biogas from manure, from which LBG is produced in an external production facility as in Scenario 1.

In Scenario M3, it was assumed that a wind turbine, owned by a farm cluster, supplies electricity and heat to the farm (via heat pumps). Required capacity is 70 kW, embedded in a larger wind tower. Tractors run on LBG produced from manure, upgraded to LBG as in Scenario 1 and 2b.

The CHP systems in scenario M1, M2a and M2b were dimensioned to work on full load 95% of the time. The heat produced was used to heat buildings on the farm and for the anaerobic digestion process.

Grain produced on the farm was assumed to be dried in a straw-fuelled furnace in all M-scenarios (capacity 232 kW). Total energy consumption was 115 GJ which came from approximately 4 ha of wheat straw.

The biogas production took place under anaerobic conditions at mesophilic temperature, in a one-stage anaerobic digester of 250 m^3 in Scenario M1 and 150 m^3 in Scenario M2 and M3. The raw biogas was assumed to be upgraded to vehicle fuel quality in a large-scale cryogenic upgrading plant separating off methane (CH_4), carbon dioxide (CO_2), water vapor and impurities at their different condensing temperatures (Johansson 2008). The process requires 1.62 MJ electricity/Nm^3 and the methane losses in existing facilities are 0.5%. Here, the plant was assumed to be located 50 km from the farm and LBG was delivered back to the farm in vacuum-insulated trucks.

The biogas production in each scenario was dimensioned based on the availability of substrate and not on the demand for energy on the farm, as this was considered the most feasible solution. This might result in overproduction of LBG and electricity. Surplus LBG was assumed to be sold and replace use of diesel. Surplus electricity was assumed to be sold to the market and electricity produced in natural gas-fired condensing plants.

In the arable system, one scenario (A1) was based on biogas from ley. Assumptions were largely the same as in the milk system. The upgrading process produces liquid CO_2 (LCO_2) as a by-product, which can be sold as a refrigerant. In this case, it was assumed that the LCO_2 replaced the HFC type R404a as a refrigerant in grocery shops. However, this market is currently rather undeveloped and therefore only 1% of the LCO_2 was assumed to be sold and the rest used as coolant in the upgrading process (Johansson, 2008). Biogas was used also for the grain dryer.

Scenario A2 was based on straw, which was converted to ethanol via hydrothermal pretreatment, to enzymatic hydrolysis and fermentation. The lignin is separated out during hydrolysis and was assumed to be used in an integrated CHP plant for production of process steam and electricity, as well as surplus electricity to cover the power demand of the farm (51 GJ). The plant was assumed to be a single pressure combined cycle gas turbine (CCGT) and the flue gases recovered in a heat recovery steam generator (HRSG). Straw furnaces supplied heat for farm buildings and the grain dryer.

2.6. Greenhouse gas emissions

Methane emissions from storage of manure and digestion residues were calculated according to the IPCC Guidelines (IPCC 2006). Methane is formed in the digestive system of dairy cows when cellulose is decomposed via anerobic microbial activity and depends on milk production rate, feed and the cow's body weight. For a dairy cow producing 8000 kg milk/yr, assuming a metabolic weight of 600 kg/cow the emissions are 120 kg CH_4/cow and yr. For heifers, 50 kg CH_4/animal and yr was assumed (Cederberg et al., 2007).

Direct emissions of nitrous oxide (N_2O) from soil and fertilisers were calculated based on IPCC methodology (1% of added nitrogen and 1% of nitrogen content in crop residues). Indirect emissions in the form of leached nitrogen (which volatizes downstream) were set to 0.75% of leaching (IPCC 2006) and the amount leached was assumed to be 11 kg N/ha (Wivstad et al., 2009). Emissions of N_2O from storage of manure and digestion residues were set at 0.1% of the nitrogen content available in the substrate, based on emission factors for liquid manure (IPCC 2006).

The soil carbon balances of the cultivation systems studied were simulated with the ICBM (Andrén and Kätterer 2001). The model assumes two carbon pools (a 'young' pool, Y, and an 'old' pool, O) of which carbon entering the old pool is considered a carbon sink, as the decomposability is 100 times lower than in the young pool, where rapid mineralisation of carbon as CO_2 takes place. Equations 3a and 3b describe the dynamics of the carbon balance for the young and old pool, respectively:

$$dY/dt = i - k_Y r_e Y \text{ (kg)} \qquad \text{Eq 1}$$

$$dO/dt = h k_Y r_e Y - k_O r_e O \text{ (kg)} \qquad \text{Eq. 2}$$

where i is the input of crop residues or manure, k_y and k_o are the decomposition rates of Y and O respectively, r_e is the external influence component (including for example soil climate, crop type and frequency of soil tillage, and is lower for ley and grazing fields), and h is the humification coefficient (dimensionless). The humification rate depends on the composition of the respective material. Values are described further in Kimming et al., (2012). The annual mineralisation of carbon to carbon dioxide released to the atmosphere is a mean over 21 years (3 crop rotations).

4. Results

4.1 Self-sufficiency potential and energy balance

The renewable energy supply systems in Scenario M1, M2a and M3 can make the farm self-sufficient in energy under the given assumptions. In scenario M2b, only two thirds of the electricity requirement on the farm was covered by the available rapeseed oil, so Scenario 2b was excluded from further calculations of energy balance and GWP.

On the arable farm, Scenario A1 requires that ley is harvested from 25 ha, which is 13% of the total farm area. A2 requires 49 ha of straw, i.e., 25% of the total farm area. This biomass is available in both scenarios, since ley is planted on 29% of the farm area and cereals on 44% in the given crop rotation.

In the self-supply scenarios, 2.63 MJ of fossil energy (primary energy) per kg ECM was saved in the milk production system and 755 for the whole 200 ha farm in the arable system

4.2 GWP

The GHG emissions from the milk production reference scenario (962 g CO_2/kg ECM) were similar to those in previous studies of organic milk production with conventional energy supply systems (e.g. Cederberg et al., 2007; Thomassen et al., 2008). Almost half the total GWP in the reference scenario stemmed from methane in enteric fermentation (43%) and manure storage (10%). Fossil energy use accounted for 22% of total GWP. Changing to an energy supply system based on renewable sources resulted in reductions of 29-44% of total GHG emissions from milk production, with Scenario 1 giving the largest emission reduction, and Scenario 2 the lowest. Avoided fossil fuel use was the largest contributing factor to the reduction in GHG emissions in the renewable energy scenarios. The second most important parameter is the reduction in methane emissions when manure was passed through an anaerobic digestion process before storage.

In the arable farm study, The GHG emission saving was 35% in the self-sufficiency scenario based on ley (A1) and 9% in that based on straw (A2). There was less nitrous oxide from the soil in both self-sufficiency scenarios compared with the reference scenario, but the impact on the carbon content of the soil differed significantly, with a larger reduction in soil carbon content when straw was removed from the fields (Figure 2).

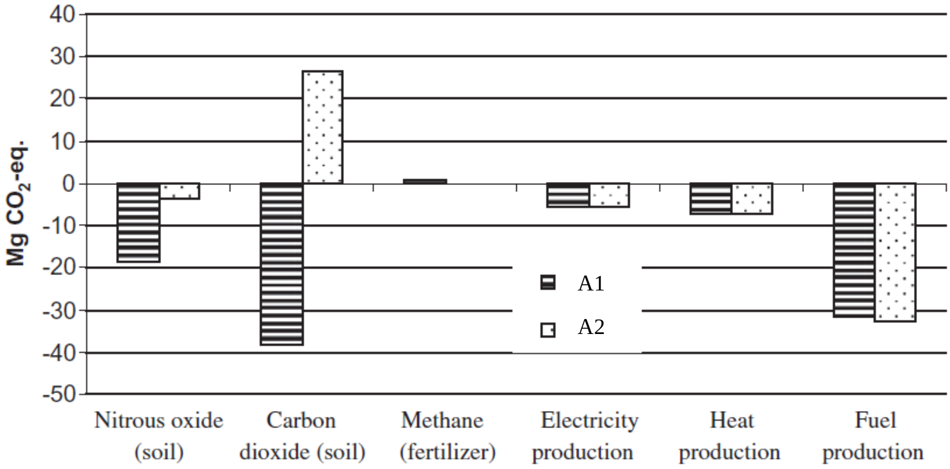


Figure 2. Disaggregated relative greenhouse gas emissions in the arable farm system. The bars above 0 are net increases in GHG emissions compared with the reference, and the bars below 0 are net decreases.

5. Discussion

In both the milk and the arable system, it was found that the biomass resources available as residues on the farm were sufficient for supply of energy for the production. There was consequently no need to reduce the production of food products, or increase the land area needed for the total production system.

For the milk study, the functional unit was 1 l of milk. This seemed reasonable since the milk farm has one major product. For the arable farm, the situation was quite different, with a number of products in a crop rotation. Therefore a functional unit was selected based on the whole farm, and the functional unit was limited to the energy supply to the system. In both cases GHG emissions are expressed as emission reduction compared to a reference scenario, whereby the functional unit is less important.

We chose to include technologies that are technologically available at least at a demonstration level, but not necessarily commercially available. This seemed most interesting, since there is little evidence of any small scale biomass CHP technology being commercially available today in Sweden. In other European countries with higher electricity prices and stronger incentives for small-scale electricity production however, the situation is quite different. Similarly, for tractor fuel, the systems evaluated are technically possible, but in most cases not commercially available. Both small-scale and large scale production of biofuels was investigated, selected based on technical feasibility.

The self-supply principle was been applied as at a farm level on an annual average basis. The energy systems have normally not been designed for a self-supply independent of existing energy infrastructure. In our part of the world, there is an electricity grid that reaches vertically all citizens and all farms. It has therefore been most relevant to investigate systems which are connected to the electricity grid, and where electricity can be supplied to, and accessed from, the grid. Local electricity production has been designed to cover the farm´s total electricity demand on an annual basis.

For gas, there is also a grid available in large parts of Europe. However, most of Sweden is not covered by a gas grid. It is possible to use natural gas infrastructure also for biogas, but that requires cleaning and upgrading of the gas, so it is not self-evident that available gas grids can be used for biogas. In most of our research we have not assumed any existing gas infrastructure available for biogas. When we have included gas grids in the system description, investment costs for building a local biogas grid were included in the analysis.

Production of bioenergy can be in conflict with the production of food, since land and water resources are limited. In life cycle assessment, as well in the bioenergy debate at large, this has been referred to as “indirect land use change”, an aspect that has traditionally not been included in LCA of agricultural products and systems, but in some recent attempts indirect land use change has been included. In this work, we have avoided this issue by only using agricultural residues as biomass sources for bioenergy, and thus not reducing the amount of land available for food production at all. We have not considered any effects on farm yields by these modifications. For example, it is plausible that yields can be increased by producing biogas from ley and optimizing the application of digestate, as compared to ploughing down the yield. Another case is when straw s removed and soil carbon content is reduced, which could result in reduced soil fertility and lower yields.

6. Conclusions

Swedish milk production can become self-sufficient in energy by utilizing renewable sources available on the farm, and thereby reduce GHG emission from production of 1 kg of ECM by 29-44% compared with a conventional farm system. The highest GHG emission reductions were found in a system where energy was supplied only with biogas, while a system with RME was least favourable.

The arable organic farm studied could be self-sufficient in energy by using the residues available in the crop rotation: using ley for biogas production or straw for ethanol, heat and power production. Because of due to soil carbon losses the greenhouse gas emission savings are lower with the straw system (9%) than the ley system (35%).

7. Acknowledgements

The research was funded by FORMAS, the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning.

8. References

Ahlgren S., Baky A., Bernesson S., Nordberg Å., O. Norén O. Hansson P.A, 2010. Future Vehicle Fuel Supply for Swedish Agriculture. JTI Report 392. Lantbruk och Industri. Uppsala, JTI - Swedish Institute for Agricultural and Environmental Sciences.

Andrén, O., Kätterer, T., 2001. Basic principles for soil carbon sequestration and calculating dynamic country-level balances including future scenarios. Assessment Methods for Soil Carbon, 495–511.

Cederberg, C., Flysjö A., Ericson, L., 2007. Livscykelanalys (LCA) av norrländsk mjölkproduktion. SIK-rapport 761. Stockholm, Swedish Institute for Food and Biotechnology.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventory. Volume 4 Agriculture, Forestry and Other Land Use.

Johansson, N., 2008. Production of liquid biogas, LBG, with cryogenic and conventional upgrading technology. . Department for Environmental and Energy Systems Studies. Lund, Lund University. Master thesis.

Kimming M., Sundberg C., Nordberg A., Baky A., Bernesson S., Norén O., Hansson P-A., 2011. Life cycle assessment of energy self-sufficiency systems based on agricultural residues for organic arable farms. Bioresource Technology. 102, 1425-32.

Kimming M., Sundberg C., Nordberg A., Baky A., Bernesson S., Norén O., Hansson P-A., 2012. Life cycle assessment of energy self-sufficiency systems based on agricultural residues for organic arable farms. Background report. Department of Energy and Technology, SLU

Kimming M., Sundberg C., Nordberg A., Baky A., Bernesson S., Hansson P-A., 2012. Renewable energy supply for organic milk production: self-sufficiency potential, energy balance and greenhouse gas emission. Submitted manuscript

Thomassen, M. A., van Calker K. J., Smits, M. C. J., Iepema G. L., de Boer I. J. M., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. Agricultural Systems. 96, 95-107.

Wivstad, M., Salomon E., Spångberg J., Jönsson, H., 2009. Organic production - possibilities to reduce eutrophication. Uppsala, Swedish University of Agriculture, Centre for Sustainable Agriculture.

Peri-urban expansion: application of consequential LCA to assess land for food or housing

Alison Rothwell[1], Brad Ridoutt[2], Girija Page[1], Bill Bellotti[1]

[1]University of Western Sydney, School of Science and Health, Locked Bag 1797, Penrith, NSW 2751, Australia
[2]Commonwealth Scientific and Industrial Research Organisation (CSIRO), Sustainable Agriculture National Research Flagship, Private Bag 10, Clayton South, Victoria 3169, Australia
[*] Corresponding author. E-mail: *alison.rothwell@uws.edu.au*

ABSTRACT

Competing land uses such as urban expansion are impacting fresh food production in peri-urban Sydney. Providing both housing and fresh food at lower relative emissions rates should be a priority in policy development. This paper presents a method that enables a comprehensive view of environmental trade-offs at the urban-agricultural intersection. Using consequential LCA, global warming potential (GWP) was calculated for two peri-urban land use scenarios involving two distinct types of urban housing systems whilst accounting for agricultural diversity at the farm scale. In Scenario 1 Sydney's horticultural production was displaced by low density outer suburban housing. Scenario 2 involved retention of Sydney's horticultural production with an infill housing model. Results show that although differences were observed within the agricultural system, it is the urban housing system dominating GWP impacts, with scenario 1 having a GWP 74 t ha^{-1} higher than for scenario 2. Policy focus on urban housing systems for greenhouse gas abatement in the regional context is recommended.

Keywords: food security, lettuce, greenhouse gas emissions, land use change, urban development.

1. Introduction

This paper describes the use of consequential LCA (CLCA) to assess land use options at the agriculture-urban interface. Competing land uses such as urban expansion are impacting fresh food production in peri-urban Sydney (Australia). Research is divided regarding the status of peri-urban horticultural land management in the Sydney basin. Although many qualitative papers on the benefits of peri-urban agricultural land retention exist both nationally (Choy et al., 2008; Houston, 2005) and internationally (Condon et al., 2010; van Veenhuizen and Danso, 2007), quantitative, evidence based planning is predominantly absent. Although the value of production for vegetables in the Sydney basin represents approximately 46% of the NSW state total, at only 6% of the value of Australian production (ABS, 2010), proponents of urban development argue that Sydney's horticultural production is of little consequence. Urban development is politically and economically well supported. Government policies are trending in favour of development for urbanisation (NSW Department of Planning, 2010) with the number of horticultural farms in Sydney anticipated to halve under current planning policy (ARUP, 2010b). If this trend continues, horticulture may cease as a viable land-use and economic function in the Sydney basin. Sydney's balance of horticultural produce will then be imported from regional locations. It is not known, however, if the alternative food producing locations present a more or less sustainable environmental situation for vegetables on the Sydney market compared with local production. While there is a need to provide housing, opportunity exists for a data driven methodology to evaluate the environmental consequences of the decision to urbanise, for the purpose of informing policy and driving greenhouse gas (GHG) emissions abatement.

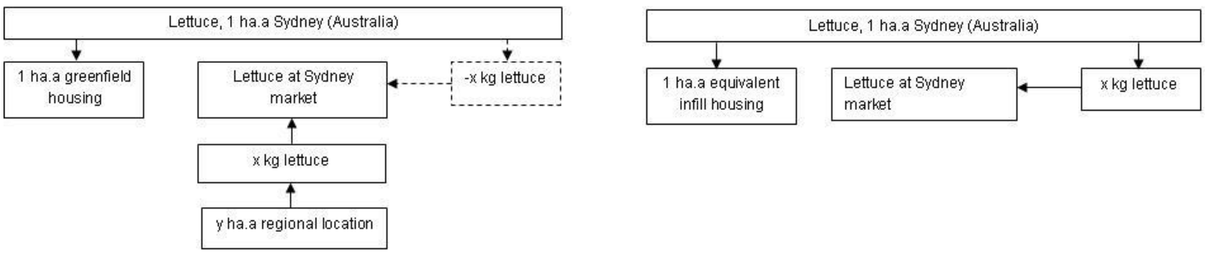
Australia currently exhibits the highest GHG emissions of any OECD country per capita, with average emissions exceeding four times the world average (Garnaut, 2008). Suburban residential development is a key contributor to these emissions, with differences reported in environmental impacts between different urban forms, notably between more energy intensive car reliant low-density outer urban greenfield style housing development versus lower energy intensive, infill development of inner suburban areas with existing public transport infrastructure (ARUP, 2010a; Camagni et al., 2002; Duffy, 2009; Fuller and Crawford, 2011). Despite this, urban expansion into new greenfield areas is the accepted norm for new housing development, with resistance to implementing higher density infill housing along key transportation corridors in established suburbs. Providing both housing and fresh food at lower relative GHG emissions rates should be a priority in policy development. The contradictory stance and poor integration between policy direction on emissions and metropolitan planning has been highlighted (ARUP, 2010b). Agricultural emissions are also known to be high relative to other OECD countries (Garnaut, 2008). Both agriculture and urban systems have differing and relatively large environmental impacts. At the peri-urban interface, what do the combined environmental impacts look like under different peri-urban land use scenarios?

Consequential LCA (CLCA) provides a methodology that can assess environmental impacts relating to using land for housing versus land for food, and assist answering the question "What are the environmental consequences of urbanising Sydney's horticultural lands?" Should peri-urban land be seen as land in waiting for conversion to low density housing, or do other housing decisions possess better environmental impact trade-offs? Specifically, in this paper, the benefits and challenges of applying CLCA to generate decision making evidence to support peri-urban land-use decisions are evaluated. In agriculture, CLCA has been used to assess land use decisions (Brandao, 2010). LCA is also used in the built environment, typically as attributional studies investigating different building forms and building materials (Rossi et al., 2012). In Australia, recent studies have compared embodied, operational and transportation energy and greenhouse gas emissions of alternative housing types, supporting the opportunity for energy consumption and emissions to be significantly reduced by shifting to inner-suburban apartment type dwellings over outer suburban detached homes (Crawford, 2011). Environmental impacts are typically assessed in isolation within each of the two disciplines of agriculture or urban planning. Using CLCA, the system boundaries can be defined to include the activities contributing to the environmental consequence of the change, meaning that impacts associated with both the urban and agricultural systems affected by this change can be included. The benefit of CLCA at the urban-agricultural intersection is to provide a new perspective on peri-urban land-use change. Scenarios for producing vegetables versus producing housing can be ranked according to their environmental impacts and consequential displacements associated with land use decisions assessed. Environmental impacts were assessed for both a certain amount of product to be produced for the Sydney market, and for a certain number of people to be housed. The method implemented in this paper permits for environmental impacts to be calculated for scenarios that include two distinct types of urban housing systems whilst accounting for agricultural diversity at the farm scale.

2. Methods

2.1 Scenarios

In applying CLCA to assess the combined environmental impacts involving both horticulture and urban land use at the peri-urban intersection, hypothetical scenarios were established as illustrated in Fig. 1. The use of CLCA allowed alternative means of increasing marginal production to be evaluated. Marginal production refers to the system that would be affected due to changes in demand. The system could be a technology (such as greenhousing versus field production) or a location (one growing region may expand in preference to another).



Scenario 1. Scenario 2.

Figure 1. Scenarios to be analysed for land-use change from 1 ha of lettuce to greenfield housing (scenario 1) or to equivalent infill housing (scenario 2).

In Fig. 1, scenario 1 illustrates how fresh food production is met via production in a marginal region, due to displacement from the Sydney region by housing. Scenario 2 illustrates how the housing burden is fulfilled by adoption of a predominantly infill housing system with no displacement of Sydney horticulture. A third scenario, scenario 3, analyses the trade-offs provided by a high technology greenhouse combined with increased parkland and infill urban housing compared to the scenarios 1 and 2. Modelling of scenario 3 will be completed in the next phase of this project and will not be discussed further in this paper.

2.2 Functional units and impact categories

SimaPro 7.3.3 software was used for modelling, selecting data from Australian Unit Process LCI or Ecoinvent unit processes. Two functional units were used for each scenario: one functional unit to represent a certain number of people being housed (in units of per hectare); the second functional unit to represent a certain amount of lettuce being produced for the Sydney market (in units of per kilogram). Having two functional units facilitates improved understanding of environmental burdens. Environmental impacts were estimated using the Australian indicator set. The indicators of interest include GWP, eutrophication, land use, water use and energy demand. However, only a subset will be presented within this paper, that being GWP.

2.3 Horticultural systems

Within the framework of this project, modelling at farm level was required in order to minimise use of average data to represent farms of differing sizes and in different regions. This is consistent with the purpose to compare and contrast impacts from farms in different geographical locations, with differing levels of inputs and outputs. A case study approach was taken, with lettuce supplied to the Sydney market selected as the commodity of choice. Lettuce was selected for the following reasons: it is a perishable crop and is ideally situated close to market; its relative economic importance within Sydney's horticultural production; a reasonable field planting area is occupied by lettuce in Sydney; and lettuce is a dietary staple in the Australian diet, unlikely to be displaced from the supply chain. The lettuce market, being highly competitive, price driven and with an absence of market policies, is likely to expand using growers that can increase their production with the least expense. Regional geographical locations identified based on ability to expand at minimal cost, recent production growth and seasonality considerations included specific regions in the states of Victoria and Queensland. Growing areas for lettuce in Victoria and Queensland are approximately 900km from the Sydney market by road.

Data was collected from two field farms in Sydney (LF1 and LF2), a hydroponic farm in Sydney (LF3) and a larger field farm in Victoria (LF4). Hydroponic growing (LF3) represented a low technology growing system, with no climate control and simple plastic shadecloth structures. Additional data will be collected from Queensland in the near future. Data was collected for iceberg, cos and/or baby cos lettuce. Process inputs captured within the system boundary for each farm, and calculated emissions to the compartments of air, soil and water included those identified in Figure 2.

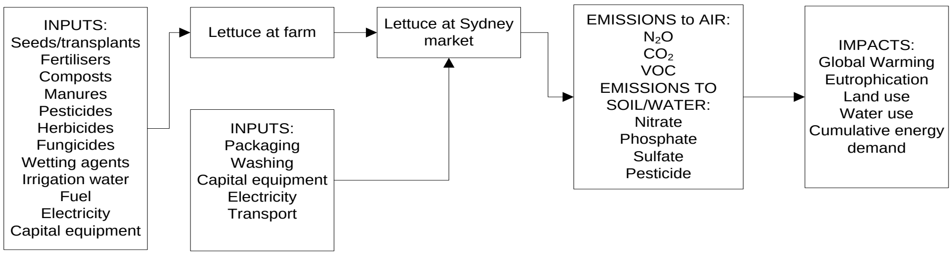


Figure 2. Process inputs captured within the system boundary for farms.

2.4 Urban systems

According to Sydney's Metropolitan Plan (NSW Department of Planning 2010) there are two dominant forms of urban housing: greenfield and infill. Greenfield requires large tracts of peri-urban land to be converted en masse into detached low density housing. Infill development requires the purchase and demolition of existing, dated, structures, and the construction of new, mid to high-rise apartment complexes in inner suburban areas. From scenario 1 the model requires the impacts from 1 ha of greenfield development to be estimated, while scenario 2 requires impacts from 1 ha (equivalent) of infill development (equivalency based on housing a certain population). An initial challenge was how to model infill development given that infill does not directly replace peri-urban horticultural land as greenfield housing systems do, on a per hectare basis. By calculating the number of houses per hectare in a greenfield development, and the associated persons per hectare based on population statistics, the corresponding number of infill style apartments to house the same population could be determined. For the purposes of this study, the typical greenfield house is a $250m^2$ detached single level brick veneer house, with concrete slab and tiled roof on a $550m^2$ block of land. An average of 2.5 persons lives in a typical greenfield house in Australia (ABS in Crawford and Fuller 2011). Using this information, and determining the number of houses (18.18 houses per ha) and population per hectare (45.45 persons per ha) for a typical greenfield development, the equivalent number of infill apartments was ascertained. The average population of a mid-rise apartment in inner suburban Australia is estimated at 1.6 per apartment (Crawford, 2011). Therefore 28.4 apartments would be needed to house the same population as 1 ha of greenfield housing, for those scenarios requiring infill development. It was assumed that these apartments were housed in one mid-rise (four or five storeys) high-density apartment block.

Inventory for greenfield and infill housing systems was compiled from various local sources to represent construction, operation and transportation to the extent that they were able to be characterised. Hybrid Input-Output (I-O) energy data for embodied, operational and transportation energy was adapted from a contemporary study of alternative housing types in Australia by Crawford and Fuller (2011). As hybrid I-O studies calculate on an energy basis, to obtain a more complete picture of environmental impacts, the embodied energy for each of the two housing styles was broken down into constituent material quantities such that the total embodied energy reflected the value being modelled from Crawford and Fuller (2011). Representative material quantities for greenfield housing construction were obtained from a residential case study of similar housing in Crawford (2011). For infill style housing material inputs were obtained from a bill of quantities for a concrete apartment building (OECD, 1999). Operational energy was represented as the NSW average energy mix of coal fired electricity 90% and gas 10% (Dart Energy, 2011). Transportation energy was represented as the proportional mix of train versus car travel taken by households in each urban system, with one passenger car allocated to each household.

2.5 Combining agricultural and urban impacts

To combine impacts, reference is made to scenarios 1 and 2 (Figure 2) respectively. For scenario 1, total environmental impacts included the addition of one hectare of greenfield housing with the amount of land in the marginal location associated with the same production yield as displaced Sydney farms. Scenario 1 therefore combined impacts from 1ha of greenfield housing with impacts from LF4. Total impacts for scenario 2 included the addition of 1 ha of a Sydney farm, plus 1 ha-equivalent of infill housing. For scenario 2, the environmental impacts from 1 ha equivalent of infill housing was combined with the average impacts from LF1 and LF2. As these scenarios each require lettuce to be modelled at the Sydney market, inputs included, in addition to farm processes, post-harvest operations such as washing of lettuce, coolroom operation, transportation and fabrication of capital equipment for transport, sheds and coolrooms.

3. Results

3.1 Lettuce at farm and lettuce at Sydney market

Preliminary data are indicating that field grown lettuce (LF1, LF2 and LF4) exhibit similar GWPs with results ranging between 1.5 to $1.9\text{x}10^4$ kg CO_2-e/ha (0.25-0.35 kg CO_2-eq per kg lettuce). Hydroponic lettuce (LF3) exhibited significantly larger GWP at $10.9\text{x}10^4$ kg CO_2-e/ha (0.58 kg CO_2-eq per kg lettuce), primarily due to the contribution to farm inputs from electricity. LF1, LF2 and LF3 exhibited increases in GWP of 6 to 9 percent when factoring inputs to deliver the lettuce to the Sydney market. GWP for interstate farm LF4 increased 40 percent to $2.1\text{x}10^4$ kg CO_2-e/ha.

3.2 Urban housing style comparison and combined impacts

Figure 3 illustrates the GWP for both greenfield and infill urban housing systems (without any horticultural implications) and provides comparison to the impacts from each of the farms. Embodied, operational and transportation energy are included in the total per hectare results for each of the urban systems. Per hectare results for greenfield housing systems exhibited the largest GWP at 44 kg CO_2-e/ha, over twenty-fold

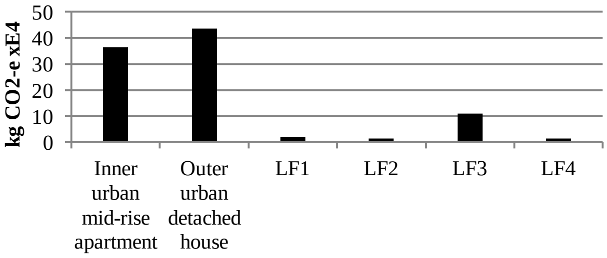


Figure 3. GWP for the two housing systems and farms LF1 to LF4, per hectare.

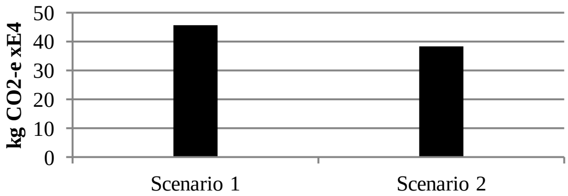


Figure 4. GWP for Scenarios 1 and 2, per hectare

higher than for field lettuce systems. Figure 4 shows the combined impacts for the scenarios 1 and 2, illustrating that scenario 1, greenfield housing with displaced lettuce production, has a GWP approximately 74 t ha^{-1} higher than for scenario 2.

4. Discussion

Within the agricultural system at both farm and market levels, comparisons can be made both intra-region and inter-region. The larger commercial field farm, LF4, although producing with better economies of scale, exhibited only a marginally smaller GWP at farm level than local Sydney field farms, highlighting that smaller farms can compare favourably with respect to environmental impacts (see Figure 3). For lettuce at the Sydney market, supply chain activities (including road transport) impacted GWP most negatively for the interstate farm, effectively reversing the order of the GWP impacts for the three field farms, resulting in LF4 having the higher GWP impact for lettuce at the Sydney market. The benefit of this method is that both field and supply chain impacts have been assessed, with proximity to market having a positive impact on emissions abatement. Emissions for LF3 were relatively high as a result of infrastructure requirements and electricity demand, which in NSW is predominantly coal fired. Farm level data has illustrated that hydroponic systems are not necessarily more environmentally friendly, however, if an alternate electricity mix were available for which emissions could be significantly reduced, the hydroponic farm would see the most dramatic benefits.

Avoiding the use of 'average' data obtained a more realistic analysis of environmental impacts. Preliminary results are indicating dramatic reductions in global warming potential when farm-level data is used compared to Australian average estimates. For example, results from this project are orders of magnitude less than average estimates calculated for horticultural sectors, where farm gross margin data was used to make GWP claims (O'Halloran et al., 2008). Confidence in the values reported in this project was assured by comparison to contemporary lettuce LCAs (Marton et al., 2010; Milà i Canals et al., 2008). Even with truncation errors of up to 87% reported as part of process LCA (Crawford 2011), the benefits of using local data over use of averages are dramatic.

Collecting data for urban systems presented a greater challenge than collection from farms. Building LCA literature was oriented to individual buildings or building components, with a lack of literature providing detailed bills of quantities for the housing styles to be modelled or data on housing systems. Land preparation, demolition, waste, infrastructure installation and revised bills of quantities will be included in a subsequent phase of the project in order to improve authenticity and reduce uncertainty in the greenfield and infill models. Although inter-farm differences in GWP of up to 25 percent were observed for field lettuce, GWP impacts within the agricultural production systems proved less significant compared to the larger impacts of the urban systems (Figure 3). With the urban system dominating impacts, the interstate farm LF4 would need to demonstrate significant improvements in environmental impacts to trade-off against greenfield expansion, which as shown in scenario 1, was not observed. This indicates that the region where lettuce is produced is less relevant to overall GWP impacts than the urban housing model implemented. The focus becomes less on how best to produce lettuce and more on how housing systems can be leveraged to support emissions abatement.

A further challenge within the method included the need to use European derived Ecoinvent data in SimaPro due to absence of representative Australian processes. The electricity mix in European processes using nuclear energy may give more conservative impacts than using energy from the dominantly coal fired mix in Australia. Future work will include further data collection from regional locations, improving the urban modelling and analysing the consequences of urbanising Sydney's horticultural lands in a resource constrained and climatically uncertain future. Applying time limits to future scenarios, by which time marginal suppliers and technologies may have changed, is a further challenge for management.

5. Conclusion

The benefit of CLCA at the urban-agricultural intersection has been to provide a new perspective on peri-urban land-use change. Scenarios for producing lettuce versus producing housing were evaluated for GWP. Preliminary analysis for the two scenarios modelled has illustrated that, although differences were observed within the agricultural system, it is the urban housing system dominating GWP impacts. Policy focus on urban housing systems for greenhouse gas abatement in the regional context is recommended.

6. References

ABS 2010. Value of Agricultural Commodities Produced, Australia, 2008-09, 'Table 1 VACP - All Commodities - National, Australia', data cube: excel spreadsheet, cat. no. 7503.0, (Data Cube All commodities, by Australia, State and Statistical Division).

Commonwealth of Australia, Canberra. Viewed 5 October 2010 <http://www.abs.gov.au/AUSSTATS/abs@.nsf/DetailsPage/7503.02008-09>.

ARUP, 2010a. Greenhouse Gas Emissions Snapshot: 2036 Metropolitan Plan Review. NSW Department of Planning, Sydney.

ARUP, 2010b. World Leading Practice in Emissions Reductions - Spatial Policy. NSW Department of Planning, Sydney.

Brandao, M. 2010. Food or fuel: how to best use land, in: Proceedings of LCA food 2010, vol. 1, pp. 59-64.

Camagni, R., Gibelli, M.C., Rigamonti, P. 2002. Urban mobility and urban form: the social and environmental costs of different patterns of urban expansion. Ecol. Econ. 40, 199-216.

Choy, D.L., Sutherland, C., Gleeson, B., Sipe, N., Dodson, J., 2008. Change and Continuity in Peri-urban Australia: Peri-urban Futures & Sustainable Development. Griffith University, Brisbane.

Condon, P.M., Mullinix, K., Fallick, A., Harcourt, M., 2010. Agriculture on the edge:strategies to abate urban encroachment onto agricultural lands by promoting viable human-scale agriculture as an integral element of urbanization. Int. J. Agr. Sustain. 8, 104-15.

Crawford, R., Fuller, R., 2011. Energy and greenhouse gas emissions implications of alternative housing types for Australia. Paper presented to State of Australian Cities (SOAC) National Conference, Melbourne, Australia, 29 November - 2 December 2011. Viewed 28 March 2012 <http://soac2011.com.au/full-papers-list.php>.

Crawford, R.H., 2011. Life cycle assessment in the built environment. Spon Press, Oxon, USA.

Dart Energy, 2011. NSW Market Overview. Dart Energy. Viewed 11 May 2012, <http://www.dartenergy.com.au/page/Worldwide/Australia/CBM_Australia/>.

Duffy, A., 2009. Land use planning in Ireland - a life cycle energy analysis of recent residential development in the Greater Dublin Area. Int. J. Life Cycle Ass. 14, 268-77.

Fuller, R., Crawford, R., 2011. Impact of past and future residential housing development patterns on energy demand and related emissions. J. Hous.and the Built Environ. 26, 165-83.

Garnaut, R., 2008. The Garnaut Climate Change Review. Viewed 9 May 2012, <http://www.garnautreview.org.au/2008-review.html>.

Houston, P., 2005. Re-valuing the fringe: Some findings on the value of agricultural production in Australia's peri-urban regions. Geogr. Res. 43, 209-23.

Marton, S., Kägi, T., Wettstein, D., 2010. Lower global warming potential of cucumbers and lettuce from a greenhouse heated by waste heat, in: LCA Food Conference Proceedings, Bari, Italy, vol. 1, pp. 531-6.

Milà i Canals, L., Muñoz, I., Hospido, A., Plassman, K., McLaren, S., 2008. Life Cycle Assessment (LCA) of domestic vs. imported vegetables. Centre for Environmental Strategy, University of Surrey, Guildford, UK.

NSW Department of Planning 2010, Metropolitan Plan for Sydney 2036, NSW Department of Planning, viewed 19 May 2010, <http://www.metroplansydney.nsw.gov.au/>.

O'Halloran, N., Fisher, P., Rab, A., 2008. Vegetable Industry Carbon Footprint Scoping Study Discussion Paper 4: Preliminary estimation of the carbon footprint of the Australian vegetable industry, VG08107: Viewed 19 April 2011, <http://www.dpi.qld.gov.au/documents/PlantIndustries_ FruitAndVegetables/Hort-Fruit-Drought-Carbon-Report4.pdf>.

OECD, 1999. 505.4 Bill of quantities for an apartment: Eurostat-OECD PPP Programme 1999 Comparison of construction prices for residential buildings, vol. rev. 12-89. Viewed 1 May 2012, <http://siteresources.worldbank.org/ICPINT/Resources/oecd_apartmt_guidelines.DOC.>.

Rossi, B., Marique, A.F., Reiter, S., 2012. Life-cycle assessment of residential buildings in three different European locations, case study. Build. Environ. 51, 402-7.

van Veenhuizen, R., Danso, G., 2007. Profitability and sustainability of urban and peri-urban agriculture. FAO, Rome.

Soil, climate and cropping system effects on N_2O accounting in the LCA of faba bean and cereals

Pietro Goglio[1, 2,*], Caroline Colnenne-David[3,4], Claudia Di Bene[2], Simona Bosco[2], Patricia Laville[1], Romain Roche[1], Giorgio Ragaglini[2], Thierry Doré[4,3], Marco Mazzoncini[5,6], Benoit Gabrielle[1], Enrico Bonari[2]

[1] UMR INRA-AgroParisTech Environnement et Grandes Cultures, 78850 Thiverval-Grignon, France
[2] Landlab, Institute of Life Sciences, Scuola Superiore Sant'Anna, Via Santa Cecilia 3, 56127 Pisa, Italy
[3] INRA, UMR 211, BP 01, 78850 Thiverval-Grignon, France
[4] AgroParisTech, UMR 211, BP 01, 78850 Thiverval-Grignon, France
[5] Dipartimento di Agronomia e Gestione dell'Agroecosistema, Faculty of Agriculture, University of Pisa, Via San Michele degli Scalzi, 2, 56124 Pisa, Italy
[6] CIRAA Interdepartimental Centre for Agroenvironmental Research, via Vecchia di Marina, 6, 56122 San Piero a Grado Pisa, Italy
[*] Corresponding author. E-mail: *pgoglio@grignon.inra.fr, p.goglio@sssup.it*

ABSTRACT

Greenhouse gas (GHG) emissions from soils cause uncertainties within Agricultural LCA. N_2O affects global warming and is estimated with IPCC guidelines, agroecosystem models or direct measurements. CERES-EGC model was used to estimate N_2O emissions from faba bean and winter cereals grown in two trials (ICC and CIMAS) with different climates. Model outputs were compared with IPCC estimates. Simulated N_2O emission patterns showed that emissions can be independent from fertiliser application dates or rates. This was due to soil moisture, farming practices such as fertiliser applications and tillage. Results showed the IPCC procedure estimated higher annual cereals emissions of 740 g N_2O-N ha^{-1} y^{-1} than simulation results and a lower estimation of 304 g N_2O-N ha^{-1} y^{-1} for faba bean. Results revealed inclusion of climate, soil properties and management resulted in major variations of N_2O emissions which CERES-EGC was able to capture. Thus, model estimates may increase accuracy of soil GHG emission in Agricultural LCA.

Keywords: agricultural LCA, N_2O emissions, CERES-EGC model, cereals, faba bean

1. Introduction

Agriculture contributed for 18% to the Greenhouse gases (GHG) emissions of France in 2010 and 7% for the Italian emission in the same year. Despite this contribution in both countries, the agricultural sector was responsible for 87% of nitrous oxide (N_2O) total emissions in France and 69% in Italy (CITEPA, 2012; Romano et al., 2012). N_2O has a global warming potential 298 times higher than carbon dioxide (Forster et al., 2007).

N_2O emissions can be estimated with IPCC emissions factors, agroecosystem models or by direct field measurements. IPCC methodology is based on emissions factors and accounts for mineral and organic fertiliser applications. It also considers crop residues decomposition, organic matter mineralisation and other sources of organic nitrogen (N) (De Klein et al., 2006). N_2O emissions from cropping systems may also be directly measured in field trials using micrometeorological systems or chambers with either automatic or manual sampling (Hénault and Germon, 1995, 2000; Laville et al., 1999, 2011). N_2O field emissions are hard to detect due to their low concentration variations; therefore high sensibility and costly instruments are needed. Thus, direct measurements are expensive and time consuming and their use is restricted to a limited set of experimental fields (Hastings et al., 2010; Laville et al., 2009; Rochette and Eriksen-Hamel, 2008).

Agroecosystem models simulate most of the processes governing N_2O emissions and their controls. They may compare favourably with field observations for assessing GHG emissions, but require detailed inputs and are complex to run (Del Grosso et al., 2008). For N_2O emissions, DAYCENT, DNDC, CERES-EGC models have been extensively used for various types of ecosystems (Chen et al., 2008; Del Grosso et al., 2008; Gabrielle and Gagnaire, 2008; Lehuger et al., 2009; Hastings et al., 2010). CERES-EGC was adapted from the CERES suite of soil-crop models (Jones and Kiniry, 1986), with a focus on the simulation of environmental outputs such as N_2O and NO to the atmosphere. It was tested and calibrated for these environmental impacts mostly with data coming from Europe, with reasonable success due to a Bayesian calibration procedure that proved effective when testing the model against independent field data (Gabrielle and Gagnaire, 2008; Lehuger et al., 2009, 2011).

N_2O emissions, like all the other GHG emissions from soils, are a major source of uncertainties within Agricultural LCA, such as soil erosion, soil organic matter dynamics, biodiversity estimation (Guinée et al., 2009) and ecotoxicity (Margni et al., 2002; van Zelm et al., 2009). Indeed, in recent years LCA has been used for a wide range of agricultural systems and proved to be an effective assessment to evaluate environmental impacts (Nemecek et al., 2011a, 2011b; Goglio et al., 2012). Notwithstanding in most cases soil born emissions were estimated with the IPCC method which often proved to be less accurate than agroecosystem models (Gabrielle and Gagnaire, 2008).

Here, the agro-ecosystem model CERES-EGC was used to estimate direct N_2O emissions from faba bean, wheat, durum wheat and barley crops grown in two experimental trials with different climates in France and Italy, considering also residues decomposition. Model outputs for direct N_2O emissions were then compared with IPCC estimates in order to evaluate possible discrepancies between the two methods (De Klein et al., 2006).

2. Methods

2.1. Field trials

The first agronomical trial is the ICC (Innovative Crops with Constraints) trial, which started in 2008 next to the NitroEurope trial, in the Paris area (less than 10 m of distance, geographic coordinates for the ICC trial N 48.842° E 1.954°, NitroEurope N 48.844° E 1.951°). The soil is a silt loam with 24 g kg^{-1} soil of soil organic matter. Each cropping system, in the ICC trial, has three randomly distributed replicates (4000 m² in area), bearing each year three different crops in rotation. N_2O measurements were carried out on two cropping systems: (1) the PHEP system, which aims at reaching altogether High Environmental Performances and Productivity and (2) a cropping system under a greenhouse gas mitigation constraint (50%-GHG), aiming at halving GHG emissions compared to the PHEP system, both by increasing C sequestration in the soil and decreasing N_2O emissions. Aside from GHG emissions, this system has to meet the same environmental criteria as the PHEP system. The latter cropping system was designed on the basis of the following agronomical principles: to reduce N fertiliser inputs by including legumes in the rotation; to use crop varieties with high N use efficiency and a high level of resistance to diseases; to lengthen the duration of crop rotations; to increase crop diversity. Instead in the 50%-GHG cropping system, GHG reduction is further enhanced with the following farming practices: reducing systematically mineral nitrogen fertiliser inputs with legumes both as cover crop and cash crop, extensively using cover crops to decrease the accumulation of soil nitrate and subsequent emissions of N_2O from nitrate denitrification, utilising no tillage practices to avoid mineralisation, introducing rapeseed (*Brassica napus* L.) to reduce nitrate leaching and the ensuing emissions of N_2O. Both cropping systems include as main crops faba bean (*Vicia faba* var *minor* (Harz) Beck), wheat ((*Triticum aestivum* L.), however wheat was absent during the 2010-2011 season in the 50%-GHG system) and barley (*Hordeum vulgare* L.).

The second trial is CIMAS (Conventional *vs* Integrated Management Agricultural System) that was established near Pisa (N 43.662° E 10.299°). It has been extensively described in Nassi o Di Nasso (2011), here main aspects are summarised. The soil is a clay loam with 20 g kg^{-1} of organic matter. The trial is composed of two cropping systems (High Input, HI and Low Input, LI) which differ in fertiliser application rates and frequencies, herbicide (frequency and amount of active ingredients), pesticide treatment (frequency and amount) and tillage operations (reduction of deep tillage operations). The rotation included faba bean and durum wheat (*Triticum durum* L.) (Nassi o Di Nasso et al., 2011).

2.2. CERES-EGC model and IPCC method accounting

CERES-EGC comprises sub-models for major processes governing the cycles of water, carbon and nitrogen in soil-crop systems. A physical sub-model simulates the transfer of heat, water and nitrate down the soil profile, as well as soil evaporation, plant water uptake and transpiration. The soil profile is constituted of seven soil layers (Gabrielle et al., 1995; Lehuger et al., 2009). A biological sub-model simulates crop growth and phenology. Nitrogen uptake is computed through a supply/demand scheme. For N-fixing crops, a specific module was developed for nitrogen supply through nitrogen fixation. A micro-biological sub-model simulates turnover of organic matter. Direct field emissions of CO_2, N_2O, NO and NH_3 into the atmosphere are simulated with different gas modules (Lehuger et al., 2009). CERES-EGC runs on a daily time step, and hence was provided/fed with daily rainfall; mean air temperature and Penman potential evapo-transpiration data taken from weather stations located less than 10 km away from the field trials under study for both trials. Residue inputs for the model, including nitrogen content were accounted according to Lehuger et al., (2009).

Simulated N_2O emissions were cumulated and attributed following two methods: (i) from harvest of the previous crop to harvest of the current crop, according to Topp et al., (2011); (ii) from harvest of the previous crop to sowing of the following crop or the end of the year. The second option should allow for taking into account residues decomposition emission. In fact, most of the N_2O emissions occur during autumn/winter when soil moisture tends to be higher due to rainfall events which allow denitrification of nitrate produced from residues mineralisation (Barton et al., 2011; Laville et al., 2011).

The IPCC method is based on emission factors which take into account a series of farming practices and land uses. It considers both direct and indirect emissions from managed soils. The IPCC methodology estimates N_2O emissions using human-induced net N additions to soils (e.g. in particular N content in synthetic or organic fertilisers, deposited manure, crop residues, sewage sludge) (Table 1), or of mineralisation of N in soil organic matter following drainage/management of organic soils, or cultivation/land-use change on mineral soils (e.g. Forest Land/Grassland/Settlements converted to Cropland)(De Klein et al., 2006). Residues amounts were estimated using the shoot to root ratio and harvest index from measured yields for both the ICC and the CIMAS trial (Giardini, 2001). Nitrogen content of residues was accounted using factors from literature with measured yields (De Klein et al., 2006; Jensen et al., 2010).

3. Results

Simulation results (Figure 1 and 2) showed clearly that most of the emissions do occur during fall following residues decomposition. The effect of N fertiliser input is very much limited, considering most of the emission during spring period laid below 5 g N_2O-N ha^{-1} d^{-1}. This is the resulting effect of very dry spring which kept the soil dry (less than 35.2 m^3 m^{-3}). Indeed, the peaks following fertiliser application (Abdalla et al., 2009; Saggar, 2010), which might have been expected during this period, did not occur.

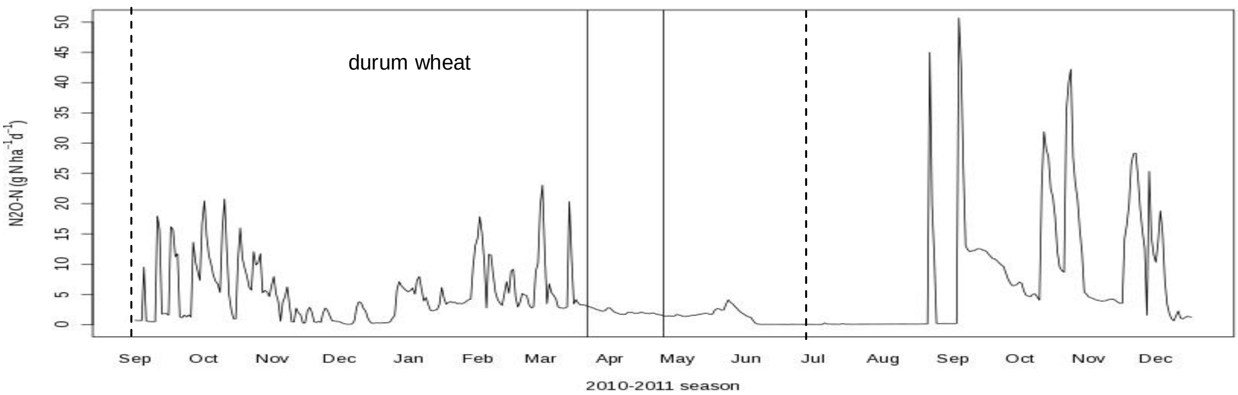


Figure 1. Model results of the analysed period for durum wheat in CIMAS high input system (from previous crop harvest to the end of the year); vertical lines: full: date of fertiliser application, dashed: harvest dates

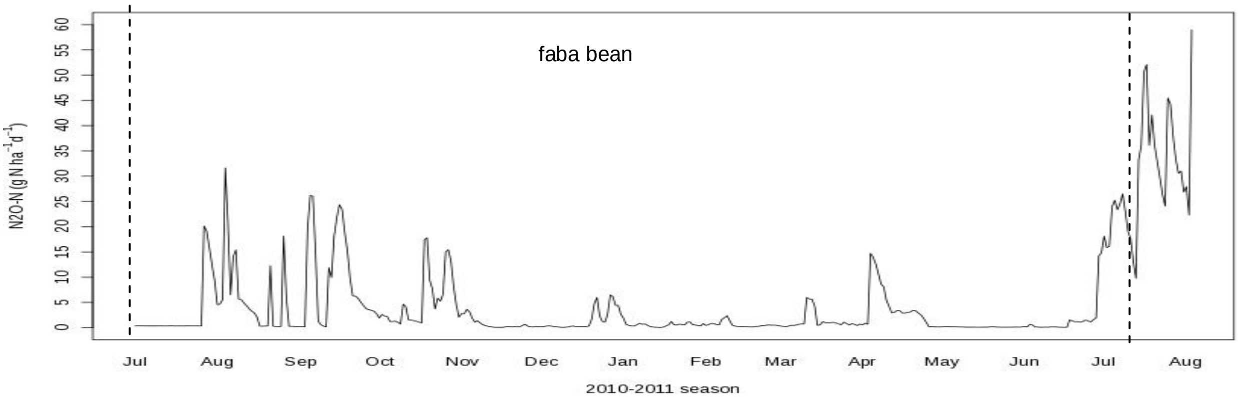


Figure 2. Model results of the analysed period for faba bean in ICC 50%-GHG system (from previous crop harvest to sowing of the following); dashed vertical lines: harvest dates

During fall, a series of emission peaks occurred which were due to the coupled effect of residues decomposition and relatively high soil moisture. This is a common pattern for every crop present in the two trials, including faba bean plots where no N fertiliser was applied. During fall-winter, rainfall events are more frequent and intense causing soil moisture to remain high (>20 m^3m^{-3} for some sampling dates) allowing denitrification (Figures 1 and 2).

Results showed that the IPCC procedure estimated higher annual emissions for cereals of 740 g N_2O-N ha^{-1} y^{-1} (428 g N_2O-N ha^{-1} y^{-1} including post harvest periods) on average compared to simulation results and lower estimation of 304 g N_2O-N ha^{-1} y^{-1} (609 g N_2O-N ha^{-1} y^{-1} including post harvest periods) for faba bean (Table 1). Disregarding crops, the highest difference between CERES-EGC simulations and IPCC estimates was due to the ICC trial with 361 g N_2O-N ha^{-1} y^{-1}, excluding residues; while more limited discrepancies resulted from the CIMAS trial (170 g N_2O-N ha^{-1} y^{-1} as difference of absolute values between model and IPCC figures). Including residues with both methods, the opposite trend occurred with a greater gap for the CIMAS trial than the ICC but with lower values for both (67 g N_2O-N ha^{-1} y^{-1} and 159 g N_2O-N ha^{-1} y^{-1} for ICC and CIMAS, respectively).

Table 1: Amounts of N fertiliser applied and estimated N_2O-N emissions with the two methods: CERES-EGC and IPCC (cumulated values over the season were reported over a year period)

Crop	**Trial**	**Cropping system**	**N applied with fertiliser (kg N ha^{-1})**	**CERES-EGC N_2O-N emissions (g ha^{-1} y^{-1})**	**CERES-EGC N_2O-N emissions (g ha^{-1} y^{-1}) including post harvest period**	**IPCC N_2O-N emissions (g ha^{-1} y^{-1})**
Winter Wheat	ICC	PHEP	90[a]	1791	1976	2380
Barley	ICC	PHEP	60[a]	533	723	1542
Barley	ICC	50%GHG	80[a]	312	460	1755
Faba bean	ICC	PHEP	0	1163	1480	659
Faba bean	ICC	50%GHG	0	1246	1871	512
Durum Wheat	CIMAS	HI	92[b] +52[c]	1481	1987	2140
Durum Wheat	CIMAS	LI	46[b]+26[c]	1363	1894	1361
Faba bean	CIMAS	HI	0	727	876	776
Faba bean	CIMAS	LI	0	707	836	681

[a] NH_4NO_3 N content 33.5%; [b] Urea N content 46%; [c] NH_4NO_3 N content 26%

Cumulated values estimated here showed clearly that the PHEP emissions were on average 1.08-1.32 fold higher than the 50%-GHG, while HI emissions were 1.06-1.36 fold higher than LI with CERES-EGC, depending on the estimation method. Higher differences were reported for cereal (on average for all the methods 24%) than for faba bean (mean value among the three methods 11%) (Table 1).

4. Discussion

N_2O emission patterns obtained here were in agreement with previous works carried out in Grignon both with model and continuous chamber measurements (Laville et al., 2011). The pattern obtained for the simulation underlined the effect of crop residues on the overall N_2O emission which is confirmed by other studies (Barton et al., 2011; Petersen et al., 2011). The effect of residues is also dependent on the farming practice of the following crop, therefore farm management effects on N_2O emissions can be better understood when the evaluation is carried out at cropping system scale (Mazzoncini et al., 2008; Petersen et al., 2011; Goglio et al., 2012).

Regarding fertiliser application, it appears from our elaboration that this farming practice had little effect on N_2O emissions, probably in result of a very dry period. Indeed, for the ICC trial, during spring 2011, the total precipitations during February-May period was 88 mm, when monthly average should be around 50 mm. The same happened for the CIMAS trial, where the total cumulated precipitations in March-May period were 154 mm concentrated in March; while commonly they are mostly concentrated in April and they are 30 mm greater (Nassi o Di Nasso et al., 2011). Under these conditions, N_2O peak emissions would not occur because of reduced bacterial activity (Bateman and Baggs, 2005; Szukics et al., 2010). Indeed, N_2O is emitted when the soil is wet with WFPS (Water Filled Pore Space) comprised between 55%-70%, depending on soil type (Bateman and Baggs, 2005; Szukics et al., 2010). Due to limited rainfall, these soil moisture conditions were partially met (<35.2 m^3 m^{-3} as maximum value for each sampling date during spring for both trials).

Model results and IPCC estimates diverged more for faba bean (54% and 106% with and without residues, respectively) than cereals (41% and 22% with and without residues, respectively) due to lack of emission peaks. Indeed, most of cumulated emissions are due to emission peaks (Abdalla et al., 2009) which often occur after fertiliser application (Saggar, 2010). Other results showed a limited difference of 8% for cereals between DNDC and IPCC method, while here the average discrepancy was higher than 22% (Hastings et al., 2010). Notwithstanding, CERES-EGC proved to be very effective in N_2O estimation with field data (Lehuger et al., 2009). This suggests its use within the context of agricultural LCA would increase assessment precision in comparison to IPCC method estimate which is very much desirable for LCA of agricultural systems (Guinée et al., 2009). Nevertheless, input requirements to run the model might be a limitation of its use for LCA of agricultural systems (Del Grosso et al., 2008).

5. Conclusion

The present work showed that CERES-EGC simulations both with and without residue decomposition presented a major difference in N_2O emissions estimates with the IPCC procedure. These variations were mainly due to the interaction of soil and climatic conditions with farming practice. In particular, the N fertiliser application resulted in a low effect on N_2O emissions in dry conditions. On the contrary, most of the emissions were due to residue decomposition.

Despite its limits due to data needed to run the model, CERES-EGC was able to take into account the interaction between farming practices, climate and soil characteristics. Thus its use will improve precision in the estimation of environmental impacts due to the cultivation phase of agricultural LCA.

6. References

Abdalla, M., Jones, M., Ambus, P., Williams, M., 2009. Emissions of nitrous oxide from Irish arable soils: effects of tillage and reduced N input. Nutr. Cycl. Agroecosyst. 86, 53–65.

Barton, L., Butterbach-Bahl, K., Kiese, R., Murphy, D.V., 2011. Nitrous oxide fluxes from a grain-legume crop (narrow-leafed lupin) grown in a semiarid climate. Glob. Chang. Biol. 17, 1153–1166.

Bateman, E.J., Baggs, E.M., 2005. Contributions of nitrification and denitrification to N2O emissions from soils at different water-filled pore space. Biol. Fertil. Soils 41, 379–388.

Chen, D., Li, Y., Grace, P., Mosier, A.R., 2008. N2O emissions from agricultural lands: a synthesis of simulation approaches. Plant Soil 309, 169–189.

CITEPA, 2012. Rapport Nationale d'inventaire pour la France au titre de la Convention cadre des Nations Unies sur les Changements Climatiques et du Protocol de Kyoto. CITEPA Centre Interprofessional Technique d'Etudes de la Pollution Atmosphérique, Paris.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. Changes in Atmospheric Constituents and in Radiative Forcing, in: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, and New York, pp. 129–234.

Gabrielle, B., Gagnaire, N., 2008. Life-cycle assessment of straw use in bio-ethanol production: A case study based on biophysical modelling. Biomass Bioenerg. 32, 431–441.

Gabrielle, B., Menasseri, S., Houot, S., 1995. Analysis and field-evaluation of the CERES models water-balance component. Soil Sci. Soc. Am. J. 19, 1403–1412.

Giardini, L., 2001. Agronomia generale, ambientale e aziendale. Pàtron editore, Bologna.

Goglio, P., Bonari, E., Mazzoncini, M., 2012. LCA of cropping systems with different external input levels for energetic purposes. Biomass Bioenerg. 42, 33–42.

Del Grosso, S.J., Halvorson, A.D., Parton, W.J., 2008. Testing DAYCENT Model Simulations of Corn Yields and Nitrous Oxide Emissions in Irrigated Tillage Systems in Colorado. J. Environ. Qual. 37, 1383–1389.

Guinée, J.B., Heijungs, R., Voet, E., 2009. A greenhouse gas indicator for bioenergy: some theoretical issues with practical implications. Int. J. Life Cycle Assess. 14, 328–339.

Hastings, A.F., Wattenbach, M., Eugster, W., Li, C., Buchmann, N., Smith, P., 2010. Uncertainty propagation in soil greenhouse gas emission models: An experiment using the DNDC model and at the Oensingen cropland site. Agr. Ecosyst. Environ. 136, 97–110.

Hénault, C., Germon, J., 1995. Quantification de la dénitrification et des émissions de protoxyde d'azote (N2O) par les sols. Agronomie 15, 321–355.

Hénault, C., Germon, J.C., 2000. NEMIS, a predictive model of denitrification on the field scale. Eur. J. Soil Sci. 51, 257–270.

Jensen, E.S., Peoples, M.B., Hauggaard-Nielsen, H., 2010. Faba bean in cropping systems. Field Crop. Res. 115, 203–216.

Jones, C.A., Kiniry, J.R., 1986. Ceres-N Maize: a simulation model of maize growth and development., A&M University Press, College Station. ed. Temple, TX: Texas.

De Klein, C., Novoa, R.S.A., Ogle, S., Smith, K.A., Rochette, P., Wirth, T.C., McConkey, B.G., Mosier, A., Rypdal, K., Walsh, M., Williams, S.A., 2006. Chapter 11: N2O Emissions from Managed Soils, and CO2 Emissions from Lime and Urea Application, in: Gytarsky, M., Hiraishi, T., Irving, W., Krug, T., Penman, J. (Eds.), 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC International Panel on Climate Change, Geneva, pp. 4:11.1–11.54.

Laville, P., Jambert, C., Cellier, P., Delmas, R., 1999. Nitrous oxide fluxes from a fertilised maize crop using micrometeorological and chamber methods. Agricultural and Forest Meteorology 96, 19–38.

Laville, P., Lehuger, S., Loubet, B., Chaumartin, F., Cellier, P., 2011. Effect of management, climate and soil conditions on N2O and NO emissions from an arable crop rotation using high temporal resolution measurements. Agr. Forest. Meteorol. 151, 228–240.

Lehuger, S., Gabrielle, B., Laville, P., Lamboni, M., Loubet, B., Cellier, P., 2011. Predicting and mitigating the net greenhouse gas emissions of crop rotations in Western Europe. Agr. Forest Meteorol. 151, 1654–1671.

Lehuger, S., Gabrielle, B., van Oijen, M., Makowski, D., Germon, J.-C., Morvan, T., Hénault, C., 2009. Bayesian calibration of the nitrous oxide emission module of an agro-ecosystem model. Agr. Ecosys. Environ. 133, 208–222.

Margni, M., Rossier, D., Crettaz, P., Jolliet, O., 2002. Life cycle impact assessment of pesticides on human health and ecosystems. Agr. Ecosyst. Env. 93, 379–392.

Mazzoncini, M., Di Bene, C., Coli, A., Antichi, D., Petri, M., Bonari, E., 2008. Rainfed Wheat and Soybean Productivity in a Long-Term Tillage Experiment in Central Italy. Agron. J. 100, 1418–1429.

Nassi o Di Nasso, N., Bosco, S., Di Bene, C., Coli, A., Mazzoncini, M., Bonari, E., 2011. Energy efficiency in long-term Mediterranean cropping systems with different management intensities. Energy 36, 1924–1930.
Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011a. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agr. Syst. 104, 217–232.
Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., Schaller, B., Chervet, A., 2011b. Life cycle assessment of Swiss farming systems: II. Extensive and intensive production. Agr. Syst. 104, 233–245.
Petersen, S.O., Mutegi, J.K., Hansen, E.M., Munkholm, L.J., 2011. Tillage effects on N2O emissions as influenced by a winter cover crop. Soil Biol. Biochem. 43, 1509–1517.
Romano, D., Arcarese C, Bernetti A, Caputo A, Condor D R, Contaldi M, De Lauretis R, Di Cristofaro E, Federici S, Gagna A, Gonella B, Lena, F., Liburdi R, Taurino E, Vitullo M, 2012. Italian Greenhouse Gas Inventory 1990-2012 National Inventory Report 2012. ISPRA Institute for Environmental Protection and Research, Varese.
Saggar, S., 2010. Estimation of nitrous oxide emission from ecosystems and its mitigation technologies. Agr. Ecosyst. Environ. 136, 189–191.
Szukics, U., Abell, G.C.J., Hödl, V., Mitter, B., Sessitsch, A., Hackl, E., Zechmeister-Boltenstern, S., 2010. Nitrifiers and denitrifiers respond rapidly to changed moisture and increasing temperature in a pristine forest soil. FEMS Microb. Ecol. 72, 395–406.
Topp, C.F.E., Tolkamp, B.J., Houdijk, J.G.M., Stephen, K.L., Kyriazakis, I., 2011. Environmental analysis of pig production systems: the production of home-grown proteins. Presented at the Nitroeurope 2011 conference, Edimbourgh.
van Zelm, R., Huijbregts, M., Posthuma, L., Wintersen, A., van de Meent, D., 2009. Pesticide ecotoxicological effect factors and their uncertainties for freshwater ecosystems. Int. J. Life Cycle Assess. 14, 43–51.

Using a model-based LCA to explore options for reducing national greenhouse gas emissions from crop and livestock production systems

Eric Audsley[1,*], Mike Wilkinson[2]

[1] Centre for Environmental Risks and Futures, School of Applied Sciences, Cranfield University, Bedford, MK43 0AL, UK.
[2] School of Biosciences, University of Nottingham, Sutton Bonington Campus, Loughborough, Leicestershire, LE12 5RD, UK..
[*] Corresponding author. E-mail: *e.audsley@cranfield.ac.uk*

ABSTRACT

Agriculture has a devolved commitment to reduce national greenhouse gas emissions (GHGE). Using a model-based LCA we explored GHGE-reducing options in systems used to produce twelve crop and seven livestock commodities. With a functional unit of kg of product, GHGE differences between crops reflect differences in yield. Metabolisable energy (ME) or crude protein (CP) could be used, but deriving an economic value of £8.6/GJ ME and £0.62/kg CP, leads to a relatively consistent 2.6 kg CO_2e/£ nutrient value. Potential GHGE reductions ranged from 2% (sugar beet) to 15% (cereals) with husbandry changes, and 4 to 12% with increased crop yields. The best alternative livestock systems reduced GHGE ranging between 7% (beef from the dairy herd) and 21% (extensive sheep meat). Half of the options reduce national production hence increase imports. No-till increases pesticide use. Overall, improvements in productivity and efficiency of resource use are the best options for reducing GHGE.

Keywords: policy, global warming potential, crops, livestock

1. Introduction

Governments have made international commitments to reduce greenhouse gas emissions (GHGE) and the United Kingdom government has set a target of an 80% reduction in emissions of GHGE by the year 2050 compared to the baseline of 1990 (Office of Public Sector Information, 2011). These have been devolved to equivalent commitments in each sector of the economy including agriculture. This paper examines the effects on GHGE of implementing theoretically a range of agronomic and livestock husbandry options in conventional systems of food production operated on farms in the UK with the sub-objective of not reducing the food produced within the UK.

2. Methods

A range of UK crop and livestock production systems was studied using the Cranfield system model based agricultural life-cycle analysis (LCA) (Williams et al., 2006). Burdens are expressed in terms of the functional unit, in this case per kg of product fresh weight, per MJ of edible energy or per kg edible protein at the farm gate. The GHGE from post farm gate processing of crops and livestock products are not included in this analysis. This model-based LCA approach includes the impact of changes within the farm system, for example a decrease in fertiliser input reduces crop yield per hectare and long-term soil nitrogen. Equally, an increase in the crop yield from plant breeding requires additional fertiliser input. A change in GHGE therefore represents the total effect of a change on the farming system. The methods and data inputs to the LCA model have been described in detail for the production of bread wheat, oilseed rape and potatoes in England and Wales by Williams et al., (2010).

Ten UK cropping systems were included in the present study to cover the range of major agricultural food crops, the range in soil types, and a range of contrasting agronomic practices. Typical cropping systems were defined in relation to soil texture, soil cultivation technique, straw incorporation, irrigation and the average total input of nitrogen (N) per hectare (Table 1). The analysis also includes two non-UK feed crops – soya beans and maize grain. The typical systems and their emissions were considered as baseline (2005) values for agricultural GHGE. GHGE are expressed as a global warming potential (GWP_{100}) in tonnes CO_2 equivalent (CO_2e) per unit of product, using a 100 year time frame and the GWP values for gases from the Intergovernmental Panel on Climate Change (IPCC, 2006). For each system, emissions of nitrous oxide (N_2O) were calculated using the IPCC Tier 1 methodology (IPCC, 2006). Other emissions such as carbon dioxide (CO_2) and methane (CH_4) were calculated systematically by considering each aspect of the system in turn. Emissions associated with the production of imported fertilisers were calculated and included in the analysis. The systems models determined the new long-term steady state for the soil, but as the soil was in steady state, no contribution was assumed for changes in the concentration of soil carbon. The proportion of soil types nationally (Table 1) remained as a fixed constraint. The typical composition of each crop product in terms of concentration of dry matter, energy and crude protein is shown in Table 3.

Table 1. Typical values for soil, cultivation and nitrogen input for crop systems, Williams et al., 2006.

Crop	Soil type (%)			Cultivations (%)			Straw (%)	Irrig'n (%)	Total N (kg/ha)	Type of N fertiliser (% of total)	
	Clay	Loam	Sand	Plough	Red'd tillage	No-till				Ammonium nitrate	Urea
Winter bread wheat	34	48	18	57	41	2	25	0	219	80	20
Winter feed wheat	34	48	18	57	41	2	25	0	204	82	18
Winter barley	22	54	24	57	41	2	85	0	163	82	18
Spring barley	9	75	16	57	41	2	100	0	123	82	18
Winter oilseed rape	43	29	28	50	45	5	0	0	204	69	31
Sugar beet	7	82	12	100	0	0	-	0	122	96	4
Main-crop potatoes	7	82	12	100	0	0	-	56	191	96	4
Second-early potatoes	7	82	12	100	0	0	-	48	171	96	4
Field beans	39	33	28	57	43	0	0	0	0	-	-
Soya beans	30	28	42	27	53	20	-	0	0	-	-
Maize grain	30	28	42	30	58	12	0	0	134	90	10
Forage maize	55	16	29	57	41	2	-	0	212	90	10

Table 2. Examples of the livestock systems modelled for each commodity (from Williams et al., 2006)

Sector	Milk	18 to 20 month beef	Cereal beef	Upland suckler beef	Lowland suckler beef	Upland sheep	Lowland sheep	Pig meat	Poultry meat	Eggs
System	Autumn calving	Spring-born, dairy-bred	Dairy cross-bred bulls	Spring calving, grass finishing	Autumn calving, winter finishing	Cross-bred hill ewes	Cross-bred from upland	Indoor heavy bacon	Housed	Housed layers
Days housed	190	180	399	182	182	0	30	126	42	385
Concentrates (kg DM)	2047	960	2398	579	792	76[4]	80[4]	366	4.9	52
Forage[1] (kg DM)	6792	2281	120	4982	4840	1018	902	-	-	-
Live weight gain (kg/day)	-	0.90	1.23	0.88	1.03	0.17	0.19	0.56	0.06	-
Output (kg/year)	7850	285	276	232[3]	225[3]	60[5]	63[5]	-	-	14.8
Slaughter Liveweight (kg)	-	565	535	565	595	41	44	109	2.4	-
Age at slaughter (months)	-	19	13	20	18	7 to10	6 to 9	6.3	1.5	-
Feed conversion ratio, [2] kg DM/kg milk or gain	1.13	6.23	5.14	10.7	10.2	18.2	15.6	2.89	1.76	3.06
Longevity of breeding females (years)	3.2	-	-	7	6.5	4.2	4.5	2.5	-	1.1
Lambs finished off grass						0.35	0.58			
Manure as slurry (%)	88	18	18	0	0	0	0	35	0	25[6]

[1] Grazing and conserved forage. [2] kg total feed DM/kg milk, weight gain, or output. [3] Live weight of calf at weaning. [4] Includes concentrates for finishing store lambs. [5] Per ewe. [6] Proportion with belt-cleaned cages, remainder on deep cages.

Livestock systems were modelled as a series of component systems within each commodity. There is a wide range in the total period of time the animals are housed. Similarly there is a wide range between systems in terms of the intensity of feed inputs, in output of animal products and in feed conversion ratios. Thus the range of modelled milk production systems included autumn and spring calving with different intensities of concentrate feeding. Sheep systems included hill, upland and lowland, pure and cross-bred flocks, with transfers of ewes and lambs between the systems and the option of early lambing. Beef production included suckler and dairy-sourced calves with different levels of finishing based on their levels of grass and concentrates. The main components are shown in Table 2 for a range of systems. Inputs of concentrate and forage DM refer to the complete system and include both the dam and her offspring.

3. Results and discussion

3.1. Crops

Typical GHGE are shown in Table 3. The range in GHGE between crops is considerable, with oilseed rape and sugar beet having the highest and lowest emissions per tonne of crop fresh weight, respectively. With a functional unit of kg of product, differences between crops in GHGE per kg product reflected differences in yield per hectare. Standardising potato and sugar beet yields to 860 g DM/kg fresh weight to make them comparable with the cereal crops produces values of 0.59, 0.44 and 0.20 kg CO_2e kg^{-1} for main-crop potatoes, second early potatoes and sugar beet, respectively. Forage maize had the lowest GHGE per kg of the cereal crops because, being harvested in its entirety, it had a substantially higher yield per hectare than the other crops, though of lower quality. Options to unify crops are to use metabolisable energy (ME) or crude protein (CP) as the functional unit. This provides apparently more consistent GHGE per unit, but crops that produce mainly ME (sugar and potatoes) have a very low GHGE per unit ME, whereas crops which produce a high concentration of protein have high GHGE per unit ME. GHGE per kg CP were higher than average for potatoes and sugar beet and lower than average for field and soya beans and forage maize. From the market price of all the crops (excluding potatoes), it can be estimated by regression that the economic value of a unit of ME is £8.6/GJ and CP is £0.62/kg, leading to a relatively consistent 2.6 kg CO_2e/£ nutrient value with a smaller range. Nitrogen fixing crops are slightly better and high nitrogen crops slightly worse.

Table 3. Greenhouse Gas Emissions (GHGE) of different crops and the effect of different functional units

Crop	Yield	DM	ME	CP	GHGE, kg CO_2e per			
	t/ha	g/kg	MJ/kg DM	g/kg DM	kg	GJ ME	kg CP	£ value
Winter bread wheat	7.7	860	13.6	130	0.51	0.044	4.56	3.00
Winter feed wheat	8.1	860	13.6	116	0.46	0.039	4.61	2.83
Winter barley	6.5	860	13.2	123	0.42	0.037	3.97	2.57
Spring barley	5.7	860	13.2	116	0.38	0.033	3.81	2.38
Winter oilseed rape	3.2	930	23.1	212	1.05	0.049	5.33	3.42
Sugar beet	63	220	13.2	68	0.04	0.015	2.87	1.25
Main-crop potatoes	52	200	13.3	93	0.14	0.053	7.53	2.57
Second-early potatoes	48	200	13.3	93	0.10	0.038	5.38	2.90
Field beans	3.4	860	13.3	298	0.51	0.045	1.99	1.98
Soya beans	2.4	860	14.5	415	0.70	0.056	1.96	2.13
Maize grain	7.2	860	13.8	102	0.38	0.032	4.33	2.43
Forage maize (DM)	11.2	280	11.0	101	0.30	0.027	2.97	1.91

DM=dry matter, ME=metabolisable energy, CP=crude protein. Concentrations of DM, ME, CP from Thomas, 2004

Table 4. Predicted yields and greenhouse gas emissions (GHGE) for typical crop systems and for agronomic options to reduce greenhouse gas emissions.

Crop	Typical yield[1]	Yield with agronomic options[2] to reduce GHGE	Reduction in yield (%)	Typical system	No-till	No-till + no straw incorporation	No-till + no straw incorporation + 20% reduced N	20% increase in crop yield per hectare
	(tonnes fresh weight ha^{-1})			GHGE (kg CO_2e kg^{-1} product fresh weight)				
Winter bread wheat	7.7	7.0	9	0.51	0.50	0.46	0.42	0.48
Winter feed wheat	8.1	7.2	11	0.46	0.45	0.41	0.38	0.43
Winter barley	6.5	5.9	9	0.42	0.40	0.39	0.36	0.39
Spring barley	5.7	5.2	9	0.38	0.35	-	0.32	0.36
Winter oilseed rape	3.2	2.9	9	1.05	-	1.03	0.97	0.95
Sugar beet	63.0	58.1	8	0.043	-	-	0.04	0.04
Main-crop potatoes[4]	52.0	49.6	5	0.14	-	-	0.13	0.13
Second-early potatoes[5]	48.0	46.1	4	0.10	-	-	0.10	0.09
Field beans	3.4	3.3	4	0.51	0.46	-	0.46	0.46
Soya beans	2.4	2.3	2	0.70	0.64	-	0.64	0.61
Maize grain	7.2	6.7	7	0.38	0.37	-	0.33	0.36
Forage maize	11.2[3]	10.8[3]	4	0.30	0.29	-	0.26	0.29

[1] Systems as described in Table 1. [2] See text. [3] t DM ha^{-1}. [4] Cool-stored until May: weighted cooling energy applied. [5] No storage.

Four crop husbandry options to reduce GHGE were considered: i) 20% decrease in applied N; ii) no-till (cereals and legumes only); iii) no straw incorporation and iv) irrigate all potatoes. Fresh weight yields for the typical cropping systems and for the options to reduce GHGE are shown in Table 4. These options to reduce GHGE also reduce crop yields but to a relatively small extent ranging from 5% or less for potatoes, field beans, soya beans and forage maize to between 7 and 11% for the other crops. Irrigation of main-crop potatoes was associated with a progressive reduction in GHGE, from 0.14 kg CO_2e kg^{-1} without irrigation to 0.13 kg CO_2e kg^{-1} with 100% irrigation – a 6% decrease. However as the majority of potato crops are either irrigated or do not need irrigation, the overall potential reduction in GHGE is probably only about 1%.

Although no-till is associated with reduced crop yield compared with ploughing, there is a reduction in GHGE, mainly as a result of lower primary energy use. An exception is oilseed rape where the change to 100% no-till is associated with an increase in GHGE of 0.04 kg CO_2e kg^{-1} because the relatively high yield penalty (13%) outweighs the saving on primary energy. The restrictions of applying the IPCC Tier 1 emission factors mean that the model assumes there were no changes in soil N_2O emissions for different cultivation techniques. However there may be an increase in N_2O compared to the typical system because of increased soil anaerobic conditions (Robertson et al., 2000).

The main source of GHGE due to incorporating straw into soil is N_2O emission from soil during the winter. The model determines the long-term steady state system for all processes. This includes nitrogen from the rotation, nitrate leaching and soil organic matter. Hence incorporating (or not incorporating) straw continues indefinitely, so the soil is in steady state and there is no contribution from the change in the soil organic matter. In the transition period, soil organic matter would be reduced giving a release of CO_2 which the benefit of reduced N_2O would take some years to counteract, and *vice-versa*. The magnitude of the effect of a change away from straw incorporation depends on the proportion of straw incorporated for each crop.

A reduction in the total quantity of N input is associated with decreased primary energy use and reduced emissions of N_2O since under the Tier 1 IPCC methodology the emission factor for N_2O is a fixed percentage (1%) of total N applied (IPCC, 2006). An effect of reducing total N input is that the concentration of N in the crop is also reduced. This reduces the likelihood of bread wheat grain being of a suitable quality for bread-making. A switch to a variety with a higher inherent protein content might be feasible, but these varieties are also lower-yielding (HGCA, 2011). Reduced N content is unlikely to be consequential in the case of potatoes and sugar beet as it is not a quality criterion for these crops. Reductions in total N input were analysed to determine an appropriate level which might reduce GHGE by more than crop yields to give a net environmental benefit per unit of crop produced. An average reduction of 20% in total N input produced a net GHGE benefit for all crops and was therefore considered to be the most appropriate option. Progressive decreases in total N not only reduce crop yields and soil nitrate concentrations but also reduce emissions of ammonia.

Where all three agronomic options were appropriate to the crop, reduced N had the greatest effect on GHGE. The combined effect of the options on the percentage reduction in GHGE was lowest for sugar beet (2%) and highest for the cereal crops (average 15% reduction). The percentage reduction in GHGE was similar for the two potato crops (3%), and was also similar for the two grain legumes (9%).

The output of the major grain crops has increased steadily over the years and there is undoubtedly scope for them to be increased further - for example through improved plant breeding and crop health (see review by Godfray et al., 2010). Table 4 shows GHGE per kg product were significantly reduced by a theoretical increase in yield of 20%. The system models increase the fertiliser N input to the crops to balance the increased N off-take. For crops other than cereals and forage maize the effect on GHGE of a 20% increase in yield alone was greater than the combined effects of the agronomic options, ranging from a 5% reduction for main-crop potatoes to a 14% reduction in GHGE for soya beans.

3.2 Livestock

Differences between semi-intensive (18 to 20 month) and intensive (cereal) dairy beef, between upland and lowland suckler beef, and between upland and lowland sheep were small in terms of GHGE/kg of product at the farm gate, in agreement with farm-based studies in the UK (EBLEX, 2010; QMS, 2011). GHGE from livestock systems are average values for each sector - milk, dairy beef, suckler beef, sheep meat, pig meat, poultry meat and eggs (Table 5). Milk production has apparently lower GHGE per kg product, but this is due to the fact that milk is largely water. On a dry matter basis primary energy use for milk production is similar to that of poultry production, reflecting the energetic efficiency of converting feed into milk rather than live weight. However GHGE is always higher for ruminants due to the methane emitted during rumination. GHGE per kg product are higher for suckled beef and sheep meat production than for beef produced from calves born in the dairy herd (dairy beef) and non-ruminant systems, reflecting the relatively high overhead feed cost of the breeding female (Table 3). Differences in GHGE between the meat production systems per unit of edible energy and edible protein are similar to those per kg fresh product, with suckler beef having the highest, and poultry meat the lowest GHGE per MJ of edible energy and per kg edible protein.

The best alternative system in terms of reduced GHGE compared to the combined typical systems was identified for each livestock sector using the Cranfield model (Table 5). Alternative systems were defined using the model with the most extreme feasible improvement in each factor in order to estimate the maximum potential for reducing GHGE. By increasing fertility (number of successful conceptions per female inseminated), fecundity (number of offspring per breeding female in sheep) and longevity (number of years in production), the overhead costs of rearing herd and flock replacements are reduced.

Using the system model identified a problem with the statements “increase annual milk yield” and “increase daily growth rate”. Both can be achieved by having a larger animal. Thus a cow which is 10% larger will be expected to require 10% more food for maintenance, give 10% more milk and require 10% more food because of that milk and will have the same GHGE per kg milk. There are three options: 1) larger animal 2) same size of animal giving more milk but eating more food for that milk 3) same size of animal giving more milk and eating no more food. Increased annual milk output should also not be confused with yield per lactation, which can be increased by having a longer calving interval. Similarly improving feed conversion ratio (FCR) – defined as kg feed (at constant dry matter) per kg weight gain, milk or eggs (at constant dry matter) – makes more efficient use of resources. Increased daily live weight gain and lower age at slaughter may save resources but an animal that is simply larger may achieve a greater daily live weight gain but consume *pro-rata* more feed with no improvement in its feed conversion ratio. The analysis presented here does not distinguish between methods to improve FCR. In some cases, diet re-formulations may improve FCR but increase the environmental burdens of feed production and not reduce GHGE.

Table 5. Estimated GHGE for typical and alternative livestock systems

Sector	Typical system kg CO_2e per kg product[1]	MJ edible energy	kg edible protein	Best alternative system	GHGE from alternative system kg CO_2e per kg product[1]	% Redn
Milk	1.0	0.4	30.6	Autumn-calving cows, housed 190 days/year. 8000 litres per year, 7 lactations per cow. 15% crude protein housed diet based on maize silage.	0.89	12
Dairy beef	8.5	1.0	49.5	Lower protein and lower forage diet, housed throughout lifetime.	7.95	7
Suckler beef	15.9	1.9	90.0	Extended grazing. Spring calving. High genetic merit cow for fertility and calf growth.	14.1	12
Sheep meat	14.6	1.6	69.3	Extensive. Ewes of high genetic merit for fecundity and longevity. No housing.	11.5	21
Pig meat	4.0	0.7	19.7	High fertility and piglet growth. Sows and weaners outdoors. Finishing indoors on slurry system, applied slurry immediately incorporated into land.	3.49	14
Poultry meat	2.7	0.3	14.2	Housed. Immediate incorporation of applied manure into land. FCR as for top 10% of sector.	2.54	7
Eggs	3.0	0.5	23.2	Housed, slurry, under-floor drying of manure, covering of manure store, Immediate incorporation of applied manure into land. FCR as for top 10% of sector.	2.57	13

[1] Whole milk and eggs, bone-in carcase weight

The potential reductions in GHGE range from 7% for dairy beef and poultry meat to 21% for sheep meat. The major factors affecting GHGE per unit of milk are annual yield per cow, longevity and reduced protein diets. The best alternative milk production system is longevity at 7 lactations per cow rather the current average of 3.2 lactations per cow. The best alternative beef production system uses calves from the dairy herd. The use of sexed semen in dairy herds was examined as a possible option. There was little effect on the total number of male and female dairy-bred calves available for beef. The scope for reducing GHGE from suckler beef systems is limited by the relatively low output of beef per breeding female per year. However, suckler beef herds make use of grassland that is not good enough for milk production or arable cropping (Wilkinson, 2011). Overall feed conversion ratio is substantially poorer than that of the monogastric livestock systems. The best alternative suckler system comprises spring-calving suckler cows with extended grazing (i.e. minimal housing) to minimise N_2O emissions from farmyard manure. The best alternative pig production system comprised sows of high genetic merit for fertility and piglet growth, sows and weaners kept outdoors and indoor finishing system with manure as slurry. Greater emissions of N_2O from the outdoor system are more than offset by the reduction in methane which would otherwise be produced from stored manure or slurry. There is, however, an increased risk of nitrate leaching from the outdoor system compared to fully-housed systems. Poultry production is relatively efficient compared to other livestock sectors, and there was relatively little scope for reductions in GHGE. The best alternative system of poultry meat production is indoor-housed as is the case with egg production, which conflicts with modern welfare preferences.

Criteria other than GHGE need to be taken into account in determining the best options. Half of the cropping options reduce national production of the commodities, which conflicts with the requirement not to increase imports. Apart from the country's food security, increased imports affect global agriculture and carry the risk of increased deforestation with consequent severe increases in GHG emissions. No-till increases pesticide use. Whilst decreasing nitrogen fertiliser reduces nitrate leaching, increased yields from crop breeding had negligible effect on nitrate leaching even though the model requires that nitrogen input is increased pro-rata with yield. Overall the results indicate that improvements in productivity and efficiency of resource use are the best options for reducing GHGE per unit of product, without other deleterious effects.

4. Conclusions

Of the options found to reduce crop GHGE, reduced fertiliser N and increased yield per hectare were the most significant, giving reductions in GHGE of between 5% and 15% compared to typical systems. Options found to reduce GHGE in livestock production were increased fertility, fecundity and longevity of breeding females, increased annual milk yield per dairy cow, improved FCR in meat animals and immediate incorporation of slurry following its application to land giving reductions of between 7 and 21%. However the best that is likely to be achieved overall is around a 10% improvement, in agreement with the aspiration of the UK Greenhouse Gas Action Plan (Agricultural Climate Change Task Force, 2010). There is scope to reduce GHGE in all sectors by applying existing knowledge.

5. Acknowledgement

The research on which this paper is based was funded by the Department for Environment, Food and Rural Affairs (DEFRA, projects IS0205, AC0208 and IS0222), whose support is gratefully acknowledged.

6. References

Agricultural Climate Change Task Force, 2010. Agriculture Industry GHG Action Plan: Framework for Action. www.nfuonline.com/Our-work/Environment/.../GHGAP-10Feb2010/ Accessed 17 Feb 2012.

English Beef and Lamb Executive (EBLEX), 2010. Testing the Water. The English Beef and Sheep Production Environmental Roadmap – Phase 2. EBLEX, Kenilworth, UK 47 p.

Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S and Toulmin, C. 2010. Food security: The challenge of feeding 9 billion people. Science 327: 812-818.

Home-Grown Cereals Authority (HGCA), 2011. HGCA Recommended List Winter Wheat 2012/13. HGCA, Kenilworth, UK.

Intergovernmental Panel on Climate Change (IPCC), 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4. Agriculture, Forestry and Other Land Use. Task Force on National Greenhouse Gas Inventories.

Office of Public Sector Information, 2011. Climate Change Act 2008. http://www.legislation.gov.uk/ukpga/2008/27/contents Accessed 4 January 2012.

Quality Meat Scotland (QMS), 2011 Estimating Greenhouse Gas Emissions from Scottish Livestock Enterprises. QMS, Ingelston, Scotland. 17 p.

Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse Gases in Intensive Agriculture: Contributions of Individual Gases to the Radiative Forcing of the Atmosphere. Science 289, 1922-1925

Thomas, C. (Ed.) 2004. Feed into Milk. A New Applied Feeding System for Dairy Cows. Feed Database. Nottingham University Press, Nottingham, UK. 68 p.

Wilkinson, J.M., 2011. Redefining efficiency of feed use by livestock. Animal 5: 1014-1022.

Williams, A.G., Audsley, E. and Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Cranfield University, Bedford, UK. 97p. http://randd.defra.gov.uk/

Williams, A.G., Audsley, E. and Sandars, D.L., 2010. Environmental burdens of producing bread wheat, oilseed rape and potatoes in England and Wales using simulation and system modelling. International Journal of Life Cycle Assessment 15: 855-868.

Improving estimates of life cycle nitrous oxide emissions from crop-based food products

Jeremy Wiltshire[1,*], Sarah Wynn[1], Roger Sylvester-Bradley[1], Daniel Kindred[1], Nigel Mortimer[2], Anna Evans[2], Charlotte Hatto[2], Onesmus Mwabonje[2]

[1] ADAS UK Ltd, Battlegate Road, Boxworth, Cambridge, CB23 4NN, UK
[2] North Energy Associates Ltd, Watsons Chambers, 5–15 Market Place, Castle Square, Sheffield, S1 2GH, UK
[*] Corresponding author. E-mail: *jeremy.wiltshire@adas.co.uk*

ABSTRACT

A research project called "Minimising nitrous oxide intensities of arable crop products" (MIN-NO) is addressing the scientific and practical challenges of minimising nitrous oxide emissions from UK arable cropping, and is exploring improved N_2O emissions factors. The average emission of greenhouse gases (GHGs) for 220 UK wheat crops was 405 kg $CO_2e\ t^{-1}$ based on the Intergovernmental Panel on Climate Change (IPCC) Tier 1 method (IPCC, 2006). The N_2O contribution to GHG emissions per tonne of wheat grain was approximately 72%, and the coefficient of variation for fertiliser-related N_2O emissions per tonne of wheat grain was large, at 31.9%. Results suggest that soil organic matter status had a large effect on N_2O emissions.

The uncertainty ranges in the IPCC N_2O emissions factors gave a range of GHG emissions per tonne of grain that is more than double the value for the GHG emissions assessed using IPCC default emissions factors.

Keywords: food product, nitrous oxide, N_2O, greenhouse gas, crop

1. Introduction

Most non-leguminous arable crops in Northern Europe respond positively to applications of nitrogen (N) fertilisers. The Intergovernmental Panel on Climate Change (IPCC) provides a method for assessing emissions of the greenhouse gas (GHG) nitrous oxide (N_2O) that occur as a consequence of applying N fertiliser: at the national scale the default value for applied N lost directly as N_2O is 1% (range: 0.3–3%), for all crops regardless of management practices.

Using this method within the UK, N_2O emissions have been estimated to contribute 6.1% (111,640 t N_2O) to the total of GHG emissions, with 78% of all N_2O emissions originating from agricultural practices (MacCarthy et al., 2011). In the absence of more specific information IPCC methods and their associated emissions factors have been adopted for lifecycle analysis at a product scale (BSI, 2011) and, using this approach, the contribution of primary food production to lifecycle GHG emissions of food products has typically been estimated at around 50%, sometimes more (Wiltshire et al., 2008). Thus, N_2O makes an important contribution to the lifecycle GHG emissions of food products.

A study in the UK is addressing the scientific and practical challenges of minimising N_2O emissions from UK arable cropping. The project is called "Minimising nitrous oxide intensities of arable crop products (MIN-NO)" and objectives include the following:

1. To gauge the importance of and variability in N_2O emissions associated with crop products;
2. To determine a more robust relationship between N_2O emission and the rate of mineral N fertilisers applied, both during crop growth and from crop residues;
3. Through expert estimation and debate, to identify practices which could lower the greenhouse gas emissions footprint of arable products such as bread, sugar, oils, peas, chicken, whisky and biofuels;
4. To assess how emissions might be estimated more accurately at farm and at national level.

A key hypothesis being tested in the MIN-NO project is that, because some N_2O emissions occur after crop N uptake, emissions relate to the balance between N supply and N uptake. This hypothesis is supported by evidence for a non-linear N_2O response to applied nitrogen fertiliser in corn crops (Hoben et al., 2011). Contrary to this, most current GHG accounting methods assume a direct proportionality between N fertiliser use and N_2O emissions from land, as agreed internationally by IPCC (2006). This assumption implies that large reductions in N fertiliser use and crop productivity are required to minimise the N_2O contribution to life cycle GHG emissions of crop products. However, if N_2O emissions were N-balance related, N amounts that minimise N_2O intensities would be similar to current use, and GHG mitigation would be compatible with sustained crop productivity (Figure 1).

The project objective to gauge the importance of and variability in N_2O emissions associated with crop products, requires collection of multiple field-by-field GHG emissions data to determine variability in on-farm N_2O emissions, and their contributions of to the GHG intensities of arable crop products. In this paper we present GHG emission assessments for wheat grain from multiple fields, using lifecycle assessment (LCA) methods, and show the extent of variability in emissions (especially N_2O) for the wheat grain product to help meet objective 1 of the project.

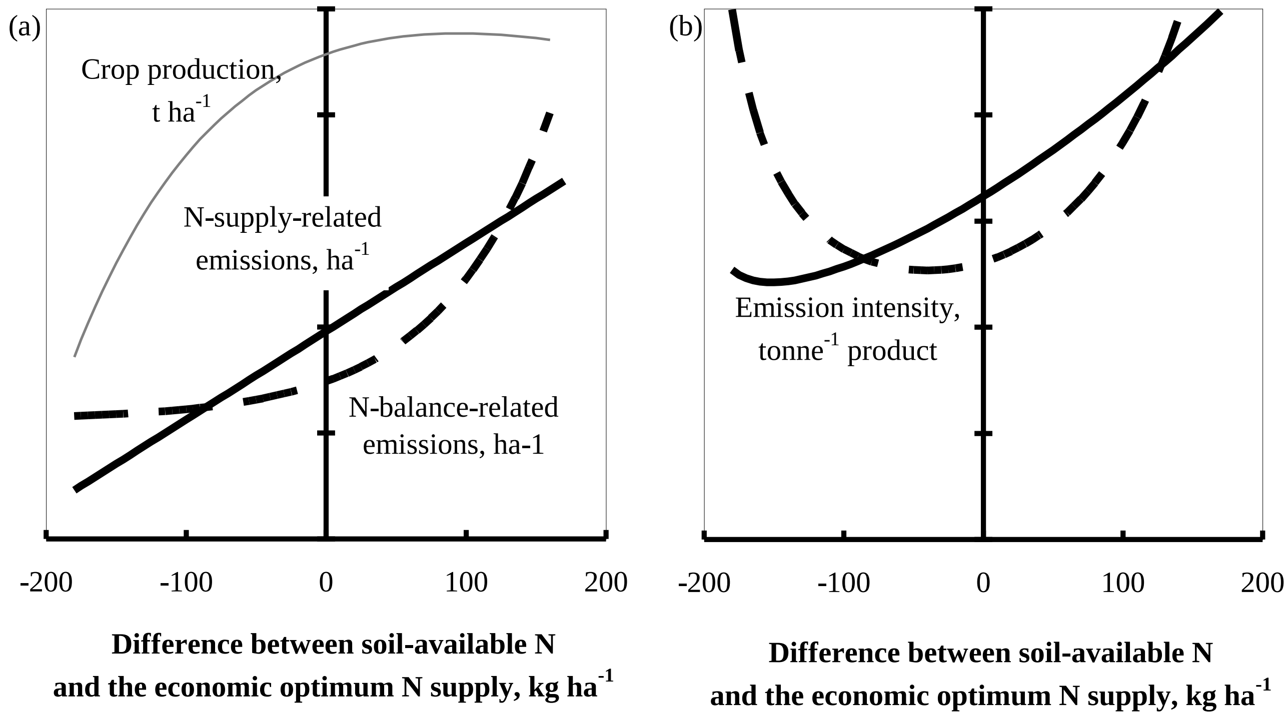


Figure 1. (a) Effects of N supply on crop production (thin grey line), and on N_2O emissions if related directly to N supply (bold continuous line; as estimated by the IPCC Tier 1 approach) or to the balance between N supply and crop N uptake (bold dashed line; as hypothesised here). (b) Consequent contrasting effects of N supply on N_2O emission-intensities of crop products for the IPCC (bold continuous line) and our hypothesised (bold dashed line) scenarios.

2. Methods

Emissions of GHGs have been assessed at an individual field scale for 220 wheat crops, of which 97 crops were grown for the animal feed or distilling markets (feed wheat) and 123 crops were grown for bread making (milling wheat). The crops were widely distributed across the wheat-growing regions of England and Scotland, and were grown in the years 2005 to 2011, with more than half of the crops grown in 2010 or 2011.

Data were provided by commercial partners in the MIN-NO project (see Acknowledgements), working with farmers. To provide the data, the farmers completed a questionnaire, and most received a visit to help them supply the data from their farm records. Important variables included: crop location, wheat variety, position in the farm crop rotation, soil type, type of cultivations or tillage, inputs of seed and crop protection chemicals, fertiliser applications, grain yield, fate of straw, and grain drying.

The GHG emissions were assessed using an LCA approach as given in PAS 2050:2011 (BSI, 2011). The system boundary was from ‘cradle to gate’, and included production and transport of raw materials (e.g. seed, chemicals, fertilisers), direct and indirect GHG emissions following application of fertilisers (including CO_2 fixed in the industrial production of urea, and indirect emissions related to NH_3 and NO_X), and GHG emissions from machine use and crop drying. The system boundary did not include GHG emissions from the production of capital goods (e.g. tractors and buildings). The PAS 2050 rules require that agricultural N_2O and CH_4 emissions should be calculated with the highest tier approach set out in the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006), or the highest tier approach employed by the country in which the emissions were produced. The method may be one of three: Tier 1, Tier 2 or Tier 3, which increase in their complexity and accuracy. We followed the Tier 1 approach (IPCC, 2006), as used to calculate the latest published UK agricultural greenhouse gas inventory, although this inventory was calculated using the revised 1996 IPCC guidelines and not the 2006 IPCC guidelines.

The IPCC Tier 1 methodology is simple and generalised, due to its intended initial wide scope of application. For example, the default emissions factor for direct soil emissions is 1.0% of total N applied lost as N_2O-N; and that for indirect N_2O losses following nitrate leaching is 0.75% of leached N lost as N_2O-N. These emissions factors have large uncertainty ranges: for direct soil emissions the range is 0.3 to 3.0%; and for indirect soil emissions following nitrate leaching the range is 0.05 to 2.5%. We have used these ranges from the IPCC (2006) guidelines to show the uncertainty in the emissions of N_2O per tonne of wheat.

The GHG emission assessments for multiple fields of wheat were disaggregated by the three main agricultural GHGs (CO_2, CH_4 and N_2O) and analysed using descriptive statistics to show the variability at the farm level.

3. Results

The mean of GHG emissions from 220 wheat fields was 405 kg CO_2e t^{-1}, with a range of 777 kg CO_2e t^{-1}. This range is large relative to the mean for both feed wheat and milling wheat (Figure 2).

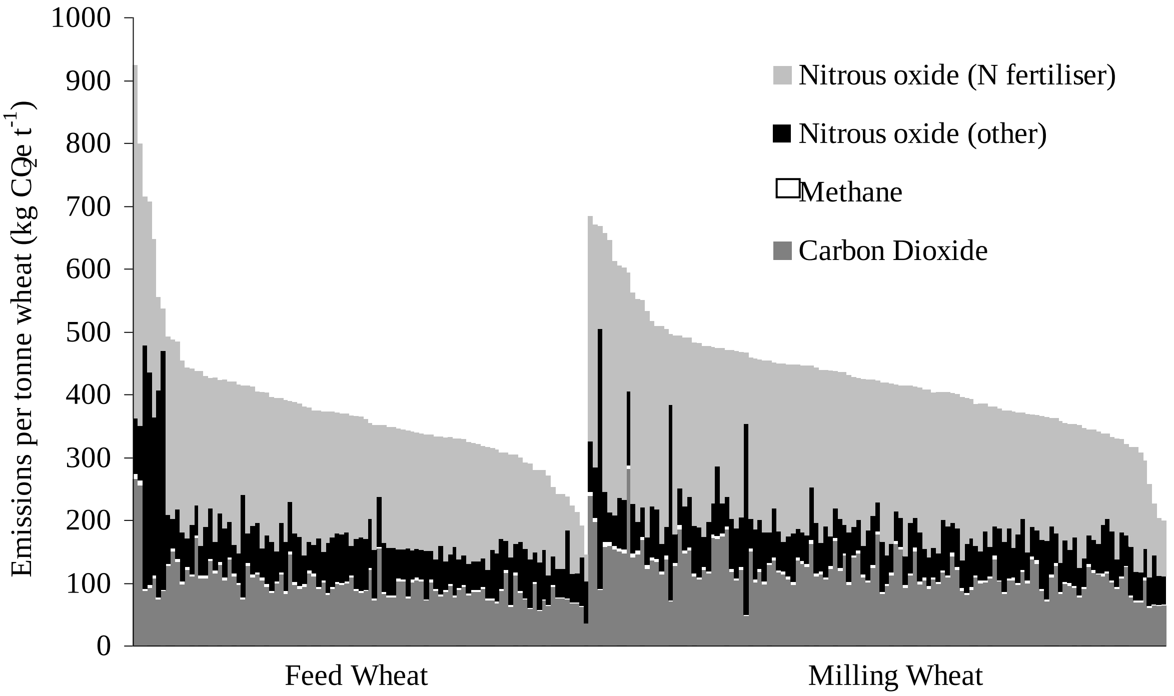


Figure 2. Greenhouse gas emissions (kg CO_2e t^{-1}) for individual fields of wheat, disaggregated to nitrous oxide from N fertiliser (manufacture and soil emissions), nitrous oxide from other sources (including from soil following incorporation of crop residues), methane and carbon dioxide. Data are divided into feed wheat (for animal feed) and milling wheat (for bread), and each of these groups is ranked by total GHG emissions per tonne of wheat grain.

Analysis of the dataset indicates that the variation in GHG emissions was influenced by many factors. There was a weak relationship between yield and GHG emissions (R^2 = 0.28), but the two fields with greatest GHG emissions also had the lowest two yields and accounted for more than one quarter of the range in total GHG emissions. Other factors affecting total GHG emissions included total N applied (R^2 = 0.21), and deisel use (R^2 = 0.28).

On average, the GHG emissions per tonne of wheat grain were dominated by N_2O (approximately 72% of total global warming potential (GWP) expressed as kg CO_2e), with important emissions of CO_2 (approximately 27%), and less than 1% of the GWP attributable to CH_4. The total of N_2O emissions from N fertiliser manufacture and from soil as a consequence of applying N fertiliser made up approximately 54% of the total CO_2e value. Other N_2O emissions were from soil following organic N applications (animal manures), crop residue incorporation, and release of N by organic soils. Table 1 shows the mean and variability for N_2O and CO_2 emissions expressed as kg CO_2e t^{-1}.

It can be seen from Figure 2 that fields with exceptionally large N_2O (other) emissions (i.e. excluding N_2O from N fertiliser manufacture and N-fertiliser-related soil emissions) are in the upper quartile of the total emissions. These fields had organic soils, leading to N_2O emissions from N in mineralised soil organic matter. The coefficients of variation (CVs) shown in Table 1 are greatest for N_2O (other) emissions (i.e. excluding N_2O from N fertiliser manufacture and N-fertiliser-related soil emissions), and the CV was 75.4% for all wheat crops, compared with 31.9% for fertiliser-related N_2O. This reflects the presence of outlying values in the N_2O (other) category due to the fields with organic soils.

Table 1. Means and coefficients of variation, for grain yield (t ha^{-1}) and GHG emissions (kg CO_2e t^{-1}; total, and CO_2 and N_2O components), and for all wheat crops, and subsets of feed wheat and milling wheat.

Statistic	Variable	All wheat	Feed wheat	Milling wheat
Mean	Yield (t ha^{-1})	8.7	8.9	8.6
	GHG emissions (kg CO_2e t^{-1})	405	378	427
	CO_2 (kg CO_2e t^{-1})	109	98	119
	Fertiliser-related N_2O (kg CO_2e t^{-1})	218	197	235
	Other N_2O (kg CO_2e t^{-1})	74	80	69
Coefficient of variation (%)	Yield (t ha^{-1})	15.5	15.8	15.1
	GHG emissions (kg CO_2e t^{-1})	25.5	30.3	20.7
	CO_2 (kg CO_2e t^{-1})	32.3	33.5	29.3
	Fertiliser-related N_2O (kg CO_2e t^{-1})	31.9	35.0	28.0
	Other N_2O (kg CO_2e t^{-1})	75.4	80.7	68.6

A source of uncertainty in estimating GHG emissions of farm products is the choice of emission factors for N_2O emissions. Figure 3 uses an example of GHG emission assessment for one UK wheat crop to compare use of the default IPCC emissions factor for N_2O emissions with the upper and lower limits of the uncertainty ranges for N_2O emissions factors given by IPCC (IPCC, 2006). The range between the total emissions using either the upper or the lower limits was 959 kg CO_2e t^{-1}, more than twice the total GHG emissions using the IPCC default emissions factors.

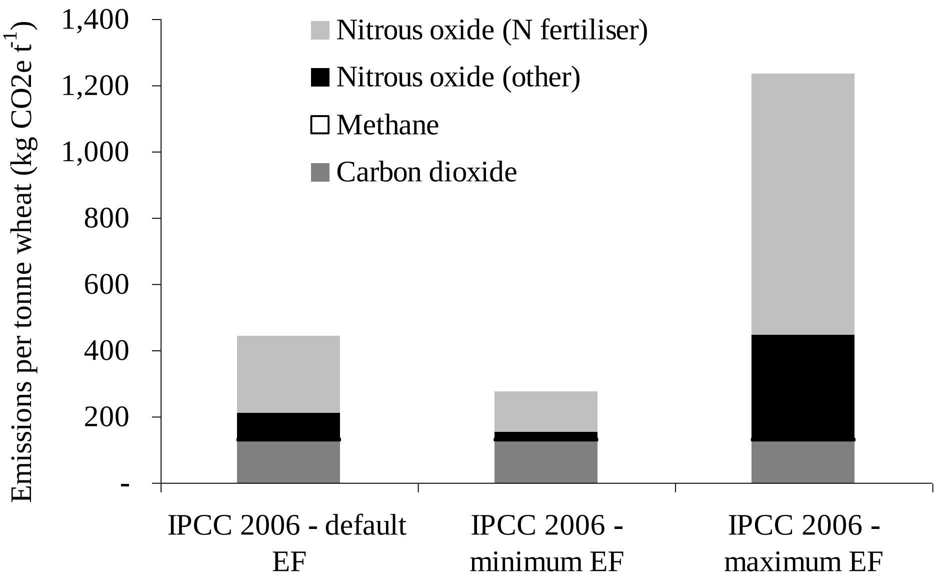


Figure 3. Greenhouse gas emissions (kg CO_2e t^{-1}) for one tonne of wheat at the farm gate, assessed using alternative emissions factors (EFs) from IPCC (2006). The emissions are disaggregated to show CO_2e values for nitrous oxide from N fertiliser (manufacture and soil emissions), nitrous oxide from other sources (including from soil following incorporation of crop residues), methane and carbon dioxide.

4. Discussion

Estimates of wheat GHG emissions indicate that N_2O is the dominant GHG, accounting for 72% of CO_2e per tonne of grain. This supports IPCC conclusions (De Klein et al., 2007) that GHG emissions associated with most crop products are dominated by N_2O. Because of this, variability and uncertainty in N_2O emissions have a large influence on variability and uncertainty in the total emissions for crop products at the farm gate, and for retailed products made from crops, such as bread.

Understanding the sources of variation in GHG emissions is useful to guide strategies for mitigation of emissions, by adoption of on-going improvements in efficiency of energy and resource use. Collection of more data and further data analysis in this project will provide more detail of the activities that lead to GHG emissions on farms. The analysis to date indicates that there are multiple sources of variation between crops, including yield, quantity of applied N fertiliser, soil type, and energy use. A small minority of fields had organic soils with high soil organic matter content, where mineralisation of organic matter led to release of plant-available N, and emission of N_2O, contributing to emissions variation between fields. The marked effect on GHG intensity of describing a soil as 'organic' implies that more finesse may be required at the field scale than at national scale in defining a soil's organic matter status; some low emission intensities were associated with low fertiliser N use on fields which might almost have been deemed 'organic'. However, it is interesting to note that the fields with largest emissions per tonne of grain, for feed wheat and milling wheat, did not have organic soils. The fields with highest emissions per tonne of grain tended to have low yields and/or high applications of N fertiliser.

Fig. 2 shows that the range of GHG emissions values was greater for feed wheat (486 kg CO_2e t^{-1}) than for milling wheat (777 kg CO_2e t^{-1}). Reasons for this are not clear, but we speculate that milling wheat crops are managed to meet a tighter product specification (especially for grain protein content), leading to less variation in agronomic practices.

Because it is technically difficult to measure emissions of N_2O from soil over the life-span of a crop, it is not practical to measure N_2O emission directly at an individual field scale, for multiple fields. Thus, an estimation method is necessary for field scale GHG accounting of crop products. The internationally accepted IPCC tier 1 method has a large uncertainty range (the default value for applied N lost directly as N_2O is 1% with an uncertainty range of 0.3–3%). This reflects uncertainty in the relationship between applied N and N_2O emission from soil, which is mediated by microbial processes that are influenced by soil conditions, especially temperature, moisture, organic carbon and available N. These soil factors are very variable, and their effects interact strongly (Brown et al., 2000) causing emissions to be spatially and temporally episodic (Dobbie and Smith 2001). Other work in the MIN-NO project is exploring improved N_2O emission factors for use in GHG accounting for crop products.

The GHG emissions for milling wheat grain represent a large component of the GHG emissions of a loaf of bread at the retail stage. Other work has shown that a standard 800 g white loaf (typical for UK consumption) has GHG emissions of 0.6 kg CO_2e, of which 0.2 kg CO_2e was for processing, packaging (Wiltshire *et al*., 2009). In that study, also using PAS 2050 as the LCA method, an emissions total for wheat of 640 kg CO_2e t^{-1} was assumed. This indicates that, correcting for a lower emissions total for milling wheat as assessed in this work (427 kg CO_2e t^{-1}) more than half of the GHG emissions for a loaf of white bread are from on-farm wheat production. Therefore, the emissions during crop production, and the variability in these emissions, strongly influence the emissions of a retailed loaf of bread.

5. Conclusion

Emissions of N_2O during wheat crop production are important, in both scale and variability, for the GHG emissions associated with retailed bread. There is an urgent need for (a) better precision and certainty in estimation of N_2O emissions, and (b) mitigation of GHG emissions on farms.

6. Acknowledgements

We acknowledge the UK Department for Environment, Food and Rural Affairs and the Scottish Government in sponsoring Sustainable Arable LINK Project LK09128, and the contributions of ADAS, Agricultural Industries Confederation, Bayer CropScience, British Sugar, Country Land and Business Association, The Co-operative, Frontier, GrowHow, HGCA – the cereals and oilseeds division of the Agriculture and Horticulture Development Board (AHDB), Hill Court Farm Research, National Farmers Union (NFU), North Energy Associates, Rothamsted Research-North Wyke, The Processors and Growers Research Organisation (PGRO), Renewable Energy Association, Scottish Agricultural College (SAC), Scotch Whisky Research Institute, Soil Essentials, Vivergo fuels, Warburtons and Yara.

7. References

Brown, L., Armstrong Brown, S., Jarvis, S.C., Syed, B., Goulding, K.W.T., Phillips, V.R., Sneath, R.W., Pain, B. 2000. An inventory of nitrous oxide emissions from agriculture in the UK using the IPCC methodology: emission estimate, uncertainty and sensitivity analysis. Atmospheric Environment 35, 1439-1449.

BSI, 2011. PAS 2050:2011, Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. Published: BSI, UK.

De Klein, C., Novoa, R.S.A., Ogle, S., Smith, K.A., Rochette, P., Wirth T.C., McConkey, B.G., Mosier, A., Rypdal, K. 2007. N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. Chapter 11 in 2006 IPCC Guidelines for National Greenhouse Gas Inventories Volume 4: Agriculture, Forestry and Other Land Use. 54 pp.

Dobbie, K.E., Smith, K.A. 2001. The effects of temperature, water-filled pore space and land use on N_2O emissions from an imperfectly drained gleysol. European Journal of Soil Science 52, 667-673.

Hoben J.P., Gehl R.J., Millar N., Grace P. R., Robertson G. P., 2011. Nonlinear nitrous oxide (N_2O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. Global Change Biology 17, 1140-1152.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan.

MacCarthy J., Brown K., Webb N., Passant N., Thistlethwaite G., Murrells T., Watterson J., Cardenas L., Thomson A., and Pang Y., 2011. UK Greenhouse Gas Inventory, 1990 to 2009: Annual Report for Submission under the Framework Convention on Climate Change. Didcot UK, AEA Technology plc.

Wiltshire J.J.J., Fendler A., Tucker G.S., Wynn S., 2008. The contribution of primary food production to lifecycle greenhouse gas emissions of food products. Aspects of Applied Biology 87, 73-76.

Wiltshire J., Wynn S., Clarke J., Chambers B., Cottrill B., Drakes D., Gittins J., Nicholson C., Phillips K., Thorman R., Tiffin D., Walker O., Tucker G., Thorn R., Green A., Fendler A., Williams A., Bellamy P., Audsley E., Chatterton J., Chadwick D., Foster C., 2009. Scenario building to test and inform the development of a BSI method for assessing greenhouse gas emissions from food, technical annex to the final report. Report to Defra, UK. Project Reference Number: FO0404.

Introduction of a national method to estimate direct nitrous oxide emissions from mineral soils for Finnish product carbon footprinting

Hannele Pulkkinen[1,*], Kristiina Regina[2], Juha-Matti Katajajuuri[1]

[1] MTT Agrifood Research Finland, Biotechnology and Food Research, FI-00790 Helsinki, Finland
[2] MTT Agrifood Research Finland, Plant Production Research, FI-31600 Jokioinen, Finland
[*] Corresponding author. E-mail: *hannele.pulkkinen@mtt.fi*

ABSTRACT

Direct nitrous oxide emissions from managed soils play a very significant role in climate impacts of food products. However, an IPCC default emission factor is applied almost without exception to estimate emissions in climate impact studies of food products. A more detailed method was developed for estimating direct nitrous oxide emissions in Finland for increased accuracy. Research on nitrous oxide fluxes under Finnish conditions was conducted and a new method for estimating fluxes from mineral soils was developed for Finland. The method results in markedly larger emissions to annual spring crops and smaller emissions to perennial crops compared with the IPCC default. The new method also significantly increases emissions at the national level. Acknowledging the substantial variation in nitrous oxide emissions, more accurate estimation methods need to be developed to understand the impacts of food produced in different climatic and geographic circumstances.

Keywords: climate impact, food, nitrous oxide emissions, carbon footprint, potato

1. Introduction

Direct nitrous oxide emissions from managed soils play a very significant role in climate impacts of food products. According to different studies they often contribute 20-30% to total climate impact of different food products and even more to emissions from cultivation. However, IPCC default emission factors (IPCC 1996, 2006) are used to estimate direct nitrous oxide emissions from applied N from managed soils almost without exception in carbon footprint studies of food products (Pulkkinen 2010). One exception is an Australian study of bread (Biswas et al., 2008), which employed locally derived emission factors of Barton et al., (2008). Similar direct nitrous oxide emission factors are used for the Australian National Inventory Report (Australian National greenhouse accounts 2012).

In the Finnish Foodprint-programme, harmonised national guidelines were developed to assess climate impacts of food products. Calculation guidelines are based on other international life cycle assessment standards and guidelines, and best practices, and give more practical instruction to the food industry than previously published general standards. In addition to harmonising carbon footprint methodology, a more detailed national method for estimating direct nitrous oxide emissions from mineral soils in Finland was developed to gain a more accurate understanding of climate impacts of food products.

2. Methods

The uncertainties of nitrous oxide emissions are very high due to large spatial and temporal variation (Snyder et al., 2009). To reduce the uncertainty of national estimates, Regina et al., (submitted) conducted research on nitrous oxide fluxes under Finnish conditions from 13 fields for periods of one to three years in 2000-2009. Their main finding was that the annual direct emissions of nitrous oxide were lower from grass crops than from annual spring crops. The long period between harvesting and sowing under boreal conditions, when there is no vegetation during the long winter, increases the emissions from annual spring sown crops. They were able to provide a method for estimating direct nitrous oxide fluxes from grass and annual spring crops from mineral soils in Finland that reflected national conditions better than the IPCC default method.

Statistical mixed models were based on the measured emissions of nitrous oxide and background variables (Regina et al., submitted). Environmental and management data available for the analysis included crop, fertiliser application rate, fertiliser type, soil characteristics and weather data. The crop type and the amount of mineral N applied best explained the variation in nitrous oxide emissions, and the model is consequently based on these two effects.

To estimate the burden of human activity (cultivation) only, a background emission was deducted from the derived emission estimates of both crops by deducting the emissions at zero fertiliser application rate. The emission estimate at fertiliser level zero of annual spring crops was 2.013 and of perennial crops 0.529. Because the number of measurements results at zero fertilisation is limited, a conservative estimate of the background emission was used and only the smaller value was subtracted (derived from measurements on perennial crops) from all calculated emission rates. This value, 0.529 kg N ha^{-1} yr^{-1}, is close to the annual emissions from the native vegetation, i.e. forest (IPCC 2006).

The derived equations for estimating nitrous oxide (N_2O) flux from mineral soils for perennial crops (Eq. 1) and for annual spring crops (Eq. 2) are therefore:

$$N_2O \text{ flux } (kgN_2O\text{-}N\ ha^{-1}\ yr^{-1}) = 10 \wedge (-0.2762 + 0.002848 * minN) - 0.529 \qquad \text{Eq. 1}$$

$$N_2O \text{ flux } (kgN_2O\text{-}N\ ha^{-1}\ yr^{-1}) = 10 \wedge (-0.2762 + 0.002848 * minN + 0.58) - 0.529 \qquad \text{Eq. 2}$$

These equations can be used to estimate the total emission from the field without dividing it between emissions from applied N, crop residues and N mineralisation. Indirect nitrous oxide emissions were not included in the field measurements and therefore need to be calculated separately.

To evaluate the new national method and to understand how it changes the results of Finnish climate impact studies, comparisons were made at two levels; a simple case study on a potato product and a national level estimate. Comparison was made with the IPCC 2006 method as it is proposed for use in Finnish carbon footprinting guidelines (developed in Foodprint-programme). To allow comparisons, all emissions were converted to CO_2 equivalents by using the 100-year global warming potentials of 25 and 298 for CH_4 and N_2O, respectively (IPCC, 2007).

The climate impact of packaged raw potatoes was first calculated according to IPCC 2006 method, including liming, and direct and indirect nitrous oxide emissions from mineral and organic soils. Then the calculation was repeated with the exception of applying the new national emission factor for nitrous oxide emissions from applied N and crop residues from mineral soils. The cradle-to-logistic terminal study included production of agricultural inputs, potato cultivation, processing, packaging, transport and storage. The activity data were collected from one large Finnish potato marketer. The data from the cultivation stage were collected from 2008-2010 and the sample covered nearly 60% of the producers. It should be noted that the nitrogen fertiliser levels in the study were a little lower than average Finnish potato fertiliser levels. This was likely due to the high level of specialisation of the farmers in potato cultivation. Data from processing (washing, separation, packing) and transport to logistic terminal from all six producers were collected from 2010. Neither carbon sequestration nor land use changes were taken into account due to lack of agreed methodology and data.

The comparisons of national level direct nitrous oxide emissions were made between the new national method and IPCC 2006 method. National statistics for cultivated area of different crops were used. Average minimum and maximum total nitrogen fertiliser levels were defined to allow understanding of the magnitude at low and high fertiliser levels. The levels were based on different farm statistics and the limits of the agri-environmental scheme. For cultivation area and fertiliser levels used in national level calculations, specific data on the 25 most common crops and aggregated data of other crops were used.

It has to be noted that in the national level estimates the crop residues were only included regarding emissions from mineral soils in national method calculations, as that was done automatically using new emission factors, but not for the IPCC method calculations. However, the calculations should be comparable as emissions from crop residues only account for around 5% of direct and indirect nitrous oxide emissions from agricultural soils in Finland (Statistics Finland, 2012).

3. Results

To demonstrate the impact of the new national method for climate impact studies at product level, a case study on a potato product was performed. The direct nitrous oxide emissions almost doubled with the new national method compared with that of IPCC 2006. This means that the greenhouse gas emissions from cultivation rose 46% and the total climate impact of packaged potatoes rose 25% (Table 1). Compared to the IPCC default methods, the new national method indicated almost twice as large N_2O emissions for many common annual crops with relatively high fertiliser levels (such as grain crops). The case study shows that application of the new national method has significant impacts at the product level, especially on low carbon footprint products.

Table 1. Shares of emission sources in potato case when using IPCC 2006 default emission factor and the new national method.

	IPCC 2006	New national method
N_2O emissions from managed soils	16%	32%
Other GHG-emissions from cultivation	38%	36%
Processing, packaging, transport	46%	36%
Total $kgCO_2$-eq./kg potato product	0.11	0.13

Comparisons between the new national method and the IPCC 2006 method were also made at national level. Applying the new national method, direct nitrous oxide emissions from applied nitrogen from managed mineral soils increased approximately by 100% compared with the IPCC 2006 method, depending on the fertiliser level assumed. The emissions from perennial crops decrease 30-46%, as emissions from annual crops increase by 232-250% (Table 2). The comparison shows that application of the new national method will increase the estimated national direct nitrous oxide emissions significantly.

Table 2. Estimated change in national nitrous oxide emission levels from applied N in mineral soils when the new national method is used instead of the IPCC 2006 method, with estimated minimum and maximum Finnish fertiliser levels.

	Minimum fertiliser levels	Maximum fertiliser levels
Annual crops	+250%	+232%
Perennial crops	-46%	-30%
Others	+176%	+164%
Total	+116%	+99%

4. Discussion

It is known that many factors in addition to fertiliser application rate have an effect on nitrous oxide fluxes. Several of these factors were also studied in long-term field measurements in Finland, such as type of fertiliser (mineral/organic), percentage of organic carbon, sand and clay in the 0-20 cm soil layer, mean temperature for the winter months (Jan-Mar) and total precipitation for the summer months (May-Sep). However, inclusion of the other parameters did not improve the performance of the models. This is a clear drawback as the models cannot take into account diverse cultivation methods or mitigation options, such as organic agriculture, autumn sown crops, reduced tillage etc.

The results indicate that the most important factor determining the annual flux of nitrous oxide is the type of crop, based on the division between annual and perennial crops. It seems justified to add this dimension also to the climate impact calculations for different agricultural products. The division is particularly important for Finnish conditions where spring-sown crops prevail and where the period when the soils are not covered by crops can be close to nine months. This increases the incidence of nitrous oxide emissions from these soils.

5. Conclusion

The new national method provides realistic estimates of nitrous oxide fluxes under boreal conditions, characterised by frozen soils in the winter, frequently renewed grasslands, and spring-sown annual crops. The more accurate method highlights that under boreal conditions, such as Finland, direct nitrous oxide emissions from annual crops on mineral soils are markedly higher than suggested by the IPCC method. It also demonstrates that total Finnish direct nitrous oxide emissions from all managed soils are likewise higher.

Acknowledging the substantial variation in nitrous oxide emissions, developing new methods to estimate fluxes in more detail should be given much more attention. Better knowledge of food production in different climates is needed. As interest in climate impacts of food grows, the need for more detailed assessment methods is urgent.

6. References

Australian national greenhouse accounts, 2010. National Inventory Report 2010. Vol. 1. The Australian Government Submission to the United Nations Framework Convention on Climate Change April 2012

Barton, L., Kiese, R., Gatter, D., Butterbach-Bahl, K., Buck, R., Hinz, C., Murphy, D.V., 2008. Nitrous Oxide Emissions from a Cropped Soil in a Semi-Arid Climate. Glob. Change Biol., 14, 177–192.

Biswas, W. K., Barton, L., Carter, D., 2008. Global warming potential of wheat production in Western Australia: a LCA. Water and Environment Journal, 22, 3, 206-216.

Statistics Finland, 2011. Greenhouse gas emissions in Finland 1990-2009. National Inventory Report under the UNFCCC and the Kyoto Protocol. Statistics Finland.

IPCC, 1996. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories: Workbook. Volume 2. Intergovernmental Panel on Climate Change.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change. National Greenhouse Gas Inventories Programme. Online at http://www.ipccnggip.iges.or.jp/public/2006gl/index.html.

IPCC, 2007. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, UK and New York, NY, USA, 2007, 996 pp.

Pulkkinen, H., Katajajuuri, J.-M., Nousiainen, J., Silvenius F., 2010. Challenges in the comparability of carbon footprint studies of food products. In: Proceedings of LCA food 2010. Vol. 2, 65-70.

Regina, K., Kaseva, J., Esala, M., 2012. Emissions of nitrous oxide from boreal agricultural mineral soils – statistical models based on measurements. Submitted to Agriculture, Ecosystems & Environment.

Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertiliser management effects. Agriculture, Ecosyst. Env. 133, 247–266.

Statistics Finland, 2012. Greenhouse gas emissions in Finland 1990-2010. National Inventory Report under the UNFCCC and the Kyoto Protocol. April 2012.

Modelling N_2O emissions from organic fertilisers for LCA inventories

Matthias S. Meier*, Christian Schader, Alfred Berner, Andreas Gattinger

FiBL - Research Institute of Organic Agriculture, Ackerstrasse, CH-5070 Frick, Switzerland
* Corresponding author e-mail: *matthias.meier@fibl.org*

ABSTRACT

Apart from mineral fertilisers, organic fertilisers act mainly via the soil C-N pool in order to provide available nitrogen to plants. This different mode of action is not reflected so far in the current IPCC N_2O emission model. Here we propose a simple model to calculate N_2O emissions from organic fertilisers and plant residues. It considers the long-term immobilisation of N within stable organic matter in the soil as well as the mineralisation of additional N from the soil pool. A first test with field data showed reliable simulations of measured N_2O emissions. By comparing values generated by our model with values generated by the IPCC model we show that the IPCC model may overestimate emissions from organic fertilisers. Therefore, within LCA inventories modelling of soil borne N_2O emissions from organic fertilisers and crop residues should consider the different dynamics of N via the C-N pool in the soil.

Keywords: soil borne nitrous oxide emissions, agricultural inventory data, organic fertilisers, crop residues

1. Introduction

The IPCC model for determining nitrous oxide (N_2O) emissions from soils (IPCC, 2006) – originally developed for the reporting of national GHG inventories – is widely used within life cycle assessment (LCA) inventories to calculate soil N_2O emissions from agricultural products and processes. This emission factor based model considers the total N input by fertilisation and plant residues to estimate cumulative direct and indirect N_2O emissions from soils. Regarding direct N_2O emissions, the model does not differentiate between different fertiliser types, i.e., nitrogen from mineral vs. organic sources including plant residues.

However, there is growing evidence that N_2O emissions from organic fertilisers may be different from emissions from mineral fertilisers. First, from nitrogen-use efficiency studies it is known that mineral and organic fertilisers differ in their mode of action through the way nutrients are transformed in the soil and utilised by plants (Gutser et al., 2005). In organic fertilisers only a fraction of the total N is readily available for plants (as NH_4^+-N and NO_3^--N). This fraction ranges from up to 85% of total N in poultry slurry down to 0% in compost. The rest of the total N in organic fertilisers is organically bound entering the C-N-pool of the soil where it is released in the mid- and long-term by microbial degradation. By calculating N_2O emissions based on total N in the IPCC model the different mode of action of organic fertilisers is ignored.

Second, ^{15}N tracer studies with monitoring periods of up to 9 years indicate that N losses from mineral fertiliser are higher than from (organic) crop residue input (Delgado et al., 2010). Again, the higher immobilisation of N from crop residues within the C-N-pool is made responsible for the lower N losses during the monitoring period. Using model simulations Delgado et al., (2008) showed that in support with ^{15}N tracer studies N losses through leaching and N_2O emissions from crop residue sources are lower compared to mineral fertiliser. Based on these simulation results they argue that the IPCC N_2O emission model (IPCC, 2006) overestimates N_2O emissions from crop residues.

Third, Alluvione et al., (2010) measured significantly lower N_2O emissions from compost compared to urea in corn fields. In this study, N_2O emissions were only measured during the vegetation period of the crop and therefore, no conclusions can be drawn regarding the long-term emissions resulting from the different fertilisers. Nevertheless, the study by Alluvione et al., (2010) showed that there are measurable differences in N_2O emissions between mineral and organic fertilisers as well as within different organic fertilisers.

Due to the different dynamics of N from organic input sources, different N_2O emissions from organic fertilisers and crop residues compared to mineral fertiliser can be expected. Therefore, N_2O emissions from organic fertilisers and plant residues should be modelled differently from mineral fertiliser. This is of special relevance when the GWP of crops fertilised mainly or exclusively with organic fertilisers (e.g. from organic farming) are to be compared with the GWP of crops fertilised with mineral fertilisers.

We developed a simple model taking into account the different mode of action of organic fertilisers. The accuracy of the model was tested using measured N_2O emissions from a 3-year study. A comparison of the model calculations from the IPCC N_2O emission model (IPCC, 2006) was made using data from a long-term field trial at the Research Institute of Organic Agriculture (FiBL), Frick, Switzerland.

2. Methods

2.1. Model description

Our model considers the C and N input through organic fertilisers and plant residues brought onto and into the soil (Fig. 1). A fraction of the total N input (N_{tot}) is readily available for plants ($N_{available}$) and the remaining fraction of N_{tot} is organically bound entering the C-N pool of the soil ($N_{C\text{-}N\text{-}pool}$). In contrast to the IPCC model (IPCC, 2006) where direct and indirect emissions are calculated from total N input our model differentiates between direct and indirect N_2O emissions from $N_{available}$ and from the plant available N arising from the C-N pool of the soil ($N_{available\ C\text{-}N\ pool}$). Total N_2O emissions from managed soils are than calculated according to Eq. 1:

$$N_2O_{total\ emission} = N_2O_{direct}\text{-}N_{available} + N_2O_{indirect}\text{-}N_{available} + N_2O_{direct}\text{-}N_{available\ C\text{-}N\ pool} + N_2O_{indirect}\text{-}N_{available\ C\text{-}N\ pool} \quad \text{Eq. 1}$$

Direct and indirect N_2O emissions from $N_{available}$ (Loss 1 in Fig. 1) are calculated using the IPCC emission factors (IPCC, 2006) according to Eq. 2 and 3:

$$N_2O_{direct}\text{-}N_{available} = 0.01 \times N_{available} \quad \text{Eq. 2}$$

$$N_2O_{indirect}\text{-}N_{available} = 0.01 \times (NH_3\text{-}N_{available} + NO_x\text{-}N_{available}) + 0.0075 \times NO_3^-\text{-}N_{available} \quad \text{Eq. 3}$$

Direct and indirect N_2O emissions from $N_{available\ C\text{-}N\ pool}$ (Loss 2 in Fig. 1) are calculated using the IPCC emission factors (IPCC, 2006) according to Eq. 4 and 5:

$$N_2O_{direct}\text{-}N_{available\ C\text{-}N\ pool} = 0.01 \times N_{available\ C\text{-}N\ pool} \quad \text{Eq. 4}$$

$$N_2O_{indirect}\text{-}N_{available\ C\text{-}N\ pool} = 0.0075 \times NO_3^-\text{-}N_{available\ C\text{-}N\ pool} + 0.01 \times NO_x\text{-}N_{available\ C\text{-}N\ pool} \quad \text{Eq. 5}$$

For the fate of $N_{C\text{-}N\ pool}$, which equals ($N_{tot} - N_{available}$), two pools are differentiated in the model (Fig. 1) based on a simplified model on sequestration of soil organic carbon (SOC) proposed by Favoino and Hogg (2008). These two pools differ in their stability of the organic matter with fractions of short-term available C and N (short-term C-N pool) and stable fractions (long-term C-N pool) where C and N is captured for several 100 of up to a 1'000 years ($N_{immobilised}$) (Favoino and Hogg, 2008). From both pools mineralisation takes place.

In soils with a build-up of SOC, e.g. through high organic matter input by organic fertilisers and/or conservation tillage, mineralisation from the long-term C-N pool takes place at a much lower rate than at which readily available organic matter is converted to stable organic matter (Favoino and Hogg, 2008). In this case the result is a net capture of N in stable organic matter ($N_{immobilised}$) and therefore, no N_2O emissions will result from this N in the long term. $N_{available\ C\text{-}N\ pool}$ is then calculated by Eq. 6:

$$N_{available\ C\text{-}N\ pool} = N_{C\text{-}N\ pool} - N_{immobilised} \quad \text{Eq. 6}$$

Once the long-term C-N pool is saturated no additional N is captured. Therefore, all N being processed via the soil C-N pool ($N_{C\text{-}N\ pool}$) will be available short-term.

In soils with degradation of SOC due to low organic matter input and/or non-conserving tillage techniques mineralisation outweighs immobilisation. Additional N from the C-N pool is mineralised ($N_{mineralised}$). In this case $N_{available\ C\text{-}N\ pool}$ is calculated by Eq. 7:

$$N_{available\ C\text{-}N\ pool} = N_{C\text{-}N\ pool} + N_{mineralised} \quad \text{Eq. 7}$$

Due to the coupled biogeochemical cycles of C and N the amounts of $N_{immobilised}$ and $N_{mineralised}$ can be determined from the C fluxes consisting of an input flux (C_{input}) into the soil, a build-up or degradation of soil organic carbon (ΔSOC) within the soil and an output flux (C_{output}) from the soil. If two of these 3 fluxes are known the third is determined by Eq 7:

$$C_{input} - C_{output} = \Delta SOC \quad \text{Eq. 8}$$

C_{input} is calculated by adding the amounts of C in the organic fertilisers and the C in the crop residues. Above and below ground crop residues can be determined by using the default factors for crop residue estimation within Chapter 11 of Volume 4 of the IPCC guidelines (IPCC, 2006). If measurements of either SOC or CO_2 emissions from the soil are available these can be used within Eq. 8. However, measurements of CO_2 emissions include CO_2 form soil microbial activity as well as from root respiration. Therefore, using CO_2 emission measurements within Eq. 8 will result in an underestimation of a gain in SOC if $C_{input} > C_{output}$ and an overestimation of a loss in SOC if $C_{input} < C_{output}$. If no measurements are available the IPCC guidelines provide in Chapter 2 of Volume 4 a simple model to estimate annual changes in SOC based on soil type, climatic conditions and cultivation practices (IPCC, 2006).

The amount of $N_{immobilised}$ can be determined by estimating the amount of stable C within the SOC for example by using the simple model described in Favoino and Hogg (2008). Using this model, a percentage typical of Northern European areas of 35% of SOC transformed into stable organic matter was used in our model calculations. Besides climatic conditions this percentage further depends on the soil texture and on how much carbon is already stored in the soil. For specific cases site-specific values have to be determined. The amount of $N_{immobilised}$ in SOC is then determined by Eq. 9:

$$N_{immobilised} = 0.35 \times (C_{input} - C_{output}) / C{:}N_{soil} \qquad \text{where } (C_{input} > C_{output}) \qquad \text{Eq. 9}$$

$N_{mineralised}$ is determined by Eq. 10:

$$N_{mineralised} = (C_{output} - C_{input}) / C{:}N_{soil} \qquad \text{where } (C_{output} > C_{input}) \qquad \text{Eq. 10}$$

The use of the model is not restricted to organic fertilisers. Of course it can also be used for mineral fertiliser. In that case $N_{available}$ equals the N from the mineral fertiliser and $N_{C\text{-}N\ pool}$ originates from plant residues only.

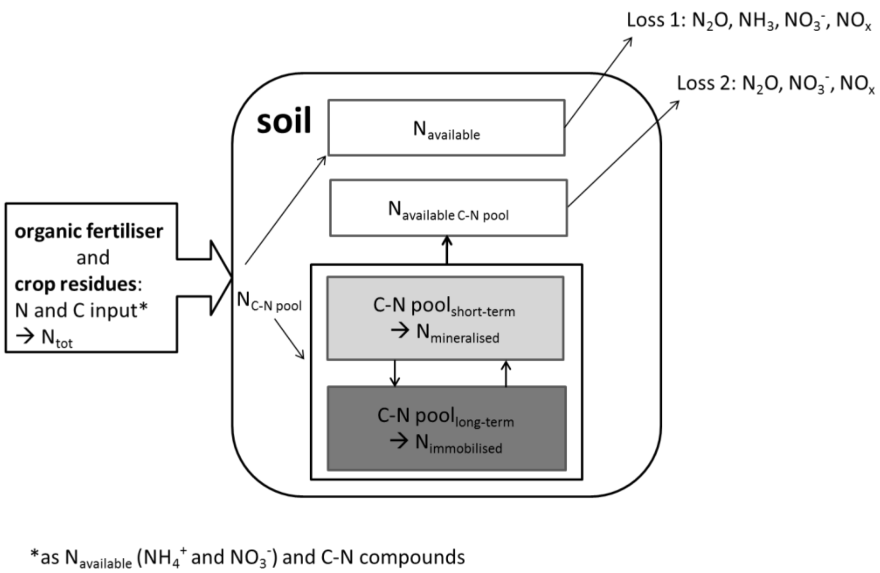


Figure 1. Soil N_2O emission model for organic fertilisers.

2.2. Model testing

The data set from Ellert and Janzen (2008) monitoring year-round N_2O, CO_2 and CH_4 emissions in different crop rotations for 3 years in Alberta, Canada was used to test the accuracy of our newly developed model. Since indirect N_2O emissions are not captured by field measurements, only direct emissions were considered in the model calculations for this test.

One crop rotation in Ellert and Janzen (2008) was legume based with alfalfa-alfalfa-alfalfa-wheat-barley and a second crop rotation was a N-demanding sequence with corn-wheat-corn-wheat-barley. Emission measurements started in 2001 when the rotations were in the 3rd phase and lasted until 2003. In each of the two crop rotations 4 different fertiliser treatments were applied: zero fertiliser, solid beef cattle manure from feedlots, ammonium nitrate, and ammonium nitrate combined with solid manure. Since the publication by Ellert and Janzen (2008) provides no data on the achieved crop yields during the years under study and these are needed in the model to calculate the C input flux and the amounts of crop residues, average yield data for Alberta, Canada was taken from official statistics of the respective years (Alberta Agriculture and Food, 2007). Data on N content, yields and amount of residues from alfalfa after consecutive years of cultivation were taken from Jung (2003). All other crop residues were determined using the default factors for estimation of N added to soils from crop residues within the IPCC guidelines (IPCC, 2006). The C:N ratio of the Chernozem soil on which the study was carried out was taken from FAO (2001). Additional information on nutrient composition of solid beef cattle manure from feedlots not provided in Ellert and Janzen (2008) were taken from Kissinger et al., (2007).

2.3. Model comparison

We further calculated N_2O emissions with our model using data from an organic long-term field trial at the Research Institute of Organic Agriculture (FiBL), Switzerland, which compares effects of reduced versus conventional tillage on soil quality (Berner et al., 2008). From this field trial measured data on soil carbon stocks, amounts and composition of fertilisers used and crop yields are available. Beef cattle slurry was used as organic fertiliser. The N_2O emissions determined by our model were compared with the emissions' calculations from the IPCC model. Regarding indirect N_2O emissions only emissions through volatilisation of N as NH_3 were considered in the comparative calculations using an emission factor specific to Switzerland.

3. Results

The correlation of simulated direct N_2O emissions with the measured values from the study of Ellert and Janzen (2008) is strong with a correlation coefficient of around 0.84 (Fig. 2). The 4 different fertiliser treatments were modelled with high reliability.

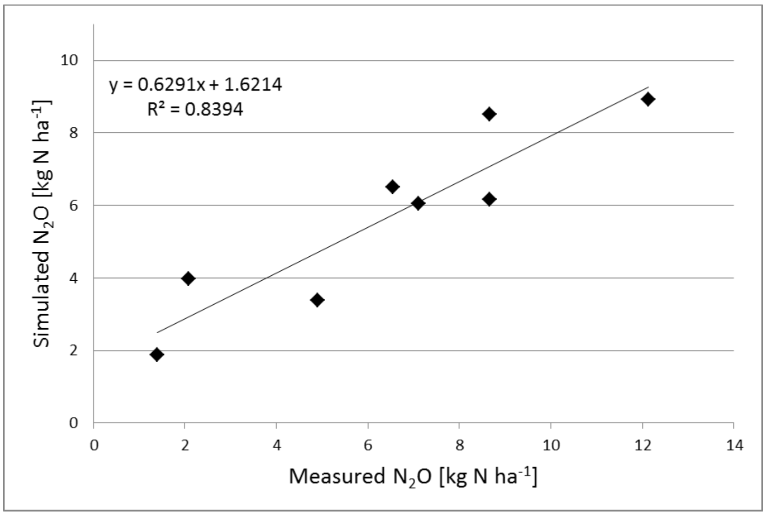


Figure 2: Simulated versus measured N_2O emissions (Ellert and Janzen, 2008) from two different crop rotations and 4 different fertiliser treatments each used to test the model.

The comparison of calculations from our model with calculations from the IPCC model (IPCC, 2006) using data from an organic field trial yielded 9% lower emissions on average over the entire crop rotation for the conventional tilled plots (Table 1). For the reduced tilled plots our model calculations resulted in 23% lower emissions on average over the entire crop rotation than the emissions calculated with the IPCC model.

Table 1. Comparison of model calculations.

	Conventional tillage			Reduced tillage		
Crop rotation	**N_2O emissons based on new model [kg/ha]**	**N_2O emissions based on 2006 IPCC model [kg/ha]**	**Difference on the basis of IPCC model in [%]**	**N_2O emissons based on new model [kg/ha]**	**N_2O emissions based on 2006 IPCC model [kg/ha]**	**Difference on the basis of IPCC model in [%]**
Winter wheat	3.52	3.80	-7	2.96	3.69	-20
Sunflower	2.26	2.44	-8	1.96	2.45	-20
Spealt	2.88	3.11	-7	2.89	3.43	-16
Clover grass	3.16	3.48	-9	2.62	3.60	-27
Clover grass	5.66	5.93	-5	5.50	6.32	-13
Silage corn	1.21	1.42	-15	1.88	2.86	-34
Winter wheat	1.93	2.14	-10	1.65	2.28	-28
Average			-9			-23

4. Discussion

Our model presented in Fig. 1 reflects the nature of organic fertilisers considering its predominant mode of action via the soil C-N pool. By integrating the coupled biogeochemical cycles of C and N, nitrogen immobilisation and mineralisation are included in the N_2O emission calculations. Different organic fertilisers are differentiated in the model by the fraction of available N, organically bound N, and the C content. Other characteristics of organic fertilisers such as the liquid phase in the case of slurry are not considered in the model. Even though due to its liquid phase slurry might enhance denitrification and by that possibly leads to higher N_2O emissions. However, compared to solid organic fertilisers this characteristic of slurry might be of significant difference in the first period after application only. Our model calculates long term N_2O emissions from annual N inputs where such short-term effects are interfered with climatic conditions such as temperature and precipitation. Site specific climatic conditions can be considered in the model by using site specific emission factors.

A first test with measured N_2O emissions shows a high correlation to the N_2O emissions calculated by our model (Fig. 2). Different fertiliser treatments including mineral fertiliser and plant residues only were reliably calculated. The reason for the fact that the model didn't reproduce the exact measured values is because we used the default IPCC emission factors (IPCC, 2006). Using site-specific emission factors the model will produce the measured values with high accuracy.

In cases where there is a build-up of SOC like under reduced or no-tillage cultivation practices combined with high organic matter input through organic fertilisers and crop residues our model calculates lower N_2O emission due to a higher long-term immobilisation of N (Table 1). There are studies, though, that show higher denitrification activities under reduced or no tillage cultivation practices than under conventional tillage resulting in higher N_2O emissions (Palma et al., 1997; Steinbach and Alvarez, 2006). However, the meaning of such studies for long-term emission patterns under reduced or no-tillage management are still unsure and discussed controversially (Mummey et al., 1998; Six et al., 2004; Steinbach and Alvarez, 2006). In a meta-analysis Steinbach and Alvarez (2006) found in Pampean agroecosystems increasing N_2O emissions under no-till. The authors conclude that the mitigation potential of no-till due to C sequestration might be overcome in about 35 years. In contrast, Six et al., (2004) showed in a literature review analysing studies that compared no- and conventional tillage, N_2O emissions seem to increase in newly converted no-tillage systems but over a period of 10 years in humid climates and a period of 20 years in dry climates N_2O emissions decrease and overall net GWP is reduced under no-tillage cultivation practices. Further, in cases where there is a build-up of SOC our model would also deliver lower N_2O emissions for mineral fertiliser applied under reduced or no-tillage cultivation practices due to immobilisation of N from crop residue decomposition.

5. Conclusion

Our model-simulations support earlier findings that the IPCC model overestimates N_2O emissions from organic fertilisers in certain cases. However, our model has to be further validated with field measurements of N_2O especially under cultivation practices that result in a build-up of SOC.

By including N mineralisation and immobilisation in N_2O emissions' calculations our model reflects field situations with higher accuracy than the IPCC 2006 model. It accounts for the different dynamics of N from organic fertilisers and crop residues. By including the C cycle, which is influenced by climatic conditions, soil conditions, and management practices, factors beyond the N input influencing N_2O emissions are also considered. The integration of the coupled biogeochemical cycling of C and N in our model further allows for inclusion of C sequestration.

Based on our results, when modelling soil borne N_2O emissions within LCA inventories from organic fertilisers and crop residues we propose to take into account the N pathway via the C-N pool of the soil using our model.

6. References

Alberta Agriculture and Food, 2007. Alberta agriculture statistics yearbook 2006. Alberta Agriculture and Food, Economics and Competitiveness Division, Statistics and Data Development Unit, Edmonton, Canada.

Alluvione, F., Bertora, C., Zavattaro, L., Grignani, C., 2010. Nitrous Oxide and Carbon Dioxide Emissions Following Green Manure and Compost Fertilization in Corn. Soil. Sci. Soc. Am. J. 74, 384-395.

Berner, A., Hildermann, I., Fließbach, A., Pfiffner, L., Niggli, U., Mäder, P., 2008. Crop yield and soil fertility response to reduced tillage under organic management. Soil Till. Res. 101, 89-96.

Delgado, J.A., Del Grosso, S.J., Ogle, S.M., 2010. (15)N isotopic crop residue cycling studies and modeling suggest that IPCC methodologies to assess residue contributions to N(2)O-N emissions should be reevaluated. Nutr. Cycl. Agroecosys. 86, 383-390.

Ellert, B.H., Janzen, H.H., 2008. Nitrous oxide, carbon dioxide and methane emissions from irrigated cropping systems as influenced by legumes, manure and fertilizer. Can. J. Soil Sci. 88, 207-217.

FAO, 2001. Lecture notes on the major soils of the world. World soil resources reports, Vol. 94; FAO, Rome.

Favoino, E., Hogg, D., 2008. the potential role of compost in reducing greenhouse gases. Waste Manage. Res. 26, 61-69.

Gutser, R., Ebertseder, T., Weber, A., Schraml, M., Schmidhalter, U., 2005. Short-term and residual abailabiltiy of nitrogen after long-term application of organic fertilizers on arable land. J. Plant Nutr. Soil Sci. 168, 439-446.

IPCC, 2006. 2006 IPCC Guidelines for national greenhouse gas inventories. Volume 4 Agriculture, forestry and other land use. Hayama, Japan.

Jung, R., 2003. Stickstoff-Fixierleistung von Luzerne (Medicago sativa L.), Rotklee (Trifolium pratense L.) und Persischme Klee (Trifolium resupinatum L.) in Reinsaat und Gemenge mit Poaceen. Experimentelle Grundlagen und Kalkulationsverfahren zur Ermittlung der Stickstoff-Flächenbilanz. Fakultät für Agrarwissenschaften. Georg-August-Universität Göttingen, Göttingen, p. 366.

Kissinger, W.F., Koelsch, R.K., Erickson, G.E., Klopfenstein, T.J., 2007. Characteristics of manure harvested from beef cattle feedlots. Appl. Eng. Agric. 23, 357-365.

Mummey, D.L., smith, J.L., Bluhm, G., 1998. Assessment of alternative soil management practices on N2O emissions from US agriculture. Agr. Ecosyst. Environ. 70, 79-87.

Palma, R.M., Rímolo, M., Saubidet, M.I., Conti, M.E., 1997. Influence of tillage system on denitrification in maize-cropped soils. Biol. Fert. Soils 25, 142-146.

Six, J., Ogle, S.M., Breidt, J.F., Conant, R.T., Mosier, A.R., Paustian, K., 2004. The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. Global Change Biol. 10, 155-160.

Steinbach, H.S., Alvarez, R., 2006. Changes in soil organic carbon contents and nitrous oxide emissions after introduction of no-till in Pampean agroecosystems. J. Environ. Qual. 35, 3-13.

Sustainable meat consumption to meet climate and health goals - implications of variations in consumption statistics

Elinor Hallström*, P. Börjesson

Department of Technology and Society, Environmental and Energy Systems Studies, Lund University, PO Box 118, SE-221 00 Lund, Sweden.
*Corresponding author. E-mail: *elinor.hallstrom@miljo.lth.se*

ABSTRACT
To develop recommendations for sustainable meat consumption, e.g. from climate and health perspectives, there is a need for a correct understanding of how much meat is produced and actually consumed. The purpose of this paper is to contribute to the understanding of critical issues regarding meat consumption statistics, its implication on LCA and recommendations of sustainable meat consumption levels. Depending on the way of presenting data, meat consumption levels per capita may differ by a factor two. This illustrates the importance of specifying the functional unit and clearly define if it refers to meat including or excluding bones, including losses along the food chain, or after weight reduction by cooking, for a correct utilisation and interpretation of meat consumption data and LCA's of meat. The need for reductions in current meat consumption to meet climate and health goals is estimated to 0-75% and 0-50%, respectively, depending on the region.

Keywords: meat consumption, statistics, LCA, climate, health

1. Introduction

Food production has been identified as one of the most important drivers of environmental pressures (EIPRO, 2006). To meet climate goals substantial mitigation efforts in the agriculture sector, estimated to account for about one third of anthropogenic greenhouse gas (GHG) emissions (Garnett, 2011), will be needed. Results from life cycle assessments (LCA) normally show that animal based foods are more climate intensive compared to plant based foods (FAO, 2006; 2009; Garnett, 2009). Due to population growth and a transition towards increased consumption of animal based products, global demand for livestock products are projected to double by 2050 compared to the year 2000 (FAO, 2006). Reduced meat consumption has been suggested to be a necessary measure for mitigating food related GHG emissions (Garnett, 2011) and to have positive effects on public health in regions with affluent diet (McMichael et al., 2007). However, how much meat consumption needs to be reduced to reach a sustainable level, e.g. including both environmental and health aspects, is still uncertain.

To develop recommendations for sustainable meat consumption, e.g. based on LCA studies, there is a need for reliable data and a correct understanding of how much meat is available for consumption and how much meat is actually consumed. Because different methods are used to produce data on meat consumption and because there is more than one definition of meat consumption (Hallström and Börjesson, 2012), divergent information on how much meat is consumed is circulating in the literature and in media. For a correct utilisation and interpretation of meat consumption statistics, e.g. in LCA, it is essential to be aware of how statistical data on meat consumption is developed and presented.

The purpose of this paper is to contribute to the understanding of critical issues regarding meat consumption statistics, its implication on LCA and recommendations of sustainable meat consumption levels. The paper describes factors contributing to discrepancies in meat consumption data and illustrates the importance of correct definition of the functional unit, while exploring estimates of sustainable meat consumption levels regarding climate change and dietary recommendations.

2. Methods

Information about factors contributing to discrepancies in meat consumption statistics and their respective impact on data are based on the findings in an assessment by Hallström & Börjesson (2012). In order to illustrate how identified factors may contribute to discrepancies in data and to make relevant comparisons with goals for increased sustainability in meat consumption, meat consumption statistics provided by the FAOSTAT (data from 2007) is processed and categorised by using conversion factors stated in the literature.

3. Results

3.1 Factors contributing to discrepancy in meat consumption data

Food consumption can be measured top-down, e.g. based on data of the agricultural supply or bottom-up e.g. based on data from Household Budget Surveys or Individual Dietary Surveys (Naska et al., 2009; Westhoek et al., 2011). Factors identified to contribute to discrepancies between meat consumption data are whether the weight of bones is included, whether food waste in stages along the food chain is accounted for, whether the weight refers to raw or cooked meat and whether ingredients of non-meat origin in mixed processed meat products and ready meals are accounted for (Hallström and Börjesson, 2012). Depending on the method used to produce meat consumption statistics the data may refer to the available supply, the purchased amount or the amount of meat actually consumed. Meat consumption data can further be expressed either in carcass weight, as a sum of pure meat and products of higher degree of processing (e.g. mixed charcuteries and prepared meals) or in cooked amounts. Methods to adjust for bones and food losses at different stages of the food chain also vary among different ways of presenting meat consumption statistics.

The quantitative example in Table 1 illustrates how meat consumption statistics may vary depending on the way of presenting data, i.e. if the data refer to the available supply of carcass including bones (A), bone-free meat (B), bone-free meat after adjustment for losses at retail and consumer level (C) or bone-free meat after adjustment for losses and weight reductions during cooking (D). Data on the available supply of carcass including bones (A) are based on meat consumption statistics from FAOSTAT (data from 2007). Bone-free meat (B) is assumed to correspond to 70% of the carcass weight. According to previous research bone-free meat represents on average 70% (Cederberg, 2009), 59% and 77% (Sonesson, 2010) of the carcass weight in beef, pork and chicken, respectively. Based on an FAO report (FAO, 2011), a waste percentage of 15% has been assumed to adjust for losses at retail and consumer level (C) in North America, Oceania and Europe. Corresponding losses for South America and Asia are assumed to be, on average, 13%, and for Africa, 11% (where the losses in sub-Sahara Africa correspond to around 9%). Weight reduction by cooking (D) is assumed to correspond to 30% of the raw weight, a mean value of previous estimates varying between 20-50% depending on the type of meat, method and degree of cooking (World Cancer Research Fund/American Institute for Cancer Research, 2007; KF & ICA Provkök, 2000). The data presented in Table 1 refer to average meat consumption, i.e. no difference is made regarding variations in type of meat and differences in intake levels within each region. Thus, to make more reliable estimations on a regional level, specific statistics regarding the actual mix of meat consumed are needed. Also conversion factors used to quantify the data presented in column B-D are estimated averages. Thus, to improve the reliability in these data more extensive information of specific conditions is needed.

Table 1. Per capita meat consumption (kg/year) in different world regions

Region	A[a] Raw meat incl. bones	B Raw meat excl. bones	C B excl. losses in distribution and at consumer level	D C after weight reduction by cooking
North America	120	84	71	50
Oceania	115	81	68	48
Europe	77	54	46	32
South America	70	49	43	29
Asia	28	20	17	12
Africa	16	11	10	7

[a] Data in column A are based on FAO statistics which refer to the average quantity of meat including most bones at the slaughter exit. Available supply is quantified as the sum of nationally produced meat plus meat imports minus exports of meat, divided by the total population.

3.2 Reductions in meat consumption to reach a sustainable level – two illustrative examples

3.2.1 Climate perspective

Due to human activities global GHG emissions have increased by 70% during the past 40 years (IPCC, 2007). Scientific evidence indicates that a temperature rise greater than 1.5-2°C compared to pre-industrialised levels, may result in adverse effects including serious impact on the environment as well as future availability of food and water (IPCC, 2007; Stern, 2006). To increase the chances of limiting global warming to 1.5-2°C degrees, global GHG emissions will have to be halved by 2050 compared to levels in 1990, and in a long term perspective be limited from approximately 6-14 tonnes of CO_2e per capita per year (global average and average in developed countries, respectively) to levels of 1 to 2 tonnes of CO_2e per capita per year (European Commission, 2007; UNEP, 2010). Food production and consumption account for a significant proportion of global anthropogenic GHG emissions and overall environmental impact. Estimates from developed countries indicate that GHG emissions embodied in the diet are in the range of 2-3 tonnes of CO_2e per capita per year (Berners-Lee et al., 2012; Nilsson et al., 2011), equivalent to about 15-28% of the overall national emissions (Garnett, 2011). Meat has been identified as the food group responsible for the majority of GHG emissions attributable to the food sector (Carlsson-Kanyama and Gonzalez, 2009; Garnett, 2011).

From a climate perspective an average global per capita consumption of 25-33 kg of meat per year (68-90 grams per day) has been suggested as a goal to stabilise global livestock related GHG emissions until 2050 at 2000-2005 level (Garnett, 2008; McMichael et al., 2007). The suggested levels by Garnett (2008) and McMichael (2007) are quantified based on FAO data and thus refer to the available supply of raw meat including bones. From an LCA perspective, the corresponding functional unit (FU) could be translated as *kg produced raw meat including bones per capita per year*. The estimated need for reductions in current meat consumption to meet climate goals is exemplified in Table 2. In this example the level of sustainable meat consumption is assumed to be 29 kg per capita per year, which is an average of the suggested amounts.

3.2.2 Health perspective

From a nutritional point of view there are no general recommendations of how much meat is considered to be optimal for health. Existing dietary guidelines are instead usually based on levels that ensure sufficient intake of critical nutrients without exceeding upper intake limits of nutrients associated to negative health effects. In dietary guidelines, meat is usually categorised with other protein rich foods and suggested portion sizes and intake levels can vary (U.S. Department of Agriculture/U.S. Department of Health and Human Services, 2010; WHO, 2003). According to dietary guidelines in five different countries an intake between 50 and 100 g of cooked meat per day (18-37 kg per year) is suggested to provide a balanced nutrient intake (Hallström et al., 2011). To decrease the risk for cancer the World Cancer Research Fund (WCRF) further recommends that consumption of cooked red meat (e.g. beef, pork, lamb) should be restricted to maximum 500 g per week (26 kg per year, i.e. 70 g per day) and that processed meat, such as bacon, salami, sausages etc., should be avoided (World Cancer Research Fund/American Institute for Cancer Research, 2007). In addition dietary recommendations for healthy vegetarian diets, without meat, are available (U.S. Department of Agriculture/U.S. Department of Health and Human Services, 2010). The recommendations of sustainable meat consumption from a health perspective are accordingly based on the consumption of cooked meat. The corresponding FU in an LCA could then be translated as *kg consumed cooked meat per capita per year*. The estimated need for reduction in current meat consumption to meet health goals is exemplified in Table 2. In this example the level of healthy meat consumption is set to 26 kg per capita per year, which is in line with suggested amounts in dietary guidelines. The amount includes red and white meat but complies with the recommendation by the WCRF which refers to a maximum intake of red meat.

Table 2. Estimated need for reductions in meat consumption to meet climate and health goals[a]

Region	Reduction needed in% to meet climate goals	Reduction needed in% to meet health goals
North America	76	48
Oceania	75	46
Europe	62	19
South America	59	13
Asia	None	None
Africa	None	None

[a] To quantify the reductions in meat consumption needed to meet climate goals, data from column A in Table 1 has been compared with a consumption level of 29 kg per capita per year (FU: produced raw meat including bones). To quantify reductions needed to meet health goals data in Table 1, column D has been compared with a consumption level of 26 kg per capita per year (FU: consumed cooked meat).

4. Discussion

Statistics on meat supply, based on the production of raw meat including bones, are often the basis in describing the need for reduction in meat consumption to meet climate goals. In describing the need for reduction in meat consumption from a health perspective, on the other hand, data on the actual intake expressed as uncooked or cooked meat should preferably be used. From an LCA perspective, these two types of statistics represent two different functional units. There is an obvious risk of mixing those different functional units when broadening the perspective in LCA's, including, for example, a nutrition and health perspective.

Depending on the type of statistics and way of describing data, the meat consumption level per capita may differ by a factor two, or more. If meat consumption statistics are used wrongly it may result in an incorrect functional unit, which may influence the results and conclusions substantially. It is therefore crucial to specify the functional unit in, for example, LCA's of meat and dietary recommendations and clearly define if it refers to raw meat including or excluding bones, including losses in distribution and consumer level, or after weight reduction by cooking.

This paper indicates that the reductions needed in meat consumption to meet climate goals are considerable in all regions except Asia and Africa. In most regions there also seems to be a room for reduced consumption from a health perspective. The estimated needs for reduced meat consumption to reach a sustainable level are rough estimates which will vary depending on the consumption levels set as a sustainable target. In order to development recommendations for sustainable food consumption the level of sustainable meat consumption needs to be studied more extensively in the future.

5. Conclusion

- Depending on the type of statistics and way of presenting data, meat consumption levels per capita may differ by a factor two, or more.
- For a correct utilisation and interpretation of meat consumption data and LCA's of meat it is crucial to specify the functional unit, i.e. to clearly define if it refers to raw meat including or excluding bones, including losses along the food chain, or after weight reduction by cooking.
- The need for reductions in current meat consumption to meet climate and health goals is estimated to 0-75% and 0-50%, respectively, depending on the region.
- The level of sustainable meat consumption needs to be studied more extensively in the future.

6. References

Berners-Lee, M., Hoolohan, C., Cammack, H., Hewitt, C.N., 2012. The relative greenhouse gas impacts of realistic dietary choices. Energy Policy 43, 184-190.

Carlsson-Kanyama, A., Gonzalez, A.D., 2009. Potential contributions of food consumption patterns to climate change. Am J Clin Nutr 89, 1704-1709.

Cederberg, C., Meyer, D., Flysjö, A., 2009. Life cycle inventory of greenhouse gas emissions and use of land and energy in Brazilian beef production. Report No 792. Swedish Institute for Food and Biotechnology. Göteborg, Sweden.

EIPRO, 2006. Analysis of the life cycle environmental impacts related to the total final consumption of the EU25. European Commission Technical Report. EUR 22284 EN.

European Commission, 2007. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions – Limiting global climate change to 2 degrees Celsius - The way ahead for 2020 and beyond. Brussels, Belgium.

FAO, 2006. Livestock's long shadow: environmental issues and options. Food and Agriculture Organization of the United Nations. Rome, Italy.

FAO, 2009. The state of food and agriculture: Livestock in balance. Food and Agriculture Organization of the United Nation. Rome, Italy.

FAO, 2011. Global food losses and food waste- extent, causes and prevention. Food and Agriculture Organization of the United Nations. Rome, Italy.

FAOSTAT. Food supply quantity of total meat, data for 2007. Food and Agriculture Organization. Available at: http://faostat.fao.org (28.05.2012).

Garnett, T., 2008. Cooking up a storm. Food, greenhouse gas emissions and our changing climate. Food Climate Research Network. Surrey, UK.

Garnett, T., 2009. Livestock-related greenhouse gas emissions: impacts and options for policy makers. Environ Sci Policy 12, 491-503.

Garnett, T., 2011. Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? Food Policy 36, 23-32.

Hallström E., Ahlgren, S., Börjesson, P., 2011. Challenges and opportunities for future production of food, feed and biofuel- A land use perspective. Report No 74. Department of Environmental and Energy Systems Studies. Lund University. Lund, Sweden.

Hallström, E., Börjesson, P., 2012 (unpublished results). Meat Consumption Statistics - Reliability and Discrepancy. Sustainability: Science, Practice, and Policy.

IPCC, 2007. Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge, United Kingdom and New York, USA.

KF & ICA Provkök, 2000. Mått för mat. Provköken förklarar. ICA bokförlag. Västerås, Sweden.

McMichael, A.J., Powles, J.W., Butler, C.D., Uauy, R., 2007. Energy and health 5 - Food, livestock production, energy, climate change and health. Lancet 370, 1253-1263.

Naska, A., Berg, M.A., Cuadrado, C., Freisling, H., Gedrich, K., Gregoric, M., Kelleher, C., Leskova E., Nelson, M., Pace, L., Remaut, A.M., Rodrigues, S., Sekula, W., Sjostrom, M., Trygg, K., Turrini, A., Volatier, J. L., Zajkas, G., Trichopoulou, A., 2009. Food balance sheet and household budget survey dietary data and mortality patterns in Europe. Br J Nutr 102, 166-171.

Nilsson, K., Sund, V., Florén, B., 2011. The environmental impact of the consumption of sweets, crisps and soft drinks. Nordic Council of Ministers. Tema Nord. Copenhagen, Denmark.

Sonesson, U., Davis, J.,Ziegler, F., 2010. Food Production and Emissions of Greenhouse Gases. An overview of the climate impact of different product groups. Report no 802. Swedish Institute for Food and Biotechnology, Göteborg, Sweden.

Stern, N., 2006. The Economics of Climate Change - The Stern Review. Cambridge University Press. Cambridge, UK.

UNEP, 2010. The Emissions Gap Report: Are the Copenhagen Accord Pledges Sufficient to Limit Global Arming to 2 degrees or 1.5 degrees? A preliminary assessment.

U.S. Department of Agriculture/U.S. Department of Health and Human Services, 2010. Dietary Guidelines for Americans 2010. 7th Edition. Washington DC, USA.

Westhoek, H., Rood, T., van den Berg, M., Janse, J., Nijdam, D., Reudink, M., Stehfest, E., 2011. The protein puzzle. The consumption and production of meat, dairy and fish in the European Union. PBL Netherlands Environmental Assessment Agency. Hauge. Netherlands.

WHO, 2003. Food based dietary guidelines in the WHO European region. Copenhagen. Denmark.

World Cancer Research Fund/American Institute for Cancer Research, 2007. Food, Nutrition, Physical Activity, and the Prevention of Cancer: a Global Perspective. Washington DC, USA.

Assessing the optimum level of detail for secondary GHG emissions databases

Ricardo Teixeira*

Bluehorse Associates, 12 rue Soyer, 92200 Neuilly-sur-Seine, France
E-mail: *ricardo.teixeira@bluehorseassociates.com*

ABSTRACT

Life Cycle Assessment (LCA) is a prohibitive initiative for companies with a high number of products because LCA can prove to be time- and resource-intensive. The LCA process can possibly be rendered less overwhelming for companies by simplifying the data-sets by using "emission factors" instead of inventories. In this paper, we test if simplified data (emission factors measured in kg CO_2e per kg of product) can provide statistically solid results in LCA studies in the agri-food sector. We build a linear statistical model to cluster agri-food records with similar average CO_2e emissions.

Preliminary results show that it is possible to obtain what we call "magic numbers" - statistically significant average CO_2e emission factors for product clusters for certain classes of products. Data clustered in high-level groups (e.g. butter) has high variance; but lower-level clustering (e.g. butter with herbs and butter without herbs) results in statistically significant "average" emissions.

Keywords: Life Cycle Assessment, database management, simplified LCA, agri-food sector, carbon footprint

1. Introduction

The past years have seen Life Cycle Assessment (LCA) moving away from academic studies and being implemented into companies' daily operations such as research and development. Product labelling using LCA is being implemented on a larger scale. Private initiatives by retailers like Tesco (Clare and Little, 2011) and the French government pilot (ADEME, 2011) are signs of this evolution, which has created the demand for LCA tools capable of providing results for a large volume of products. Tesco found that complex methods, or even traditional LCA, may be too time- and resource-intensive, thus the reason they terminated the initiative to label each product using traditional LCA methods. To overcome this barrier, a more practical, business-oriented side to LCA is required to simplify without compromising accuracy. Furthermore, for objectives such as getting in-depth knowledge on the supply chain, estimates based on full LCA methods may not provide the appropriate level of detail. There is a difference between accuracy (calculating the right number) and precision (many decimal places).

Given the needs of larger scaled agri-food companies this begs the question: How far simplification can go in LCA tools? On the tool interface and outputs side, there is virtually no ceiling to how user-friendly tools that can be built. On the methodological side, the debate is on-going. On the database side, however, there is an understanding that precision is needed - data must be specific and primary. The recurring fear of "garbage in, garbage out" has overshot the quality standards for primary data. ISO 14040 (ISO 1997) recommends the use of emissions-base unit processes so that every record in the database is a life cycle with traceable inputs and outputs. ISO-compliant tools, commonly known as "full LCA" tools, dominate the market while using this principle. A survey done on LCA practitioners in the agri-food sector last year revealed that the most commonly used tools are still "full LCA" tools such as SimaPro by PRe Consultants and Gabi by PE International (Cooper and Fava, 2006; Teixeira and Pax, 2011).

Lately, the makers of simplified tools have been trying to educate the market on another approach. One of the strategies used is referring to built-in databases with emissions for pre-recorded life cycles (Weitz et al., 1996; Graedel, 1998). Instead of giving users the possibility of rebuilding a life cycle for a product (e.g, butter – see below for more on this example), the database comprises pre-established life cycle impact assessment (LCIA) results. Users choose the most appropriate one for their case (e.g, choose between CO_2e emissions for conventionally produced simple butter in the Netherlands and for organic herb butter in Switzerland).

Graedel (1998, cit. in Lifset, 2006) argued that simplified LCA can provide around 80% of the findings from full LCA. To this day, however, a study is needed to justify the impact of simplified procedures in final LCA results and confirm or refute Graedel (1998). A true test cannot consist of running an LCA for one product using both simplified and full LCA procedures because this is always case specific and no general conclusions can be drawn. In particular cases great differences may occur, but what matters is if on average, for a large number of LCAs, the simplified methodology yields high variability or an inherent bias on results; therefore an alternative method must be found.

In this paper we propose an alternative approach to test the hypothesis that simplified data can still provide accurate, statistically solid results in LCA studies.

2. Methods

Our approach consists on analysing statistical records on carbon emissions for agri-food products. For this study to be successful, we needed a large number of secondary data that included many different sources to ensure maximum heterogeneity.

This compilation is found in the database for the Carbonostics tool (Carbonostics, 2011), which is the largest available built-in database for agri-food products (Verdantix, 2011). The database compiles more than 1,400 pre-recorded final LCIA results for CO_2e emissions from data providers such as ADEME (2010), CleanMetrics (2010), CLM (2010), the Danish LCA Food Database (Nielsen et al., 2003), DEFRA (2012), ecoinvent (Frischknecht and Rebitzer, 2005) and ESU (2012). This number of data records is much higher than any number mentioned in any of the calculators referred in Amani and Schiefer‘s (2011) survey of tools. Each record in the database has been peer-reviewed and validated by the Swiss NGO MyClimate (2012). Many assumptions built into the data records by different providers may be contradictory or inconsistent with each other, which serves the purpose of our analysis by introducing even more variability. For example, while some records include transportation steps, others do not. We did not remove these inconsistencies since the objective is to maximize the variance and replicate the error a user would make when choosing the wrong record from the database. The choice of food products is particularly suited to our objective, since variability between specific products of the same time is reputedly high, e.g the amount of fertilisers and yields change between farmers even in the same region and with the same general production method.

The Carbonostics database hierarchizes records by grouping them in three levels:

1. General category (e.g, dairy, vegetables, oils, meat, crops). This is roughly equivalent to product type. The number and distribution of these records is shown in Figure 1;
2. Product type within category (e.g., butter, buttermilk, milk – all within the category dairy);
3. Product variant within type (e.g, conventional plain butter in Europe, organic butter with herbs in Europe – all within the type butter).

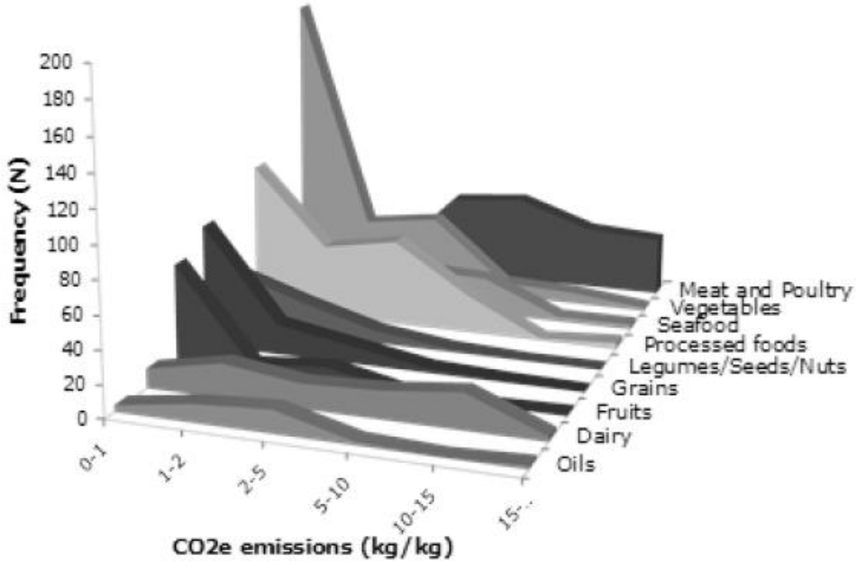


Figure 1. Number of records per type and CO_2e emissions dispersion.

Our statistical method consisted on building one linear model at each of these three levels. We determined the intra-level clustering, i.e, how many sub-divisions in sub-groups are needed, by running a cluster analysis, using the Calínski and Harabasz pseudo-F index stop rule to determine the number of clusters, for each level. In each model, CO_2e emissions are the dependent variable and the independent variables are a group of binary “dummy” variables that represent the category, type or variant. The model thus calculates averages and standard errors for each group of records, and determines their statistical significance.

Since all variants are defined by geographical region and method of production, we included both as control variables, i.e. we included them as binary variables in the model. In the results shown next, we cropped data to show results only using agricultural records that were produced conventionally and in Europe (N=878 records). We show preliminary results for the level 1 model using all categories, for level 2 using only the dairy type, and for level 3 also on the dairy type but including variant-specific variables. We conducted all the statistical calculations in software Stata v10.0 (StataCorp., 2007).

3. Results

We began by grouping records at each level. Starting from the groupings in Carbonostics, we clustered groups with similar average emission records, and not enough data to be significantly different, e.g. at level 1: fruits, grains, legumes, vegetables and processed foods all fall within the same range of emissions. Using the same procedure, we also determined which variant-specific characteristics are statistically relevant, i.e. average CO_2e emissions are significantly different between sub-groups). For significant variants (e.g., separating plain butter from butter with herbs), we included new binary variables at level 3.

Preliminary results are summed up in Figure 2. The model at level 1 (N=878) has a relatively low adjusted R^2 (0.404), and the standard error of the averages for each category are relatively high. However, the dairy type specific model at level 2 (N=75) has a much higher R^2 (0.846) and the averages start to clearly identify product types. We go to level 3 by separating plain butter (which has a higher average emission factor) from butter with herbs (which was clustered together with buttermilk, yoghurt, ice cream and milk at the lower end of the spectrum). Then, the R^2 is much higher (0.916), and errors clearly decrease.

We tested including more granularity in the analysis (e.g., dividing the group "cheese" in different types of cheese, dividing "milk" in plain milk and flavor milk). However, this did not increase the statistical fit of the model.

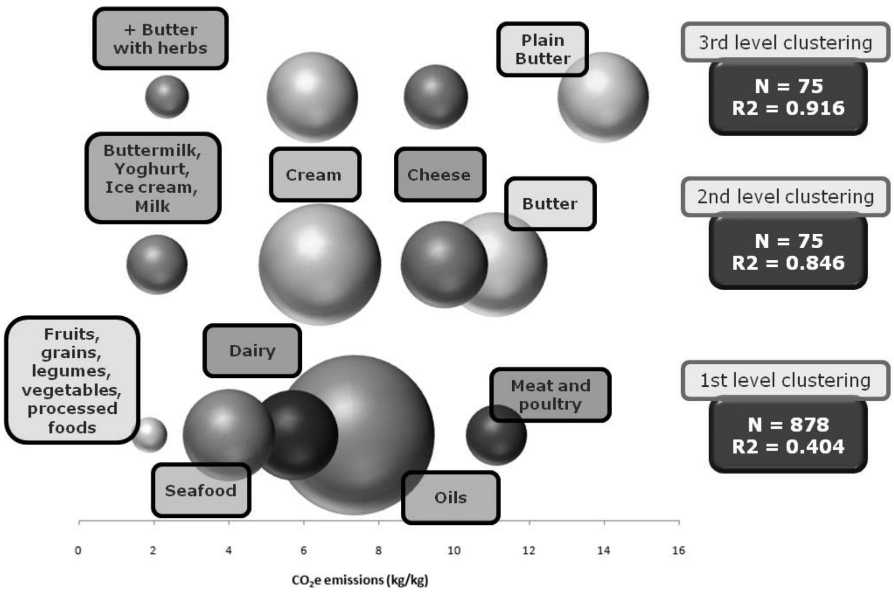


Figure 2. Results of the statistical analysis per level (bottom is category-level, up is sub-variant level) for conventional production in Europe; the centre of each bubble is the average; size depicts the standard error.

4. Discussion

Preliminary results (Figure 2) show that models 1 and 2 are not sufficiently reliable to explain most of the variance within product categories and to capture statistically significant different between categories. Modelling approach 3 provided interesting results, with enough statistical robustness. Using only four clusters for the whole category of dairy products, we found statistically significant average CO_2e emission factors. Production method was also an important variable to consider. Geographical region (full results not shown here) was not relevant for most types even though this may be due to the fact that most of the records in the database are for Europe. In this study, 90% of records included are European, but a much lower percentage of worldwide agricultural and food production is European.

It is important to underscore that it could seem like the deeper we go into the levels, the more statistical reliability is achieved; however, this is not true. There seems to be a "glass ceiling". From this point on, more detail does not provide a better statistical fit. Breaking down sub-types would be: (1) unnecessary for categories with many data records, because the standard error is overall low (20-30% of the mean), which means we hit a statistical "glass ceiling", or (2) impossible due to lack of data records for each sub-type. The combination of (1) and (2) shows that the main barrier to using secondary emissions records databases for LCA identified in this paper is lack of data, not heterogeneity. Using emissions records at a macro scale was not found to be problematic, . The study revealed that that even compiling most available databases is insufficient to cluster data in most product types. This result is perfectly consistent with the replies to the agri-food LCA practitioners survey that identified lack of data as one of the main obstacles to LCA (Teixeira and Pax, 2011).

After its completion, the main output of this work will be a table that, for each clustering group of agri-food products produced with each method in each region, provides the average CO_2e emissions and an error estimate. The results in that table can fit several purposes in LCA, namely:

- As a reference or benchmark in future LCA studies;
- For the environmental assessment of large-scale agricultural policies;
- In hybrid input-output analysis (IO-LCA), which uses data at this level of detail;
- In simplified LCA methods and tools for screening calculations, as well as streamlined simple algorithms. Although the data in this paper can hardly replace individual data records, it will be very useful to suppress data gaps or make informed choices between available emission records;
- To help developers of Product Category Rules set category-specific rules and hierarchies for data quality.
- To help LCA practitioners determine a product-specific hierarchy for data quality criteria. For example, if records from a certain category are very sensitive to geographic region, then that attribute must be privileged when choosing from secondary databases.

It should also be noted that it is not advised to use this type of data when users have objectives such as process optimisations, labelling/external communication and comparative assertions (except for preliminary hot-spot analyses); because the data can only be used for generic macro assessment of static situations. Changes in the baseline numbers will change the structure of the data itself and comparisons must be updated.

5. Conclusion

In this study, we tested whether it is possible to use “magic numbers” in secondary databases for LCA studies and, if acceptable, we sought to determine the optimum level of detail. These results suggest that, even though there is a high variability in the way agri-food products are produced, there are representative averages that may be used when LCA practitioners, depending on their objective, provided sufficient statistical work is done beforehand. Our work also exposes the dramatic lack of LCA data today. More than tools, methodological developments or standards, practitioners identify lack of the data as the main problem in LCA. This work shows analytically why.

6. References

ADEME, 2010. Guide des Facteurs d’Emission, Version 6.1, Chapitre 6 – Prise en compte des autres produits entrants: produits servant aux activités agricoles, d’élevage, et agro-alimentaires (“Emission factor guidelines, Version 6.1, Chapter 6 – Accounting for other products: agriculture, livestock and food", in English). ADEME, Paris

ADEME, 2011. Affichage environnemental des produits de grande consummation. [Environmental labeling of large consumption products.] ADEME, Paris. Available from: http://affichage-environnemental.afnor.org/. Accessed December 2011.

Amani P. and G.Schiefer, 2012. Data Availability for Carbon Calculators in Measuring GHG Emissions Produced by the Food Sector. Int. J. Food System Dynamics 2, 392-407.

Carbonostics. 2012. http://www.carbonostics.com/. Accessed March 2012.

Clare, H. and Little, M.,2011. Breaking the link between consumption and emissions: improving corporate environmental performance through carbon footprinting. Proceedings of the 17th SETAC Europe LCA Case Studies Symposium, 28 February – 1 March, Budapest.

Cleanmetrics, 2010. http://www.cleanmetrics.com/html/lci_methodology.htm. Accessed January 2010.

CLM, 2010. http://www.clm.nl/index_eng.php?id=methodology. Accessed January 2010.

Cooper, J.S. and Fava, J., 2006. Life-Cycle Assessment Practitioner Survey Summary of Results. J. Ind. Ecol. 10, S12-S14.

DEFRA, 2012. http://www.defra.gov.uk/. Accessed January 2012.

ESU, 2012. http://www.esu-services.ch/. Accessed January 2012.

Frischknecht, R. and Rebitzerb, G., 2005. The ecoinvent database system: a comprehensive web-based LCA database. J. Clean. Prod. 13, 1337–1343.

Graedel, T., 1998. Streamlined Life-cycle Assessment. Prentice Hall, Upper Saddle River, New Jersey.

ISO, 1997. ISO 14040, Environmental management: Life cycle assessment—Principles and framework. International Organization for Standardization, Geneva.

Lifset, R.J., 2006. Industrial Ecology and Life Cycle Assessment: What's the Use? Int. J. Life Cycle Ass. 11, 14 – 16.

MyClimate, 2012. http://www.myclimate.org/. Accessed April 2012.

Nielsen, P.H., Nielsen, A.M., Weidema, B.P., Dalgaard R. and Halberg, N., 2003. LCA data on basic food products produced and consumed in Denmark. Danish Institute of Agriculture Sciences. Available from: http://www.lcafood.dk.

StataCorp, 2007. Stata Statistical Software: Release 10. StataCorp LP, College Station, TX.

Teixeira, R. and Pax, S., 2011. A Survey of Life Cycle Assessment Practitioners with a Focus on the Agri-Food Sector. J. Ind. Ecol. 15, 817-820.

Verdantix, 2011. Smart Inovators Product LCA Software Report. Verdantix Ltd., London.

Weitz, K., Todd, J., Curran, M. and Malkin, M., 1996. Streamlining Life Cycle Assessment considerations and a report on the state of practice. Int. J. Life Cycle Ass. 1, 79-85.

A protocol for approaching uncertainties in life cycle inventories

P.J.G Henriksson*, J.B. Guinée, R. Heijungs, A. de Koning

Institute of Environmental Sciences, Leiden University, the Netherlands
* Corresponding author. E-mail: *Henriksson@cml.leidenuniv.nl*

ABSTRACT

Results of life cycle assessments (LCAs) of identical food production systems often experience great spread in outcomes between studies. Part of this divergence relates to the lack of standardised methodological choices, including different views on system boundary setting, allocation, carbon offsetting and storage, data sourcing, end-of-life of products, land use change, characterisation factors, etc. However, studies also experience great divergence as a result of inventory data sourcing. While a common set of "benchmarking" standards could be agreed upon with respect to adopted methodology, underlying inventory data will chronically be subject to a dynamic reality and imperfect measurements.

Uncertainty and/or variability in LCIs derive from primary foreground data (field-data), secondary foreground data (literature data) and background data (databases). Given the different characteristics of primary and secondary data, a practical approach to unit-process data, using soybeans as an example, is therefore here presented. The example focuses on the selection of secondary foreground data. An approach for assigning each unit-process flow with a weighted mean, a holistic standard deviation and a distribution is suggested. These parameters could serve as inputs for Monte-Carlo simulations and many other uncertainty tests.

The importance of defining the uncertainty parameters in unit process data for use in any uncertainty test is here discussed as a pivotal step towards the inclusion of quantitative uncertainty in LCA results; rather than focusing on a specific test. Uncertainty ranges in LCIs, to date, have largely been derived from a Numeral Unit Spread Assessment Pedigree (NUSAP) approach. The NUSAP approach, however, only generates estimates of the representativeness of datasets, excluding inherent uncertainty or variability. Therefore, a method for weighting inventory flows between secondary data sources is presented where representative means can be produced alongside inherent standard deviation parameters. The proposed approach also helps identify cross-references and outdated values between studies. Certain assumptions in the ecoinvent database with influence on LCAs of food production systems are also highlighted, e.g. the exclusion of variability in yields within their pre-defined standard deviations. Knowing the uncertainty (including variability) of LCI results and related impacts, is crucial for justifying results and derived decision-making. Knowledge in this field is still fragmented although growing. We will here illustrate a new critical approach to secondary foreground data and quantify the uncertainty of unit process data.

Keywords: LCI, unit-process, uncertainty, variability, NUSAP

1. Introduction

Uncertainty is an intrinsic part of the scientific method and defines the quality of a prediction (Palmer and Hardaker, 2011). In the field of life cycle assessment (LCA), uncertainties have largely been addressed qualitatively, if at all, despite a long-time desire for its quantitative inclusion (Ross et al., 2002). With many historical hurdles being overcome, e.g. lack of data and limited computer power, the LCA community now faces a fundamental challenge, that of moving beyond point values towards uncertainty ranges. With much emphasis focusing on which methodology to adopt in order to produce uncertainty ranges (Heijungs 1996; Huijbregts et al., 2001; Lloyd and Ries, 2007), less attention has been given to the origin of unit process uncertainty parameters and what these should embed.

The most commonly used database at present, ecoinvent v2.2, has defined uncertainty ranges to its inventory data. These ranges are the products of the Numeral Unit Spread Assessment Pedigree (NUSAP) approach derived from Weidema and Wesnaes (1996) who introduced the concept of process data quality indicators. However, as first presented by Funtowicz and Ravetz (1990) the NUSAP approach was originally intended to evaluate the system uncertainties and decision stakes related to applied sciences and professional consultancy (Ravetz, 1999). The NUSAP outcomes, consequently, only provide an indication of the representativeness of a dataset to its proposed application, and complements, rather than replaces, inherent uncertainty and variability within datasets.

Inherent variability is especially prominent in the food production sector where production is directly governed by natural fluctuations and where models often are based upon empirical data (Schau and Fet, 2008; Röös et al., 2010; Johnson et al., 2011). Frischknecht et al., (2007) highlight this by acknowledging that sample sizes exceeding 100 samples might be needed in order to retrieve reliable results for agricultural systems. Such extensive primary datasets are, however, rarely available in LCIs, nor are inventories characterising complete sets of economic and environmental flows.

As part of the on-going SEAT project (www.seatglobal.eu), primary process data have been collected for aquaculture farms in Bangladesh, China, Thailand and Vietnam. Additional primary data have also been collected for feed mills, hatcheries, nurseries, processing plants, fishmeal factories and reduction fisheries in Asia. As coverage of Asian processes is currently limited in existing background databases, most processes need to be modelled based upon literature sources (secondary foreground data). In order to generate representative inventories and inherent uncertainty ranges for secondary foreground data from different sources, we here present a methodology for selecting and weighting inventory values, and in the meantime produce estimates for inherent uncertainty parameters. The methodology will be practically exemplified by Brazilian soybean production, as soybeans often constitute more than 30% of aquaculture feeds.

2. Materials and methods

As the example here is assumed to rely upon the ecoinvent v2.2 database for background data, the aim is to be consistent with choices made therein. Therefore, in parallel with ecoinvent v2.2, the NUSAP approach described by Weidema and Wesnaes (1996) was adopted, categorising the origins of representativeness into *reliability*, *completeness*, *temporal correlation*, *geographical correlation* and *further technical correlation*. The additional category of *sample size* and the assigned uncertainty factors suggested by Frischknecht et al., (2007) were also implemented.

As an initial step, a decision tree was developed for foreground data, with a general distinction between primary and secondary data (Table 1). The decision tree guides the practitioner towards recommended approaches when sourcing process data, assuming a default log-normal distribution of datasets. Log-normal distributions are favoured as to avoid negative values, better represent large variances and to be consistent with the ecoinvent v2.2 database. Primary data are, moreover, prioritised as they are assumed to be up-to-date, highly relevant, and provide a higher level of detail. Secondary data are previously published data describing the process in focus, where the final selection of values should be in-line with the goal and scope definition of the specific study at stake. Where relevant multiple secondary data sources exist, a weighted mean approach is recommended. Each outcome in the decision tree defines a recommended type of mean, standard deviation and distribution, or alternative approaches in certain cases.

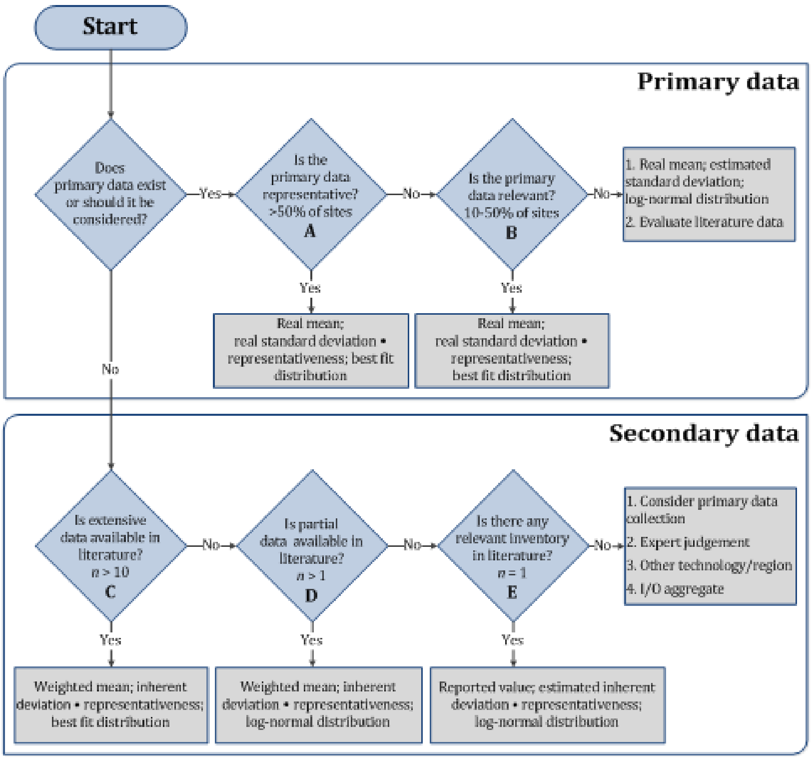


Figure 1. Decision tree for sourcing unit process data for foreground processes.

Weighted means between secondary foreground parameters were calculated using Weighted means ($\bar{x}_{(wt)}$) can be calculated from the aggregated means (x_i) of samples (n), with the NUSAP derived geometric standard deviation used as the weighting factor (w_i). The inherent variability can, in turn, be calculated amongst the different parameters where several sources exist. Missing values are excluded from the calculations and null values are calculated using 10% or the smallest value, due to the limitations of the logarithmical scale. To generate the final overall uncertainty parameter, the inherent geometric standard deviation needs to be summed with the most representative NUSAP indicator for each inventory flow (Frischknecht et al., 2007).

$$\bar{x}_{(wt)} = \exp\left(\frac{\sum_{i=1}^{n} w_i \ln x_i}{\sum_{i=1}^{n} w_i}\right) \qquad \text{Eq. 1}$$

Six inventories for Brazilian soybeans highlighted in da Silva et al., (2010), with the use of Schmidt (2007) to represent Dalgaard (2008), and the addition of FAOSTAT data (http://faostat.fao.org) were included in this study. Each inventory flow was sourced to its origins in order to avoid double counting of cross-references and to determine the actual representativeness of each inventory flow. The resulting weighted means and inherent standard deviations were then calculated amongst the original values (excluding cross-references and assumptions). Given space limitations, we here only present and discuss the inventories for fertilisers, fuel and resulting yields.

3. Results

The inventories supporting the six different soybean inventories and FAO data are reported in Table 1. All studies rely upon a divergent set of inventories describing an identical process. Many of the values are, moreover, cross-references of previous publications. The amount of diesel used, for example, stems back to two single primary data points, Ostermayer 2002 in Jungbluth et al., (2007) and Cederberg (1998). Ecoinvent adopts the average of these two publications (55.25 kg assuming 0.85 kg litre^{-1} diesel) and refers to an FNP Agro report from 2000 for fertiliser use (Jungbluth et al., 2007). Schmidt (2007) on the other hand adjusted Argentinian primary data to a Brazilian scenario. This study was therefore not considered as a primary data source for Brazilian soybeans. Da Silva et al., (2010) unfortunately failed to report on inventory values used and could therefore not be included. Most studies use the average yield over five years reported by FAO in order to eliminate any annual fluctuations.

Table 1: Unit process data for six different articles and FAOSTAT data for one hectare of Brazilian soybean crops

	unit	Ecoinvent 2.2 Database	Cederberg 1998 Report	FAOSTAT 2002 FAO	Cederberg and Flygsjö 2004 Report	Schmidt 2007 PhD thesis	Lehuger et al., 2009 Article	Cavalett and Ortega 2010 Article
Economic inputs								
Diesel	kg	55.25	51	-	55.25	44.63	-	55.25
Nitrogen, as N	kg	3.11	0	3.73	8	0	8	0
Phosphorus, as P	kg	26.18	17.46	28.8	31	0	31	33.8
Potassium, as K	kg	24.9	33.2	51.46	57	20	57	65.4
Economic outputs								
Soybeans	kg	2 544	2 200	2 613	2 500	2 680	2 500	2 830

With all cross-references and non-primary data points excluded, diesel was narrowed down to two primary data sources and fertilisers to five values. The economic input data were then standardised to one tonne of soybeans according to reported yields in order to aggregate values into a single weighted mean (Table 2).

Table 2: Reported inputs of diesel and fertilisers in the only studies presenting real novel field data, and their accompanying standard deviations in brackets as estimated using the NUSAP approach. The calculated weighted geometric mean and the inherent geometric standard deviation amongst the values are presented in the rightmost columns. All values are normalised to the production of one tonne of soybeans in Brazil.

	FNP 2000/ Ostermayer 2002	Cederberg 1998	FAO 2002	Cederberg and Flygsjö 2004	Cavalett and Ortega 2010	Wt. mean	Inherent $STDEV_g$
Diesel (kg)	23.4 (1.34)	23.2 (1.31)				**23.3**	**1.01**
Nitrogen (kg)	1.2 (1.27)	0 (1.24)	1.4 (1.22)	3.2 (1.27)	0 (1.10)	**1.4**	**3.48**
Phosphorus (kg)	10.3 (1.27)	7.9 (1.24)	11.0 (1.22)	12.4 (1.27)	11.9 (1.10)	**10.6**	**1.19**
Potassium (kg)	9.8 (1.27)	15.1 (1.24)	19.7 (1.22)	22.8 (1.27)	23.1 (1.10)	**17.6**	**1.43**

The inherent variability for yields was calculated using the yields reported by FAO between 2001 and 2005, the same time range adopted by ecoinvent. Calculating the geometric mean over these five years yielded an average harvest of 2 537 kg ha^{-1} and an inherent geometric standard deviation of 1.11.

2. Discussion

Limited data availability remains a generic problem in LCIs, mainly due to the great resources needed to collect primary data. Where datasets are available, however, they often describe limited geographical areas where practices in e.g. agriculture differ depending upon micro-climate and soil characteristics. The inconsistencies in the inventories presented above may therefore be the result of six alternative soybean processes rather any nationwide averages. In search for such more general processes, the methodology presented here enables secondary foreground data to be critically evaluated. When averaging unit process data into, for example, country-wide averages, different economic and environmental flows may also be used complementary to produce better characterised processes. This may be values for water use reported in some studies, while inventories of pesticide use only are available in other studies. Moreover, the proposed approach enables unit process data to move beyond point values and minimizes the use of unrepresentative parameters. It also helps to identify areas of great uncertainty and in the process allows for inherent variability to be calculated.

With many incentives to include quantified uncertainties into LCA results, an initial step will be to define basic uncertainty parameters in the LCI phase, allowing for different kinds of uncertainty tests/methods to be applied (e.g. not only Monte Carlo analysis). Including most sources of uncertainty in these parameters is crucial, as they will determine the application and outcome of any later implemented test/methodology. With a wide range of sources and types of uncertainties, we herein highlight the distinction between inherent standard deviations and representativeness. While the latter easily can be estimated, the former are not always available for unit processes. Even when available, inherent uncertainties are often neglected, as in the above example of soybean yield averages over several years. When calculated using the approach proposed above, inherent standard deviations often exceed those of representativeness. For example, the geometric standard deviation amongst the reported uses of potassium in the studies discussed above, amounts to 1.43 compared to an estimated geometric standard deviation of 1.1-1.27 for representativeness. The inherent geometric standard deviations of nitrogen showed even greater divergence, but these are somewhat distorted by the reported null values. Temporal inherent variability is also relevant for food production systems, where the yield average over five years, reported by FAO, exhibit a geometric standard deviation of 1.11. This is largely the result of a close link between food production systems and stochastic natural events. Agriculture practices also exhibit large model uncertainties when calculating field emissions, which add further inherent uncertainty to this field of LCA research.

Interconnected (co-variation) parameters (e.g. the amount of fertiliser used and resulting N_2O emissions) were here not accounted for, resulting in an overestimation of the uncertainty. In the meantime, inherent uncertainties are not always considered in the background database, thereby underestimating the degree of overall uncertainties. Neither were the temporal correlations embedded in the database representative of present time, as they are benchmarked at the time of release of the database. With some NUSAP categories being quantifiable (e.g. less than 3 years), others remain open for interpretation which may also result in interpretation errors.

5. Conclusion

Inventory data remains a major source of uncertainty for LCA results, where LCIs describing the same production system often experience large differences at the unit process data level. Many of the values used are also cross-references which sometimes date back over ten years. Scrutiny is therefore needed when developing process datasets, as is better reporting of the origins of data, standard deviations around means and identification of known sources of uncertainty. Using the standardised decision tree for data sourcing and weighting means amongst studies can therefore result in more representative and rigid results.

Quantitative uncertainties need to be included in LCAs in order to statistically validate conclusions and strengthen the credibility of LCA results. The herein proposed approach presents one way for entailing uncertainty parameters that complement each other (inherent and representativeness), rather than replacing each other. A standardised vocabulary is also needed for further uncertainty discussions within the field of LCA. Here we have e.g. shown that a notable distinction is needed between inherent uncertainty and variability, and representativeness. Food production systems are especially sensitive to inherent deviations, especially as yields which are subject to annual fluctuations often represent the functional unit.

6. References

Cederberg, C., 1998. Life Cycle Assessment of Milk Production – a Comparison of Conventional and Organic Farming. SIK Report 643. Gothenburg.

Dalgaard, R., Schmidt, J., Halberg N., Christensen P., Thrane M., Pengue W.A., 2008. LCA of soybean meal. Int J of Life Cycle Assess, 13:240-254.

Frischknecht, R., Niels, J., Althaus, H-J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M. and Wernet, G., 2007. Overview and Methodology. Ecoinvent Report No. 1. Dübendorf.

Funtowicz, S.O., Ravetz, J.R., 1990. Uncertainty and Quality in Science for Policy. Kluwer Academic Publishers, Dordrecht.

Heijungs, R., 1996 Identification of key issues for further investigation in improving the reliability of life-cycle assessments. J Clean Prod. 4:159-166.

Henriksson, P.J.G., Guinée, J.B., Rene, K., Snoo, G.D., 2011. Life cycle assessment of aquaculture systems – a review of methodologies. Int J Life Cycle Assess. 17(3):304-313

Huijbregts, M.A.J., 1998. Application of Uncertainty and Variability in LCA Part I□: A General Framework for the Analysis of Uncertainty and Variability in Life Cycle Assessment. Int J Life Cycle Assess. 3:273-280.

Huijbregts, M.A.J., Norris, G., Bretz, R., Ciroth, A., Maurice, B., von Bahr, B., Weidema, B., de Beaufort, A.S.H., 2001. Framework for Modelling Data Uncertainty in Life Cycle Inventories. Int J Life Cycle Assess. 6:127-132.

Johnson, D.R., Willis, H.H., Curtright, A.E., Samaras, C., Skone, T., 2011. Incorporating uncertainty analysis into life cycle estimates of greenhouse gas emissions from biomass production. Biomass Bioenerg. 35:2619-2626.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., Sutter, J., 2007. Life Cycle Inventories of Bioenergy. Ecoinvent Report No. 17. Dübendorf.

Lloyd, S.M., Ries, R., 2007. Characterizing, Propagating, and Analyzing Uncertainty in Life-Cycle Assessment: A Survey of Quantitative Approaches. J Ind Ecol. 11:161-179.

Palmer, T.N., Hardaker, P.J., 2011. Handling uncertainty in science. Philos T Roy Soc A. 369:4681-4684.

Ravetz, J.R., 1999. What is Post-Normal Science. Science. 31:647-653.

Ross, S., Evans, D., Webber, M., 2002. How LCA studies deal with uncertainty. Int J Life Cycle Assess. 7:47-52.

Röös, E., Sundberg, C., Hansson, P.-A., 2010. Uncertainties in the carbon footprint of food products: a case study on table potatoes. Int J Life Cycle Assess. 15:478-488.

Schau, E.M., Fet, A.M., 2008. LCA Studies of Food Products as Background for Environmental Product Declarations. Int J Life Cycle Assess. 13:255-264.

Schmidt, J.H., 2007. Life Cycle Assessment of Rapeseed Oil and Palm Oil. PhD thesis. Aalborg University.

da Silva, V.P., van der Werf, H.M.G., Spies, A., Soares, S.R., 2010. Variability in environmental impacts of Brazilian soybean according to crop production and transport scenarios. J Environ Manage. 91:1831-1839.

Weidema, B., Wesnaes, M.S., 1996. Data quality management for life cycle inventories—an example of using data quality indicators. J Clean Prod. 4:167-174.

Quantifying environmental impacts and their uncertainties for UK broiler and egg production systems

Ilkka Leinonen[1,*], Adrian Williams[2], Ilias Kyriazakis[1]

[1] School of Agriculture, Food and Rural Development, Newcastle University, Newcastle upon Tyne, NE1 7RU, UK
[2] School of Applied Sciences, Cranfield University, Bedford, MK43 0AL, UK
[*] Corresponding author. E-mail: *ilkka.leinonen@newcastle.ac.uk*

ABSTRACT

The environmental burdens of the main UK broiler and egg production systems were quantified using a systems modelling approach with Monte Carlo simulations for statistical analyses. Feed was the main component of the global warming potential in all broiler and egg systems. Manure was a major source of acidification and eutrophication potentials. The length of the production cycle was lowest in the standard indoor broiler system, and therefore the feed consumption and manure production were also lowest. This caused statistically significant differences in most of the impact categories between the broiler systems. The number of birds required to produce equal amount of eggs, and the amount of feed consumed per bird were highest in organic and lowest in the cage system. These general differences in productivity affected the environmental impacts of the different egg systems, although in some impact categories the differences were not always significant.

Keywords: broiler, egg, uncertainty analysis, global warming potential, energy use

1. Introduction

UK poultry production, including broilers and eggs, has been identified as being relatively environmentally-efficient, per unit weight, compared to the production of other animal commodities (Williams et al., 2006). However, like all agricultural systems, any current poultry system has scope to improve, and thus has the potential to reduce its environmental impacts. For example, with an annual production of 8862 million eggs (about 0.4 billion kg) produced in the UK (Defra, 2009) and 61 billion kg produced worldwide annually (FAO, 2011), the egg laying systems are likely to be significant contributors to both resource use and environmental burdens, and therefore have also potential for large scale reduction of impacts such as greenhouse gas emissions.

The aim of the current study was to apply the LCA method "from cradle to farm gate", to quantify the environmental burdens of the main broiler and egg production systems in the UK, and hence to identify the main opportunities to reduce these impacts within each system. The broiler systems included in the study were 1) standard indoor, 2) free range and 3) organic production. According to the Defra (2007) statistics, the total broiler chicken populations in these systems in the UK were 101, 4.4 and 1.8 million, respectively. The egg production systems considered in this study were 1) conventional cage, 2) barn, 3) free range and 4) organic laying. Although the egg production in conventional cages has been banned by the EU and is not used in the UK anymore, it is still in use in some other European countries, and therefore the results for the cage laying system are also presented in this study. Results for enriched cages now used in the UK are expected to be broadly similar.

2. Methods

2.1. Systems approach and data

The general approach taken in the current study was with systems modelling of production. This included structural models of the industry, process models and simulation models that were unified in the systems approach so that changes in one area caused consistent interactions elsewhere. This approach was applied to both feed crop and animal production. The systems modelled in this study included crop production, non-crop nutrient production, feed processing, breeding, broiler production, pullet rearing, egg laying and manure and general waste management, as described by Williams et al., (2006) and Leinonen et al., (2012a;b). All modelled animal production systems included farm energy, feed and water use and gaseous emissions from housing.

The production systems in this study were considered to represent typical UK egg and broiler production (Table 1, Table 2) as described by Leinonen et al., (2012a;b). The farm energy consumption for heating, lighting, ventilation, feeding and incineration of dead birds was based on average data from typical farms as provided by the industry. Information about the type and amount of bedding was also obtained from the industry. Additional data, such as LCI of agricultural buildings and machinery, came from Williams et al., (2006).

Table 1. Typical production and feed intake figures for the different broiler production systems in UK as provided by the industry (Leinonen et al., 2012a)

	Standard	Free range	Organic
Final age, days	39	58	73
Average final weight, kg	1.95[a]	2.06	2.17
Feed intake, kg/bird	3.36	4.50	5.75
Mortality,%	3.5	4.7	4.1

[a] 25% of birds were removed by thinning at bodyweight 1.8 kg. The final weight of remaining birds was 2.0 kg.

Table 2. Typical production and feed intake figures for the different egg production systems in UK as provided by the industry (Leinonen et al., 2012b)

	Cage	Barn	Free range	Organic
Eggs collected/hen[a]	315	300	293	280
Average egg weight, g	62	63.5	63.5	63.5
Feed consumption, g/bird/day	115	125	130	131
Mortality,%	3.5	6	7	8

[a] based on the initial number of hens

The baseline diets representative of those used in the UK were constructed using information provided by the poultry industry. The broiler diets included four and the layer diets five separate phases, according to common practice. Separate diets were applied to 1) Standard broilers, 2) Free range broilers, 3) Organic broilers, 4) Cage, barn and free range layers, 5) Organic layers and 6) Broiler breeders.

2.2. The models

The structural model for broiler and egg systems calculated all of the inputs required to produce the functional unit (either 1000 kg of expected edible carcass weight in broilers or 1000 kg eggs), allowing for breeding overheads, mortalities and productivity levels. It also calculated the outputs, both useful (broilers, eggs and spent hens) and unwanted. Changes in the proportion of any activity resulted in changes to the proportions of others in order to keep producing the desired amount of output. Establishing how much of each activity was required was found by solving linear equations that described the relationships that linked the activities together.

A mechanistic animal growth, production and feed intake model, based on the principles presented by Emmans and Kyriazakis (2001) and Wellock et al., (2003), was used in this study in order to calculate the total consumption of each feed ingredient during the whole production cycle, and to calculate the amounts of main nutrients, nitrogen (N), phosphorus (P) and potassium (K) in manure produced by the birds during the production cycle. The model was calibrated to match the real production and feed intake data, provided by the UK poultry industry for different systems (Leinonen et al., 2012a;b), by adjusting the model parameters for growth rate, energy requirement for maintenance and egg production.

The model calculated the N, P and K contents of the manure according to the mass balance principle, i.e. the nutrients retained both in the animal body and eggs were subtracted from the total amount of nutrients obtained from the feed (including the additional nutrients obtained from foraging in free range and organic production). In addition to the nutrients excreted by the birds, nutrients in the spilled feed and uncollected eggs were added to the manure in the calculations. For the purpose of the study, it was assumed that all broiler, pullet, layer and breeder manure was transported for soil improvement, excluding the proportion that was excreted outside in the non-organic free range production systems.

A separate sub-model for arable production was used to quantify the environmental impacts of the main feed ingredients, with main features as in Williams et al., (2010). All major crops used for production of poultry feed were modelled. For the crops partly or wholly produced overseas (maize, soya, sunflower, palm oil) the production was modelled as closely as possible using local techniques, and transport burdens for importing were also included. The greenhouse gas emissions arising from land use change were taken into account according to the principles of the carbon footprinting method PAS 2050 (BSI, 2011).

A separate sub-model was used also used for manure in the nutrient cycle. In the model, the main nutrients that were applied to the soil in manure were accounted for as either crop products or as losses to the environment. The benefits of N, P and K remaining in soil after land application of manure were credited to poultry by offsetting the need to apply fertilisers to winter wheat as described by Sandars et al., (2003) and implemented by Williams et al., (2006). For organic systems, N was supplied from a dedicated legume used instead of synthetic N fertiliser, with rock P and K used instead of triple superphosphate and potassium chloride. In all cases, long term emissions and yield gains were accounted for to ensure a mass balance of N.

2.3. Environmental impacts

Emissions to the environment were aggregated into environmentally functional groups as follows. Global Warming Potential (GWP) was calculated using a timescale of 100 years. The main sources of GWP in poultry industry are carbon dioxide (CO_2) from fossil fuel, nitrous oxide (N_2O) and methane (CH_4). GWP was quantified as CO_2 equivalent: with a 100 year timescale 1 kg CH_4 and N_2O are equivalent to 25 and 298 kg CO_2 respectively (Foster et al., 2007).

Eutrophication Potential (EP) was calculated using the method of the Institute of Environmental Sciences (CML) at Leiden University (http://www.leidenuniv.nl/interfac/cml/ssp/index.html). The main sources are nitrate (NO_3^-) and phosphate (PO_4^{3-}) leaching to water and ammonia (NH_3) emissions to air. EP was quantified in terms of phosphate equivalents: 1 kg NO_3-N and NH_3-N are equivalent to 0.44 and 0.43 kg PO_4^{3-}, respectively.

Acidification Potential (AP) was also calculated using the method of the Institute of Environmental Sciences (CML) at Leiden University. The main source in poultry industry is ammonia emissions, together with sulphur dioxide (SO_2) from fossil fuel combustion. Ammonia contributes to AP despite being alkaline; when emitted into the atmosphere, it is oxidized to nitric acid. AP was quantified in terms of SO_2 equivalents: 1 kg NH_3-N is equivalent to 2.3 kg SO_2.

Primary Energy Use included all the energy needed for extraction and supply of energy carriers.

2.4. Uncertainty analysis

A Monte Carlo approach was applied to quantify the uncertainties of the modelled systems (Wiltshire et al., 2009, Leinonen et al., 2012a;b). The systems model, together with the animal production sub-model was run 5000 times, and during each run a value of each input variable was randomly selected from a predetermined distribution for this variable. The outcome of the analysis was the Coefficient of Variation, which was used to evaluate the statistical significance of the differences between the systems at 5% probability level.

The uncertainties in the input variables were divided into two groups, namely “alpha” and “beta” errors. Alpha errors were considered to vary between systems, and therefore were taken into account in statistical analyses of the differences between the systems. For example, variation between farms in production, feed intake and energy use figures were all considered to represent alpha errors. In contrast, beta errors were considered to be similar between the systems, and had no effect in the statistical comparison between the systems, e.g. the emission factor for N_2O from manure or conversion factor from electricity to primary energy. The errors in the emission factors were associated with errors in the models used to generate them, and therefore considered as beta errors (Wiltshire et al., 2009). Also, farms from which activity data were obtained were not restricted to particular climatic zones. However, it should be noted that the emissions themselves were affected for example by variation related to bird performance (e.g. N excretion), and therefore contained also alpha errors.

The uncertainties of the input variables were quantified and their distribution functions specified on the basis of the data from the industry, and they also included potential errors of the mechanistic models. The error distributions of the emission factors followed the IPCC (2006) guidelines.

3. Results

3.1. Broilers

The number of broiler birds required to produce the expected edible carcass weight of 1000 kg was higher in the standard indoor system than in the free range and organic systems because the finishing weight was lowest in the standard indoor system. The length of the production cycle was much higher in free range and organic systems than in the standard indoor system, thus the feed consumption per bird was also higher in these systems. This had a major effect on the trends in environmental burdens (Table 3).

Table 3. Global Warming Potential (GWP), Eutrophication Potential (EP), Acidification Potential (AP) and Primary Energy Use per 1000 kg of expected edible carcass weight in the main broiler production systems in the UK. The Coefficient of Variation based on the alpha errors is given in the parentheses.

	Standard	Free range	Organic
GWP (t CO_2e)	4.41 (8%)[a]	5.13 (8%)[ab]	5.66 (6%)[b]
EP (kg PO_4^{3-}e)	20.3 (8%)[a]	24.3 (7%)[a]	48.8 (6%)[b]
AP (kg SO_2e)	46.8 (8%)[a]	59.7 (7%)[b]	91.6 (6%)[c]
Primary Energy (GJ)	25.4 (8%)[a]	25.7 (7%)[a]	40.3 (6%)[b]

[a,b,c] Different superscript indicates statistically significant difference ($P < 0.05$) between the systems.

Feed caused higher overall environmental impacts than any other materials involved in broiler production, for example 71 - 72% of the total GWP and 65 - 81% of the Primary Energy Use of the system. The GWP was affected by relatively high CO_2 emissions from the production and transport of some feed ingredients (e.g. non-organic soya, palm oil, fish meal and pure amino acids) in the standard and non-organic free range broiler diets. On the other hand, organic feed had generally much higher impact than the non-organic feed in other impact categories, especially EP. Although the emissions per land area are sometimes lower in organic crop production compared to non-organic, the yields are generally much lower as fertility building and cover crops are required, and this makes the emissions higher per unit of the product. Leaching after cultivating a clover ley was a major contributor to eutrophication potential.

Emissions from manure were the main component of AP in broiler production and had also a relatively high contribution to EP. This was mainly a result of ammonia emissions, which contributed to both these potentials, together with nitrate leaching (affecting only EP). The AP from manure was especially high in the organic system.

3.2. Eggs

The production of 1000 kg eggs required 51.2 laying birds in the cage system, 52.6 in the barn system, 53.8 in the free range system and 56.3 in the organic system. This general trend in productivity also affected other aspects of derived activity data, such as feed consumption. Furthermore, the average feed consumption per bird was also higher in the alternative systems than in the cage system. Much of the explanation of the trends in environmental burdens that followed resulted from these differences in the efficiency of the systems (Table 4).

As in the broiler systems, feed was the biggest component of GWP in egg production (contributing 64 - 72% to the overall GWP and 54 - 75% to the overall Primary Energy Use of the systems). Compared with broiler production, the farm electricity use had a higher relative contribution to GWP and Primary Energy Use, especially in barn egg production. Again, manure was a major source of both EP and AP, which were especially high in the organic egg production system.

Table 4. Global Warming Potential (GWP), Eutrophication Potential (EP), Acidification Potential (AP) and Primary Energy Use per 1000 kg of eggs in the main egg production systems in the UK. The Coefficient of Variation based on the alpha errors is given in the parentheses.

	Cage	Barn	Free range	Organic
GWP (t CO_2e)	2.92 (5%)[a]	3.45 (5%)[b]	3.38 (6%)[ab]	3.42 (6%)[b]
EP (kg PO_4^{3-}e)	18.5 (3%)[a]	20.3 (4%)[b]	22.0 (5%)[b]	37.6 (5%)[c]
AP (kg SO_2e)	53.1 (2%)[a]	59.4 (4%)[b]	64.1 (5%)[b]	91.6 (5%)[c]
Primary Energy (GJ)	16.9 (6%)[a]	22.2 (5%)[b]	18.8 (6%)[a]	26.4 (6%)[c]

[a,b,c] Different superscript indicates statistically significant difference ($P < 0.05$) between the systems.

4. Discussion

The results of this study show that the environmental impacts of both broiler and egg production are largely related to the efficiency of resource use of each system. In broilers, the standard indoor system had shorter production cycle compared to the alternative systems, and therefore also lowest feed consumption and manure production per functional unit. Also in egg production, the alternative systems were generally less efficient than the cage system, and therefore had also higher environmental impacts.

Feed production and processing was the main component of the global warming potential both in broiler and egg production systems. This was partly affected by the fact that some ingredients, most notably soya and palm oil, were considered to be partly produced on land that has been only recently converted from natural vegetation to agricultural use in South America and South Asia. When calculating the land use change effect on GWP, this study applied the guidelines of the carbon footprinting method PAS2050 (BSI, 2011). However, there is not a full international agreement on the method of how to account for land use changes in LCA, and this has potentially a very big effect on the estimate of the environment impact of broiler and layer feed and poultry production in general.

The environmental impacts of broiler and egg systems in different countries have been quantified in some earlier studies (e.g. Williams et al., 2006; Mollenhorst et al., 2006; Pelletier, 2008; Katajajuuri, 2008; Boggia et al., 2010, Dekker et al., 2011). However, systematic comparisons between different animal production systems are still quite rare. In general, comparison between different studies or different systems in a single study is not feasible if the range of uncertainty in the results is not available. Therefore it is quite surprising that uncertainly analysis has not been widely applied when quantifying the environmental impacts of agricultural products. The method applied in the present study allows to separate different error categories in the input data, and provides a tool for evaluating the differences between production systems in a consistent way.

In addition to the general comparison between different broiler and egg production systems, the modelling framework applied in this study provides an opportunity to carry out detailed farm level assessments on how to reduce the environmental impacts of production. Since the analysis is largely based on functional relationships built in the animal and crop production sub-models, it is possible to examine holistic effects of possible changes in the system. For example, changes in consumption and composition of feed have effects both on the impacts occurring during the crop production and feed processing, and also on the subsequent emissions from poultry manure during housing, storing and field application. Similarly, the differences in the growth rate of broilers affects the amount of feed consumed per functional unit, the amount of manure produced and the amount of energy and buildings needed, among other things.

Future options in reducing the environmental impacts of animal production include breeding programmes for better environmental performance. The current results indicate that improving feed efficiency, including not only the quantity but also composition and nutrient content of the consumed feed has potential to reduce the environmental impacts. The modelling framework with functional relationships applied in the present study will allow detailed and realistic tools for quantifying the environmental consequences of future genetic progress in animals. Further options for reducing the high environmental impacts from animal feed include the use of alternative, more environmentally friendly ingredients. For example, it can be expected that reducing the inclusion of imported soya, partly originated from recently converted agricultural land, and replacing it using locally grown protein sources may reduce the high greenhouse gas emissions related to both land use changes and long transport distances.

5. Conclusion

There were relatively large differences in many categories of the environmental impacts between different UK broiler and egg production systems and generally these reflected the differences in the efficiency in production, feed consumption (and related production of manure) and material and energy use.

The methodology used in the current study with functional relationships between different activities related to animal production and mechanistic representation of biological processes provides a realistic tool for quantification of environmental impacts of various agricultural systems. This includes the quantifications of the overall uncertainties of the model outputs, which allows systematic comparison between different production systems.

6. Acknowledgments

This research was financially supported by Aviagen Ltd, DSM Nutritional Products Ltd, Harbro Ltd, Moy Park Ltd, National Farmers' Union, Noble Foods Ltd, O'Kane Poultry Ltd, The Soil Association Ltd and Waitrose Ltd with match funding from Defra, through the Sustainable Livestock Production LINK program, DARDNI and the Scottish Government.

7. References

BSI. 2011. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. PAS 2050:2011. http://www.bsigroup.com/Standards-and-Publications/How-we-can-help-you/Professional-Standards-Service/PAS-2050/

Defra. 2007. GB Meat Chicken Population - The National Flock. Version 1. RADAR Team, Food and Farming Group, Department for the Environment, Food and Rural Affairs (Defra). http://www.defra.gov.uk/foodfarm/farmanimal/diseases/vetsurveillance/reports/documents/rp6309.pdf

Defra. 2009. Agricultural statistics http://www.defra.gov.uk/statistics/files/defra-stats-foodfarm-crosscutting-auk-2011-chapter05-commodities-livestock-120709.xls

Dekker, S.E.M., de Boer, I.J.M., Vermeij, I., Aarnink, A.J.A., Groot Koerkamp, P.W.G., 2011. Ecological and economic evaluation of Dutch egg production systems. Livestock Sci. 139, 109–121.

Emmans, G.C., Kyriazakis, I., 2001. Consequences of genetic change in farm animals on food intake and feeding behaviour. Proc. Nutr. Soc. 60, 115-125.

FAO, 2011. Food and Agriculture Organization of the United Nations. http://faostat.fao.org

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. Changes in atmospheric constituents and in radiative forcing. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories. (2006 Guidelines) http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html

Katajajuuri, J.M., 2008. Experiences and improvement possibilities — LCA case study of broiler chicken production. Proc. 6th Int. Conf. Life Cycle Assessment in the Agri-Food Sector. Zurich, Switzerland.

Leinonen, I., Williams, A.G., Wiseman, J., Guy, J., Kyriazakis, I., 2012a. Predicting the environmental impacts of chicken systems in the UK through a Life Cycle Assessment: broiler production systems. Poult. Sci. 91, 8-25.

Leinonen, I., Williams, A.G., Wiseman, J., Guy, J., Kyriazakis, I., 2012b. Predicting the environmental impacts of chicken systems in the UK through a Life Cycle Assessment: egg production systems. Poult. Sci. 91, 26-40.

Mollenhorst, H., Berentsen, P.B.M., De Boer I.J.M., 2006. On-farm quantification of sustainability indicators: an application to egg production systems. Br. Poult. Sci. 47, 405-417.

Pelletier, N., 2008. Environmental performance in the US broiler poultry sector: Life cycle energy use and greenhouse gas, ozone depleting, acidifying and eutrophying emissions. Agric. Syst. 98, 67-73.

Sandars, D.L., Audsley, E., Canete, C., Cumby, T.R., Scotford, I.M., Williams, A.G., 2003. Environmental benefits of livestock manure management practices and technology by Life Cycle Assessment. Biosyst. Eng. 84:267-281.

Wellock, I.J., Emmans, G.C., Kyriazakis, I., 2003. Modelling the effects of thermal environment and dietary composition on pig performance: model logic and concepts. Anim. Sci. 77, 255-266.

Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. http://preview.tinyurl.com/Cranfield-IS0205

Williams, A.G., Audsley, E., Sandars, D.L., 2010. Environmental burdens of producing bread wheat, oilseed rape and potatoes in England and Wales using simulation and system modelling. The Int. J. Life Cycle Assess. 15, 855-868.

Wiltshire, J., Tucker, G., Williams, A.G., Foster, C., Wynn, S., Thorn, R., Chadwick, D., 2009. Supplementary Technical Report to "Scenario building to test and inform the development of a BSI method for assessing GHG emissions from food". Final report to Defra on research project FO0404, London, UK. Accessed Oct. 18, 2011. http://preview.tinyurl.com/Fin-Rpt-FO0404

Influence of scenario uncertainty in agricultural inputs on LCA results for agricultural production systems

Kiyotada Hayashi[1,*], Naoki Makino[2], Koichi Shobatake[2], Shingo Hokazono[1]

[1] National Agriculture and Food Research Organisation
[2] TCO2 Co. Ltd.
[*] Corresponding author. E-mail: *hayashi@affrc.go.jp*

ABSTRACT

Practical applications of life cycle assessment (LCA) to agricultural production systems necessitate articulating uncertainties caused by scenario indeterminacy, because practitioners do not have sufficient knowledge about agricultural input production processes. However, current understanding about scenario uncertainties is still limited on account of insufficient knowledge. Here, we propose a method to quantify scenario uncertainty in agricultural inputs and to assess the uncertainty in comparative LCA of agricultural production systems. We formulate mathematical expressions about uncertainty intervals due to scenario indeterminacy and derive uncertainty intervals for conventional, environmentally friendly, and organic rice production systems in Japan. Scenario uncertainty in chemical fertiliser production is analysed as an example. The results indicate that uncertainty intervals are useful in understanding the stability of results. The methodology proposed in this study can be further developed as a technique to deal with uncertainty and instability in LCA of agricultural production systems.

Keywords: scenario uncertainty, agricultural inputs, adaptation, comparative LCA, inventory analysis

1. Introduction

Uncertainty analysis using the pedigree matrix for data quality together with Monte Carlo simulations is a common practice in life cycle assessment (LCA), and several LCA software products provide simulation functions. Because of certain special characteristics of agriculture, uncertainties in parameters such as crop yield and direct field emissions are integrated into the LCA for agriculture (Basset-Mens et al., 2006). In addition, uncertainties attributed to a wide variety of management practices and uncertainties in the relationship between management practices and environmental impacts have been estimated using statistical resampling (nonparametric bootstrapping) (Hayashi, 2011).

Although these studies mainly clarified uncertainties in models and parameters used to conduct LCA, practical applications of LCA to agricultural production systems necessitate articulating uncertainties caused by scenario indeterminacy, because practitioners do not have sufficient knowledge of the details in agricultural input production processes (background processes of agricultural production). For example, farmers in general do not know what kind of technologies are used for making chemical fertilisers and where fertiliser factories are located. In other words, decision makers or analysts face decision problems under insufficient knowledge.

However, current understanding about scenario uncertainties is still limited in LCA of agricultural production systems as a result of insufficient knowledge. Therefore, this study establishes a method to quantify scenario uncertainty in agricultural inputs and assesses the influence of this uncertainty on a comparative LCA of agricultural production systems.

2. Methods

2.1. Classification of uncertainties

We classify uncertainty using a tri-partition for the uncertainty typology, namely parameter, model, and scenario uncertainty. Parameter uncertainty reflects our incomplete knowledge about the true value of a parameter (Huijbregts et al., 2003) and is related to inventory data and characterisation and weighting factors. Common sources of parameter uncertainty are imprecise measurements, incomplete or outdated measurements, and no measurements (lack of data) (Huijbregts, 1998). Model uncertainty concerns assumptions and simplifications that lead to uncertainty about the validity of the model's predictions for a real world situation (Huijbregts et al., 2003). An important example of model uncertainty is the loss of spatial and temporal characteristics in inventory analysis. Scenario uncertainty was originally termed 'uncertainty due to choices' (Huijbregts, 1998) and 'decision rule uncertainty' (Hertwich et al., 2000), because it refers to uncertainty caused by normative choices on functional units and system boundaries in goal and scope definition, allocation in inventory analysis, and the number of impact categories and definitions in impact assessment (Huijbregts et al., 2003). We extend the scope of scenario construction to scenario indeterminacy, which we encounter in practical situations.

2.2. Uncertainty intervals due to scenario indeterminacy

An interval for an environmental impact of an agricultural production system, *p*, which expresses uncertainty due to scenario indeterminacy of an input, *f*, is written as $[e_f^L(p), e_f^R(p)]$. Here, the lower bound (denoted by superscript *L*) and the upper bound (denoted by superscript *R*) are defined as follows:

$$e_f^L(p) = f^L(p) + o(p) + d(p), \quad \text{Eq. 1}$$

$$e_f^R(p) = f^R(p) + o(p) + d(p), \quad \text{Eq. 2}$$

where $e_f^{L \vee R}(p)$ is the lower or upper bound of an environmental impact of *p* with respect to *f*, $f^{L \vee R}(p)$ is the lower and upper bounds of an environmental impact of *p* caused by the focused input *f* (e.g. fertilisers), *o*(*p*) is an environmental impact of *p* caused by the other inputs and *d*(*p*) is an environmental impact of *p* caused by direct emissions.

Although equations 1 and 2 express an interval with respect to an attribute of an input, we will cope with multiple attributes of an input as illustrated in Table 1, which illustrates an example with three attributes. It is supposed that there are *t* alternative scenarios in production technology, *l* alternative scenarios in production location, and *d* alternative scenarios in description types. The lower and upper bounds for production technology, for example, are determined with respect to *i* on the condition that *j* and *k* are set to the average (default) scenarios ($\bar{j}$ and $\bar{k}$).

Table 1. Derivation of uncertainty interval due to scenario indeterminacy: an example

Attribute	Production technology	Production location	Description type[a]
Alternative scenario	$i \in T = \{1,2,\ldots,t\}$	$j \in L = \{1,2,\ldots,l\}$	$k \in D = \{1,2,\ldots,d\}$
$f^L(p)$	$\min_{i \in T} f_{i\bar{j}\bar{k}}$	$\min_{j \in L} f_{\bar{i}j\bar{k}}$	$\min_{k \in D} f_{\bar{i}\bar{j}k}$
$f^R(p)$	$\max_{i \in T} f_{i\bar{j}\bar{k}}$	$\max_{j \in L} f_{\bar{i}j\bar{k}}$	$\max_{k \in D} f_{\bar{i}\bar{j}k}$

[a] Information used to connect the foreground system and the background system

2.3. Scenario indeterminacy in practical situations

We analyse the following three situations in which complete scenarios are difficult to construct. In the first situation, the information on production technology is unavailable. For example, the technical details about the production processes for fertilisers and pesticides are not publicised because they are the key to successful business management. Although the full description of the production processes is difficult to obtain, uncertainty due to unavailability of technological details is partly assessed by comparing the cases with and without adaptation in life cycle inventory (LCI) data for agricultural inputs (Ossés de Eicker et al., 2010).

The second situation concerns uncertainty due to the lack of knowledge of the production location of agricultural inputs. A typical example is the information on the transportation of domestic and imported agricultural inputs, such as fertilisers and pesticides. Uncertainty intervals are calculated by comparing different scenarios, such as cases in which all chemical fertilisers are imported (import scenario), and those in which all are made in Japan (domestic scenario). The average scenario was also estimated using national fertiliser statistics.

The third situation we analyse is the lack of specifications for agricultural inputs. In this situation, the following verbal expressions (description types) are used for the assessment: '40,000 JPY fertilisers were applied per ha per year' and '54 N-kg nitrogen fertilisers were applied per ha per year'. Uncertainty intervals are obtained by comparing the scenario in which only the total sum of chemical fertiliser costs is known and the scenario in which detailed information about the quantities of each chemical fertiliser is available.

2.4. Comparison at the level of agricultural production systems

The quantification of uncertainty is conducted at the level of agricultural production systems. Conventional, environmentally friendly, and organic rice cultivation in the central part of Japan are compared; that is, $[e_f^L(p_C), e_f^R(p_C)]$, $[e_f^L(p_E), e_f^R(p_E)]$, and $[e_f^L(p_O), e_f^R(p_O)]$ are compared to each other, where P_C is the conventional production system, P_E is the environmentally friendly production system, and P_O is the organic production system.

2.5. System description and impact assessment

An outline of each production system is illustrated in Table 2. The conventional and environmentally friendly production systems are based on the business activities of an agricultural production cooperation in 2007 and 2008. Details of the agricultural practices in the latter system are defined by the prefectural government. The organic production system is based on a field trial conducted by the prefectural extension service staff on the cooperation's fields in 2007 and 2008. Although there are several kinds of organic rice production systems in Japan, the organic production system in this study uses rice bran and weeding machinery as weed control practices.

Table 2. Outline of conventional, environmentally friendly, and organic rice production systems

	Conventional[a]	**Environmentally Friendly**[a]	**Organic**[b]
Seed disinfection method	Fungicide application	Hot-water treatment	Hot-water treatment
Density of transplanting (Number of plates for seedling raising)	28	20	20
Weeding method	Herbicide application	Herbicide application	Application of rice bran Use of weeding machinery
Disease and insect damage control method	Fungicide application Insecticide application		
Type of fertilisers	Chemical fertilisers	Chemical fertilisers Organic fertilisers	Organic fertilisers
Yield (t/ha)	5.96	5.58	4.97

[a] Based on activities of an agricultural production cooperation ('H Farm').
[b] Based on a field trial conducted by the prefectural extension service staff on the cooperation's fields

The farm gate (cradle to gate) forms the system boundary and 1 kg of brown rice was used as the functional unit. Global warming potential over 100 years is tentatively applied to the comparisons. This paper analyses the influence of scenario uncertainty, mainly in fertilisers, on LCA results as a first step to the overall assessment.

2.6. Life cycle inventory

JALCA (Japan Agricultural Life Cycle Assessment) Database (Hayashi et al., 2012), which is the updated version of the NARO (National Agriculture and Food Research Organisation) LCI database (Hayashi et al., 2010), was used to construct the LCI data for the production systems. SimaPro 7.3 was used for data management and calculation. The details are as follows:

(1) Agricultural input production processes in ecoinvent 2.2 were adapted to Japan using the JLCA-LCA database.
(2) Six-step transportation processes were constructed to generate transportation scenarios for chemical fertiliser production, as illustrated in Fig. 1. Three scenarios for chemical fertilisers (the domestic production scenario, the import scenario, and the national average scenario, which is the weighted average based on statistics) were constructed.
(3) Emission factors based on Input-Output (IO) tables (Nansai et al., 2002) were introduced to use the monetary verbal expressions.

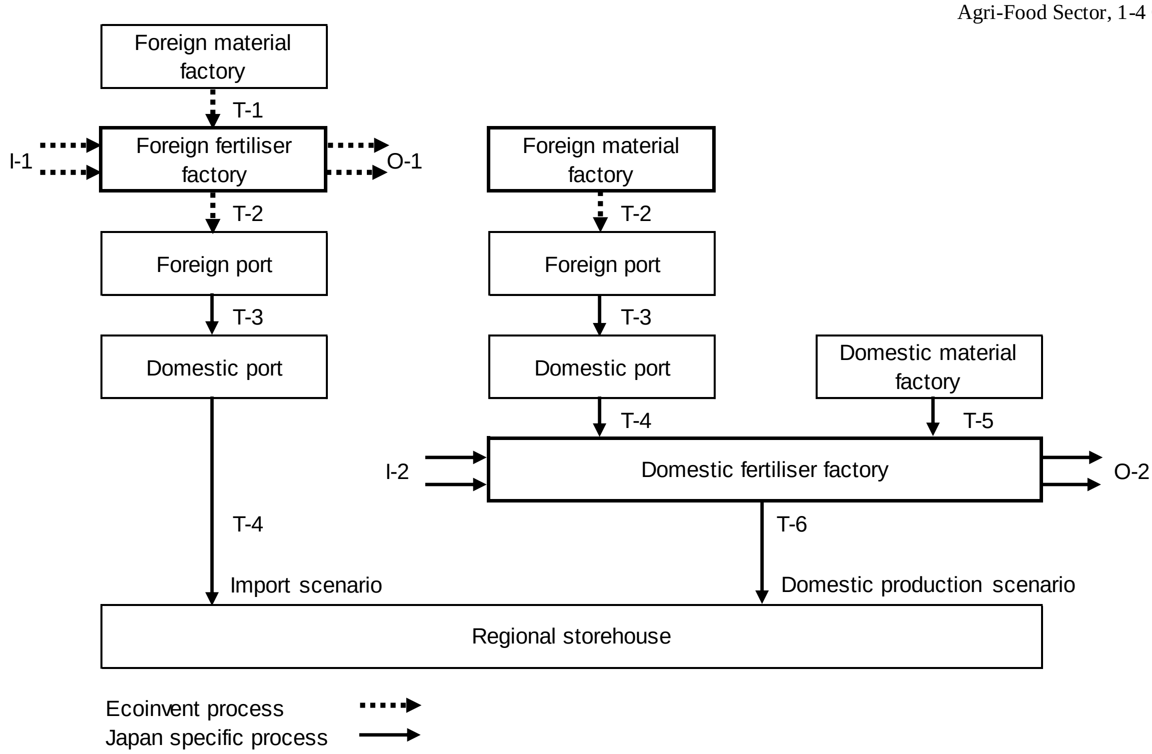


Figure 1. Generation of transportation scenarios in the case of chemical fertiliser production. T: transport stage, I: input, and O: output.

3. Results

3.1. Production technology

There were no significant differences between the case with the adapted LCI data for chemical fertilisers and that with the original fertiliser data from ecoinvent. Similar results were obtained in the case of pesticides.

3.2. Production location

Lack of information on transportation scenarios caused uncertainty (Fig. 2). However, the relative superiority among conventional, environmentally friendly, and organic rice cultivation remained unchanged.

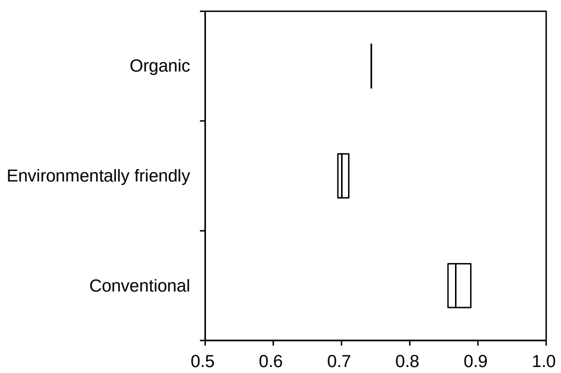


Figure 2. Greenhouse gas (GHG) emissions of the three production systems with uncertainty due to indeterminacy of transportation scenarios in the case of chemical fertiliser production (kg CO_2 eq./kg). From the left (in the boxes): the domestic production scenario, the national average scenario, and the import scenario.

3.3. Verbal expressions as scenarios

Uncertainty due to the lack of the specifications was larger than that due to the scenario indeterminacy described above (see Fig. 3). The relative superiority between environmentally friendly and organic cultivation was indeterminable.

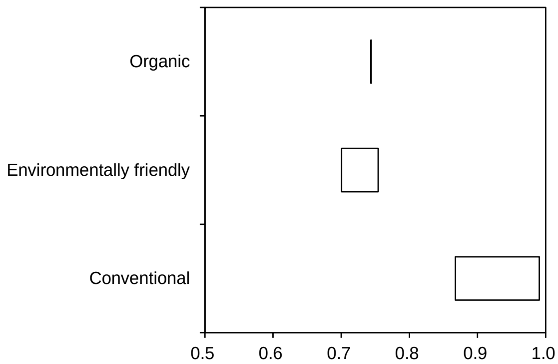


Figure 3. Greenhouse gas (GHG) emissions of the three production systems with uncertainty due to the lack of specifications in the case of chemical fertiliser production (kg CO_2 eq./kg). In the boxes, the upper bounds are GHG emissions based on input-output tables, and the lower bounds correspond to the national average scenario. It is assumed that values using emission factors based on input-output tables tend to be overestimated.

4. Discussion

Assessing the influence of scenario uncertainty in all agricultural inputs is important, although we paid special attention to chemical fertilisers. In other words, it is necessary to conduct uncertainty analysis for organic fertilisers, pesticides, machinery, and building. If we assess the influence of organic fertilisers, for example, uncertainty intervals for the organic and environmentally friendly production systems may become longer. Since direct GHG emissions from organic fertiliser production processes are estimated to be a significant proportion of the total GHG emissions, uncertainty analysis of the composting process is important when making decisions.

Although we restricted our attention to global warming in this study, a variety of impact categories should be used for analysing the influence of scenario uncertainty in various agricultural inputs. Ammonia emissions from composting processes are, for example, not negligible, and thus eutrophication and acidification are important.

With regard to the techniques for uncertainty analysis, we discuss three issues. First, we have to cope with unavailability of data on detailed production processes. Although we introduced the method of adaptation, which is equivalent to generating a child from parents in structured computer languages, further formalised techniques would be necessary for modelling production technologies. Second, problems with unavailability of detailed information are also applicable to transportation. In addition to the use of statistics, the introduction of case-based reasoning would be important. Third, we have to pay attention to the difference in the foreground-background connection between an amount and a mass. That is, the third result illustrates the reliability of simple estimation using emission coefficients based on IO tables. In addition, further development of scenario uncertainty analysis by establishing a method to convert verbal expressions to numerical information is necessary.

5. Conclusion

The primary purpose of quantifying scenario uncertainty is to understand the stability of the results in comparative LCA. The interval information is useful in intuitively understanding the degree of uncertainty. Furthermore, the results of simple comparisons between alternative productions systems may be revised by using interval comparisons. Thus, interval judgments will be a key to making results more robust. The methodology proposed in this study can be further developed as a technique to deal with uncertainty and instability in LCA of agricultural production systems.

6. References

Basset-Mens, C., van der Werf, H.M.G., Durand, P., Leterme, P., 2006. Implications of uncertainty and variability in the life cycle assessment of pig production systems. The International Journal of Life Cycle Assessment 11(5), 298-304.

Hayashi, K., Uchida, S., Hokazono, S., Sato, M., 2010. Modelling life cycle inventories for crop production in Japan: development of the NARO LCI database. In: Natarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.), LCAFOOD 2010: VII International Conference on Life Cycle Assessment in the Agri-food Sector, pp. 455-460.

Hayashi, K., 2011. Assessing management influence on environmental impacts under uncertainty: a case study of paddy rice production in Japan. in: Finkbeiner, M., (Ed.), Towards Life Cycle Sustainability Management. Springer, pp. 331-340.

Hayashi, K., Shobatake, K., Makino, N., Hokazono, S., 2012. Development of a life cycle inventory database for agricultural production systems in Japan: the JALCA Database. The 10th International Conference on EcoBalance, Yokohama.

Hertwich, E.G., McKone, T.E., Pease, W.S., 2000. A systematic uncertainty analysis of an evaluative fate and exposure model. Risk Analysis 20(4), 439-454.

Huijbregts, M., 1998. Application of uncertainty and variability in LCA. Part I: A general framework for the analysis of uncertainty and variability in life cycle assessment. The International Journal of Life Cycle Assessment 3(5), 273-280.

Huijbregts, M.A.J., Gilijamse, W., Ragas, A.M.J., Reijnders, L., 2003. Evaluating uncertainty in environmental life-cycle assessment: a case study comparing two insulation options for a Dutch one-family dwelling. Environmental Science & Technology 37(11), 2600-2608.

Nansai, K., Moriguchi, Y., Tohno, S., 2002. Embodied Energy and Emission Intensity Data for Japan Using Input-Output Tables (3EID). National Institute for Environmental Studies (NIES), Tsukuba.

Ossés de Eicker, M., Hischier, R., Kulay, L.A., Lehmann, M., Zah, R., Hurni, H., 2010. The applicability of non-local LCI data for LCA. Environmental Impact Assessment Review 30(3), 192-199.

Using systems-based LCA to investigate the environmental and economic impacts and benefits of the livestock sector in the UK

Julia Chatterton*, Eric Audsley, Anil Graves, Joe Morris, Adrian Williams

Centre for Environmental Risks and Futures, School of Applied Sciences, Cranfield University, Bedford, MK43 0AL, UK.
* Corresponding author. E-mail: *j.chatterton@cranfield.ac.uk*

ABSTRACT

The livestock industry is a significant component of the agricultural and rural sectors in the UK. Grassland for livestock accounts for almost half of the terrestrial surface of the UK and almost two-thirds of its managed agricultural land. It therefore accounts for a major proportion of rural employment and income and provides many landscape and biodiversity benefits. Taking an ecosystems-services-framework approach, an integrated livestock-ecosystems linear programming model was developed to assess economic and environmental impacts of the livestock sector. This combined life cycle assessment systems analysis with economic and valuation data and enabled plausible future scenarios to be assessed in terms of provisioning, regulating and cultural services. Analysis showed the main benefit coming from provisioning services and also the significant role the sector plays in providing cultural services and the trade-offs between these and the cost of regulating services with respect to emissions to air and water.

Keywords: livestock, ecosystem services, economics, landscape value, trade-offs

1. Introduction

The livestock industry is a significant component of the agricultural and rural sectors in the UK. Grassland for livestock accounts for almost half of the terrestrial surface of the UK and almost two thirds of its managed agricultural land. It therefore accounts for a major proportion of rural employment and income, and provides many landscape and biodiversity benefits. In 2009, the value of livestock production in the UK was estimated to be £10,833 million (Defra, 2009), equivalent to 56% of total agricultural value. However, the livestock sector is also associated with large proportions of the environmental burdens from agriculture, for example, about 8% of UK greenhouse gas (GHG) emissions and 68% of the UK agricultural ammonia emissions. Although livestock is a key component of the rural landscape and economy in the UK, particularly in northern and western regions that have comparative advantage in grassland farming, there is considerable debate about the combined economic, social and environmental impact of livestock production in the UK. The debate is further complicated by changes in market demand, whether associated with reduction in the demand for red meat, as part of healthier diets, or a strengthening of global demand for dairy and meat products as incomes rise in developing economies.

Many highly valued and historic features of the rural landscape in the UK are a result of livestock farming, notably the patchwork of fields bounded by hedgerows and stone walls that are part of distinctive landscape characteristics. Simultaneously, many grassland systems, especially those following traditional methods, are associated with high levels of biodiversity, which become more valuable with reductions in biodiversity in more intensively farmed areas. Furthermore, livestock and grassland areas in both lowlands and uplands are closely integrated with rural tourism and recreation, where enjoyment of the countryside provides a range of social and economic benefits, relieving pressures associated with increasingly urban lifestyles.

In this context, the aim of the project was to determine the economic, social and environmental performance of livestock production in the UK and also to explore the implications of alternative future scenarios associated with possible changes in the demand for livestock products or the consequences for the livestock sector of giving different priorities to economic, social or environmental objectives. It also sought to identify likely challenges to achieving a profitable and environmentally sustainable livestock industry and the new knowledge and skills that might be required.

In the last decade, the ecosystems framework has emerged as a means of explicitly linking natural capital with social welfare. In this, natural capital supports a number of interrelated ecosystem services (provisioning, regulating, cultural and supporting services) which produce a variety of goods or benefits that have value for humans (MEA, 2005, Defra, 2007 and UNEP-UKNEA, 2010). The positive impacts of the livestock sector are mainly linked to the "provisioning" of food (and the broader benefits associated with employment and linkages to related industries) and "cultural" benefits in terms of the aesthetic pleasure obtained from grazed landscapes, features such as hedgerows and stone walls, and biodiversity. Negative impacts are largely associated with the loss of "regulating" services, including emissions of greenhouse gases to the atmosphere and emissions of contaminants to water, including sediment and transport of bound pollutants such as phosphorus, pesticides, heavy metals, and pathogenic microorganisms. This became the framework of analysis for an integrated livestock-ecosystems model.

2. Methods

A review of factors driving change in the livestock sector was undertaken, including key agricultural and environmental policies. The opinions of a range of industry representatives were canvassed regarding views on the role of livestock in the rural economy and its relationship with the environment. The relationship between livestock farming systems and society was then explored using an ecosystems framework, namely the affect on provisioning (e.g. food production), regulating (e.g. GHG emissions) and cultural (e.g. landscape) services. A variety of scenarios were then developed to reflect actual and potential demands from the livestock sector, given current and future drivers.

2.1 Scenarios for modelling

Following a review of scenarios used in previous projects and the understanding developed in a review of science and policy, a set of plausible future scenarios was developed to explore how the UK Government Department for Environment, Food and Rural Affairs (DEFRA) objectives for the livestock sector could be met, namely a “profitable and competitive domestic industry which enhances the biodiversity and rural landscape of England while minimising its impact on climate change, soil, water, and air quality”. These were:

1. Business As Usual (BAU): a baseline business as usual scenario, examining the net balance between the Provisioning, Regulating, and Cultural impacts of the livestock sector and determining how individual sub-sectors contribute to this.

And also a series of optimising scenarios to determine the effect of:

2. Maximising employment generation associated with the livestock sector
3. Minimising production costs from the livestock sector
4. Reducing GHG emissions by 25%
5. Shifting from red (beef and lamb) to white meat (pork and poultry) - with red meat assumed to be provided only from dairy beef, as a by-product of milk production
6. Reducing production of each livestock sector by 25%, balanced as far as possible by plant commodities

2.2 Data inputs and model development

Data for the valuation of Provisioning, Regulating and Cultural services were developed from a variety of sources and adjusted, where needed, to 2009 values with HM Treasury GDP deflators (2011). Whilst data on provisioning services were relatively easy to find, data on the impact of livestock systems on the value of regulating and cultural services were especially difficult to develop. Valuation data were developed from a variety of sources, including the Report on the Environmental Accounts for Agriculture (Jacobs, 2008).

2.3 Development of an integrated livestock-ecosystems model

An integrated livestock-ecosystems linear programming model was developed to assess the economic and environmental impacts of the livestock sector, using an ecosystem services framework. For this, the Cranfield Life Cycle Assessment Model (Williams et al., 2006, 2007) was combined with a grassland productivity model and a soil erosion model to assess environmental consequences of the livestock sector. A model was also developed to calculate soil erosion for each 5x5km grid square in England and Wales using the Morgan-Morgan-Finney model (Morgan et al., 1984). Soil erosion was calculated per unit area for each slope angle for the different land uses and slope proportions, then allocated between the different systems, after removing non-productive land. From this, average erosion values (t ha^{-1}) were derived for arable, dairy, beef and sheep, and then split among lowland (<100m), upland (≤100m; <300m) and hill (≤300m). This was repeated for wheat up to a slope angle of 15.5° to inform those scenarios in which the extent to which arable systems could replace livestock systems was examined.

The outputs of the LCA model were linked within the linear programming framework (Table 9) to the valuation and linked industry data.

Table 9. Illustrative description of the livestock ecosystem services linear programming model. Columns represent levels of activities and rows represent constraints on those activities. For illustration these have been collapsed into a single description (e.g., sheep systems are actually 63 columns, with activities such as hill, upland and lowland ewes, and organic flocks). Similarly rows have been collapsed into 'Emissions', which includes rows for individual emissions such as greenhouse gases, nitrate leaching and pesticides.

	Sheep systems	Pig systems	Beef systems	Chicken systems	Egg systems	Dairy systems	Arable-well-suited	Arable-moderate	Arable-marginal	Unsuitable	Emissions	Cultural	Rural labour		
Objective (services)														=	P+R+C
Provisioning	+	+	+	+	+	+	+	+	+				+	=	P
Regulating											+			=	R
Cultural	+	+	+	+	+	+	+	+	+	+		-ve		=	C
Constraint															
Production	1	1	1	1	1	1								=	Demand
Arable land	+	+	+	+	+	+	1							=	L
Grassland	+	+	+	+	+	+	1	1	1	1				=	L
Hill land	+		+							1				=	L
Emissions	+	+	+	+	+	+	+	+	+		-1			=	0
Labour	+	+	+	+	+	+	+	+	+				-1	=	0
Intra-system	+	+	+	+	+	+								=	0
Inter-system			1			-ve								=	0
Land types	+	+	+	+	+	+								<	L

Notes: + denotes positive coefficients, -ve denotes negative coefficients, +/-1 denotes unit coefficients, L denotes limits/constraints on land and soil types

The objective function to be maximised is the sum of the various ecosystem services, which have been converted to a common monetary (£) valuation system (P+R+C). This comprises positive monetary values for livestock production and arable production, negative values for emissions, and positive values for cultural services and employment generation (columns, Table 1). Cultural services include landscape, biodiversity and recreation benefits services considered jointly in terms of a value of willingness to pay that varies by land use. In the main analyses, we consider employment as a benefit and value it as the minimum wage. The relations between activities and constraints (columns and rows, respectively, Table 1) are defined by an array of technical coefficients, showing, for example, farm labour requirements of a tonne of beef or nitrate emissions generated in the production of a tonne of meat on a particular type of land.

The systems-based LCA approach also enabled the implications of a range of alternative future scenarios to be explored, including a 25% reduction in livestock production balanced by plant commodities, a shift from red to white meat and arable substitution of the livestock sector. The model maximises the weighted net benefit of the ecosystems services generated from each of the livestock sectors.

3. Results

Estimates of the value of ecosystem services, classified into provisioning, regulating and cultural services were first obtained for the BAU scenario. This involved an allocation of land, both grassland and the arable area required to provide non-grass feeds, to meet current demand for livestock products. The proportional distribution of grassland and arable land for each 5x5 km square in England and Wales was estimated using spatially interpolated 2004 Agricultural Census data (Edine, 2010). These confirm the greater incidence of grassland and livestock production in the northern and western regions, associated with sheep, dairy, and beef systems, although it is worth noting the significant production of pigs and poultry in the eastern part of England. Grasslands were divided into lowland (<100m), upland (≤100m; <300m) and hill (≤300m), and further sub-divided by site class (Magic, 2011), which depended on rainfall, soil texture, temperature, and wetness.

The BAU results for the UK showed that the main benefit of livestock systems was from the provisioning service in terms of production of meat, milk and eggs (Table 10). The total product value was estimated to be £8268 million. This benefit (excluding labour) was £5337 million with the greatest benefits estimated to be provided by dairy and dairy beef, followed by chicken. Whilst these direct benefits of the provisioning service are attributable to the livestock sector, there exist a variety of linked benefits, which although typically viewed as costs on an individual enterprise basis, represent income for other sectors of the UK economy. Such associated impacts include the value of employment generated by the livestock sector (£2543 million within the livestock sector and £1220 million in linked industries), backward linkages to input industries (£2931 million), and forward linkages in the supply chain into the retail industry (£14884 million). It is worth noting that whilst we have viewed labour as a positive impact of the livestock sector here, because of its importance for livelihoods, from an individual farmer's perspective, this is a cost, in which case, the profitability of the livestock sector is reduced substantially to £2794 million.

Cultural benefits based on current willingness to pay estimates were significant (£748 million), although substantially lower than provisioning benefits, and were associated primarily with beef and sheep systems, with the majority associated with hill and upland areas.

Major ecosystem costs were associated with impacts on regulating services, namely GHG (£2063 million) and ammonia (£379 million) emissions. Emissions to water were of less significance, and mainly linked to the cost associated with nitrate leaching in terms of reduced environmental water quality and removal of nitrates from drinking water (£113 million) and the cost associated with soil erosion (£84 million) in terms of flood damage and prevention. The majority of these costs are associated with grazing systems in western and northern England and Wales and non-grazing livestock in eastern England.

Table 10. Modelled valuation impact of the livestock sector

		BAU‡ £M	Sheep £M	Pigs £M	Suckler beef £M	Eggs £M	Chicken £M	Dairy & dairy beef £M	Arable subs[a] £M
A. Ecosystem benefits									
Production	Total product value	8268	826	843	828	592	1376	3802	3398
	Inputs	-2931	-161	-407	-373	-325	-596	-1068	-1863
	Labour	-2543	-526	-420	-294	-84	-331	-888	-371
	Production (less inputs)	5337	665	436	455	267	780	2734	1535
	Production (less inputs & labour)	2794	139	16	161	183	449	1846	1163
Regulation	Total	-2701	-321	-215	-571	-96	-279	-1219	-1811
	Soil erosion	-84	-51	-3	-14	-2	-5	-10	-511
	Pesticide	-12	0	-2	-1	-1	-4	-3	-29
	Eutrophication	-4	0	0	-1	0	0	-1	-2
	N leaching	-113	-16	-8	-29	-4	-15	-42	-78
	Greenhouse gas emissions	-2063	-230	-145	-422	-65	-207	-993	-1114
	Ammonia	-379	-21	-56	-81	-23	-47	-151	-77
	Faecal contamination	-10	-1	0	-4	0	0	-5	0
	Chryptosporidium	-35	-1	0	-20	0	0	-14	0
Cultural	Cultural	748	403	9	190	5	13	128	160
B. Linked impacts									
	System inputs	2931	161	407	373	325	596	1068	1863
	System labour (as above)	2543	526	420	294	84	331	888	371
	Linked labour	1220	252	201	141	41	159	426	178
	Downstream impact	14984	733	1265	828	1449	2246	8463	N/A
C. Total areas used									
	UK arable (Mha)	1.64	0.06	0.29	0.13	0.15	0.44	0.57	1.64
	Overseas arable (Mha)	1.10	0.02	0.26	0.04	0.13	0.52	0.14	0.00
	Grassland (Mha)	4.23	1.10	0.00	1.27	0.00	0.00	1.86	3.72
	Hill (Mha equivalent)	2.68	2.16	0.00	0.52	0.00	0.00	0.00	0.00
	Total UK land (Mha)	8.55	3.32	0.29	1.92	0.15	0.44	2.43	5.36

‡Business As Usual

[a] "Arable substitution" examines the potential for arable production to substitute livestock production on current livestock land that is considered to be at least marginally suitable for arable production.

The analysis also considered the possible implications of reducing or entirely withdrawing livestock production in the UK and substituting it where possible, with arable production. From a spatial analysis of soil suitability for agriculture, an estimate was derived of the degree to which arable production might replace particular types of livestock production in the UK. The level of substitution from livestock land to arable land was limited: of the total modelled land area required for livestock production in the UK (ca. 6.89 Mha), about 21% was estimated to be well-suited to arable production, with 48% entirely unsuited, and therefore likely to be abandoned from agricultural use. This would result in the loss of current biodiversity, landscape features and probably have negative effects on the tourism and recreational opportunities associated with managed landscapes.

A variety of optimising scenarios were run to examine how the livestock sector could be configured to meet Defra's objectives for the livestock sector (Table 11). With employment viewed as a benefit (Table 11A) and the model constrained to producing the same quantity of provisioning benefits, optimisation of net ecosystem value (105% of current BAU) was achieved by increasing dairy value and making greater use of free-range egg production, despite its reduced value (94% of current BAU). Some livestock land was also allocated to arable production. The main reason for the increase in net benefit flow was, however, associated with the configuration of the different sectors to increase employment generation (107% of current BAU), for example, through greater use of free-range poultry systems and more labour intensive feeding and waste management systems.

When set to achieve a 25% reduction of GHG emissions from the livestock sector, the model suggested that the optimal route to achieving this would be through reducing dairy and beef production, with arable replacing some of the land released through this process and poultry, egg, pigs and sheep remaining relatively unaffected. This was however associated with a 20% loss in net ecosystem value, partly because of lost employment opportunities and reduced cultural value from the livestock sector.

Under the red-to-white meat scenario, in which red meat was assumed to be provided only from dairy as a by-product of milk production, the model suggested that 92% of provisioning benefit could be maintained by increasing pig, poultry, and milk production and introducing arable production on the land released. However, there was a 70% loss of cultural value, associated primarily with the loss of sheep and suckler beef systems, and the overall net flow of ecosystem benefits was reduced to 83% of current BAU.

Where employment was viewed as a cost (Table 11B), net ecosystem value in the optimised BAU scenario was achieved largely through reducing labour requirements, for example, by greater use of housed poultry systems and slurry manure management. Where production from the different sectors was allowed to increase by up to 20% above current BAU production, observing the constraints of currently available land, but defining no minimum production level for any of the sectors, optimisation of net ecosystem benefit was achieved through greater reliance on dairy, egg, and poultry systems.

On the whole, there was a tendency for optimisation to be achieved at the expense of the pig sector which disappeared altogether, replaced by arable production in the BAU+20% scenario, the 25% GHG emission reduction scenario, and the red-to-white-meat scenario. This was in contrast to optimisations in which employment was viewed as a benefit, where pig production was at least equivalent to that in the current BAU.

Table 11. Selected optimising scenarios under hypothetical future conditions relative to current business-as-usual (BAU) scenario

	A. Employment as a benefit					**B. Employment as a cost**				
	Current BAU	Optimised BAU	BAU + (up to 20% +)	GHG† reduced by 25%	Red to white meat	Current BAU	Optimised BAU	BAU + (up to 20% +)	GHG† reduced by 25%	Red to white meat
	£M	£M	£M	£M	£M	£M	£M	£M	£M	£M
Provisioning	9100	104%	106%	81%	92%	2793	117%	134%	95%	124%
Arable[a]	0	21	0	557	533	0	17	268	736	678
Labour	3764	107%	104%	81%	84%	-2543	92%	81%	59%	57%
Dairy	2306	104%	118%	58%	107%	2306	107%	128%	66%	107%
Eggs	267	94%	0%	94%	86%	267	103%	124%	103%	103%
Poultry	779	99%	113%	99%	129%	779	104%	125%	104%	135%
Beef	883	102%	105%	28%	49%	883	105%	67%	25%	50%
Pigs	436	102%	123%	102%	133%	436	102%	0%	0%	0%
Sheep	664	101%	101%	101%	0%	664	101%	101%	90%	0%
Regulation	-2700	100%	101%	81%	101%	-2700	98%	97%	79%	97%
Cultural	748	99%	100%	76%	30%	748	100%	92%	73%	35%
Net value	7148	105%	107%	80%	83%	840	162%	214%	126%	131%

[a] actual values given for arable, as the arable BAU is 0, and relative values cannot be calculated
† Greenhouse gases

4. Discussion

It is clear that livestock production in the UK makes a net positive contribution to ecosystem services, particularly when employment generation is viewed as a benefit of the provisioning service. When it is viewed as a cost, net ecosystem benefits are negative for pigs, suckler beef and also the hypothetical arable uptake on livestock land. The cost of regulating services accounts for about 30% of the value of the provisioning benefit, but about 90% if the provisioning benefit is viewed net of labour and input costs. The cost associated with the loss of regulating services is more than three times the estimated benefits of cultural services. Cultural services add a further 9% to the value of livestock production (27% if the product value is net of labour and input costs), while removing livestock production completely and substituting with arable where possible would lead to more than a 90% reduction in the value of ecosystems services from current livestock land if labour is viewed as a benefit and about 60% when labour is viewed as a cost.

The analysis clearly shows the significant trade-offs between provisioning of livestock products and regulating services, especially regarding impacts on air and water. These impacts, which constitute real costs borne by others without compensation, reduce the overall economic efficiency of the sector. They reflect a failure of markets and governance to adequately pass environmental costs to polluters. This is a generic failure, and the livestock industry should not be picked out a special case.

It is important to note that while these scenarios demonstrate how the model can be used to optimise the net flow of ecosystem benefit from the livestock sector, they have not incorporated the social acceptability of losing whole sectors of the livestock industry. This can be built into the scenarios as constraints, so that, for example, the model would optimise net ecosystem benefits by not reducing production from any sector by more than a given level. Furthermore, the systems-based nature of the model means that it also provides extensive outputs on how management and physical impacts within each of the livestock sectors change.

5. Conclusion

The model enabled trade-offs between provisioning, cultural and regulating services to be analysed, confirming, for example, the important role of livestock systems in providing cultural services, particularly the contribution of less intensive systems to landscape and biodiversity, and additionally the significant trade-offs between provisioning and regulating services, especially regarding impacts on air and water. The results show the importance of the use of a systems based-LCA approach in identifying the trade-offs between the cultural benefits of extensive systems and the potential efficiencies of more intensive systems. Taking an ecosystem-services viewpoint shows the substantial influence that livestock systems have on wider assets such as cultural services that are not incorporated into normal accounting processes. Furthermore, attempting to assign a landscape value to hill and upland livestock, combined with a land suitability analysis, shows the proportion of land that would become abandoned under a “no livestock” scenario, combined with a significant loss in cultural value of the land, and highlights the trade-offs and potential losses arising from radical changes to the livestock sector in the UK.

6. References

Defra, 2007. An introductory guide to valuing ecosystem services. Report published by the Department for Environment, Food and Rural Affairs. London, UK. 68pp. Available at www.defra.gov.uk

Defra, 2009. Agriculture in the United Kingdom 2009. 146pp. Available at www.defra.gov.uk

HM Treasury, 2011. GDP deflators. Available from: http://www.hm-treasury.gov.uk/data_gdp_fig.htm

Edina, 2010. AgCensus data. Available from: http://edina.ac.uk/agcensus/

Jacobs, 2008. Environmental Accounts for Agriculture. Final report submitted to Defra. 175 pp.

Magic, 2011. Available from: http://magic.defra.gov.uk/

Millennium Ecosystem Assessment, 2005. Available from: http://www.milleniumassessment.org/en/index.asp

Morgan, RPC, Morgan, DDV, Finney, HJ, 1984. A Predictive Model for the Assessment of Soil-Erosion Risk. Journal of Agricultural Engineering Research, 30 (3), 243-253

UNEP-UK NEA, 2010. UK National Ecosystem Assessment. Available from: http://uknea.unep-wcmc.org

Williams, A.G., Audsley, E. and Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Bedford: Cranfield University and Defra. Available at www.agrilca.org, and www.defra.gov.uk

Williams, A.G., Audsley, E., Sandars, D.L., 2007. Environmental burdens of livestock production systems derived from life cycle assessment (LCA). In: Garnsworthy, P. (Ed), 41st University of Nottingham Feed Conference, Sutton Bonington, Nottingham, Nottingham: Nottingham University, 171-200.

Modelling GHG emissions of dairy cow production systems differing in milk yield and breed – the impact of uncertainty

Monika Zehetmeier[*1], Markus Gandorfer[2], Alois Heißenhuber[1], Imke J.M. de Boer[3]

[1] Institute of Agricultural Economics and Farm Management, Technische Universität München, 85350 Freising, Germany
[2]Chair Group Economics of Horticulture and Landscaping, Technische Universtät München, 85350 Freising, Germany
[3]Animal Production Systems Group, Wageningen University, Wageningen, The Netherlands
∗ Corresponding author. E-mail: *monika.zehetmeier@tum.de*

ABSTRACT

The main objective of our study was to incorporate uncertainty in modelling greenhouse gas (GHG) emissions of dairy cow production systems differing in milk yield and breed. Stochastic simulation was undertaken to account for uncertainty of main model assumptions. The developed stochastic model accounts for two different methods for handling co-products of dairy farming (beef, surplus calves): economic allocation and system expansion. Whereas the choice of method for co-product handling depends on the scope of GHG modelling the stochastic model approach gave an insight into robustness and variation of model outcomes within each method for handling co-products. The method of system expansion is recommended if the consequences of changes or mitigation options in dairy cow production need to be evaluated. In that case stochastic models offer the advantage of predicting not just an outcome, but also the likelihood of this outcome. This is of special importance identifying cost-effective GHG mitigation options.

Keywords: greenhouse gas emissions, uncertainty, stochastic modelling, dairy cow production

1. Introduction

Dairy cow production contributes to about 23 to 70% of total agricultural GHG emissions in different countries within the EU-27 (Lesschen et al., 2011). Thus a growing interest can be observed in modelling GHG emissions from dairy cow production systems and identifying cost effective GHG abatement options.

As milk is the main output of dairy farms most studies express GHG emissions produced per kg milk delivered. However, beef can be considered as an important co-product of dairy farming (beef from culled cows and surplus calves sold to fattening systems) especially within dual purpose dairy cow production systems. To account for co-products from dairy farming different methods can be observed in literature (Flysjö et al., 2011). Two main approaches can be distinguished: economic allocation and system expansion. In case of economic allocation GHG emissions are allocated between milk and co-products at the dairy farm gate according to their economic value. This approach is mainly used in the calculation of carbon footprints. It identifies GHG emissions at the dairy farm gate caused by milk production and allocates GHG emissions based on the value of milk and beef to the consumer. In case of system expansion allocation between milk and co-products is avoided by expanding the system and accounting for the alternative way of beef production (i.e. sucker cow production). It is assumed that the beef derived from culled cows and fattening of surplus calves replaces beef from suckler cow production. The avoided GHG emissions are credited to the dairy farm. The method of system expansion is recommended by the International Organisation for Standardization (ISO, 2006). This approach is especially important if the consequences of changes or mitigation options in dairy cow production need to be evaluated (Flysjö et al., 2011).

Recent determinist studies showed that the choice of method for co-product handling has a major impact on GHG emissions outcomes of dairy cow production systems (Flysjö et al., 2011, Zehetmeier et al., 2012). Despite the impact of choice of method for co-product handling it has to be considered that assumptions and input data modelling GHG emissions from dairy cow production have known uncertainties.

Many guidelines and scientific studies point out the importance of incorporating uncertainty in GHG and economic modelling (ISO, 2006; IPCC, 2006; Pannell, 1997).

The inclusion, the discussion and the reporting of model changes due to uncertainties can be important to identify robustness and variation of model outcomes and sensitive or important variables (Pannell, 1997). To show the impact of uncertainty on GHG emission outcomes a deterministic model developed to calculate GHG emissions of confinement dairy farm systems differing in milk yield and breed (Zehetmeier et al., 2012) was further developed. A stochastic model was established that accounts for uncertainty in various components. Compared with deterministic models, stochastic models offer the advantage of predicting not just an outcome, but also the likelihood of this outcome. Thus, stochastic modelling was undertaken to answer the following questions:

- Does the inclusion of uncertainty influence the ranking of modelled dairy cow production systems in terms of GHG emissions? (6000, 8000, 10000 kg milk/cow per year)
- which uncertainties have the highest impact on variation of GHG emission outcomes?

To show the impact of uncertainty within different methods for handling co-products uncertainty modelling was undertaken for economic allocation and system expansion approach.

2. Methods

A whole system model calculating GHG emissions of confinement dairy cow production systems differing in milk yield and breed has been presented in detail in another paper (Zehetmeier et al., 2012).

2.1. Description of existing model

The whole farm model incorporated dairy cows from different breeds and milk yield (6000 and 8000 kg milk/cow per year - dual purpose Fleckvieh (FV) breed; 10000 kg milk/cow per year – Holstein-Friesian (H-F) breed). Representing a typical dairy farm calves and breeding heifers were combined with dairy cow production (Fig. 1). A typical German confinement production system with dairy cows, heifers and bulls being indoor all-year-round was assumed. Forage components were maize silage, grass silage and hay. Concentrates consisted of maize, winter wheat, barley, soybean meal, and concentrates for calves. Except soybean meal and concentrates for calves the production of all forage and concentrate components was incorporated into the model (Fig. 1).
Global warming potential (GWP) in the model was calculated considering all primary (occurring on farm e.g. during feed production, maintenance of animals and manure management) and secondary sources (occurring off-farm e.g. production of fertiliser, pesticides or diesel) of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) emissions. Primary source emissions were mainly calculated according to guidelines and standard values from IPCC (2006) and Haenel (2010). To estimate CH_4 emissions from dairy cows we followed Kirchgeßner et al., (1995). Emission factors for the calculation of secondary GHG emissions were taken from literature.

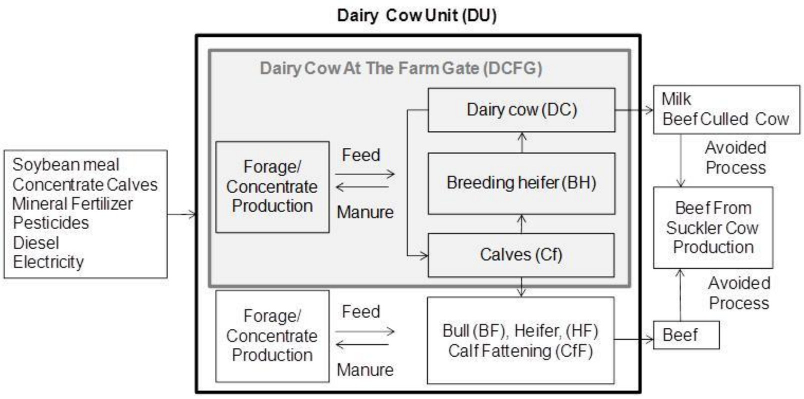


Figure 7. Illustration of system boundaries of modelled dairy cow production systems

2.2. Extension of existing model

Handling of co-products. One method to handle co-products from dairy cow production is to allocate GHG emissions between milk and co-products according to their economic value (economic allocation).
One option to avoid allocation between milk and co-products is to expand the production system by defining an alternative way to produce the co-products of dairy farming (ISO, 2006). The method named `system expansion` (Flysjö et al., 2011) was incorporated into the modelling defining suckler cow production as the alternative way to produce beef. To account for the whole potential of beef production of a dairy cow dairy units were defined (Fig. 1). A dairy unit goes beyond the dairy farm gate and considers the fattening systems of surplus calves. Thus, amount of beef of a dairy unit was made up by beef from culled cows, bull, heifer and calf fattening (only H-F dairy cows) (Figure 1). One dairy unit of a 6000 kg, 8000 kg and 10000 kg yielding dairy cow resulted in 322, 315 and 218 kg beef, respectively. Production system and calculation of GHG emissions for suckler cow production was taken from Zehetmeier et al., (2012). Suckler cows were assumed to be on pasture 185 days/year. One suckler unit resulted in 318 kg beef.
In the system expansion method, GHG emissions from suckler cow production were subtracted from GHG emissions of dairy cow production based on the potential amount of beef production (equation 1).

$$GWP_{SE}\ (\text{kg } CO_{2eq}/\text{kg milk}) = \frac{GWP_{DU}\ (\text{kg } CO_{2eq}) - \dfrac{GWP_{SU}\ (\text{kg } CO_{2eq})}{b_{SU}\ (\text{kg})} * b_{DU}\ (\text{kg})}{\text{milk delivered (kg)}} \qquad \text{Eq. 1}$$

where GWP_{SE} is GWP of milk production using the system expansion method; GWP_{DU} is GWP of one dairy unit (Fig. 1); GWP_{SU} is GWP of one suckler unit; b_{SU} is amount of beef derived from one suckler cow unit; b_{DU}= amount of beef derived from one dairy unit.

Uncertainty modelling. A deterministic model designed to simulate different yielding dairy cow and fattening production systems (Zehetmeier et al., 2012) was further developed to account for uncertainty. Stochastic simulation was carried out for main model inputs (GHG modelling, production traits, economic parameter) using @RISK (Palisade Corporation software, Ithaca NY USA). In the course of applied Monte Carlo Simulations 5000 iterations were performed to estimate probability distribution of output values.

Epistemic uncertainty (uncertainty from the modelling process, reveals due to imperfection of our knowledge, compare Walker et al., 2003) in CH_4 emissions of enteric fermentation from dairy cows was included in this model using different equations from literature (Kirchgeßner et al., 1995; Dämmgen et al., 2009; Jentsch et al., 2009). Uncertainty in N_2O emission factors from nitrogen input into soil were taken from IPCC (2006) guidelines. Emission factors chosen for soybean meal production in our model represent different assumptions of soybean meal production. Minimum value includes emissions only from soybean meal production and transport to Europe while no land use change (LUC) was assumed (0.34 kg CO_{2eq}/kg) (Dalgaard et al., 2008). A mixture of previous land use being converted to produce soybean meal was assumed for the calculation of most likely value (3.1 kg CO_{2eq}/kg) (Flysjö et al., 2012). Maximum value represents a worst case, as it is assumed that forest was converted to arable land for the production of soybean meal (10 kg CO_{2eq}/kg) (Flysjö et al., 2012). Triangle distribution function was used to describe probability distribution of CH_{4ent} and emission factors included in uncertainty modelling.

Variability uncertainty (i.e. intrinsic variability stemming from inherent variations in the real world, compare Walker et al., 2003) for three different production traits of dairy cow production systems were investigated: (1) yearly milk yield per dairy farm (kg milk/cow per year), (2) calving interval and (3) replacement rate. Data provided by LKV Bayern (unpublished data) and LKV Weser Ems (unpublished data) for 2004-2010 (LKV Bayern)/ 2009 (LKV Weser Ems) were used to identify variability uncertainty within (variability of average milk yield/cow per farm from one year to another) and between (variability of calving interval and replacement) dairy farms with equivalent milk yields. Data included 19070 dairy farms breeding FV cows and 3200 dairy farms breeding H-F dairy cows. Weighted (farm size) linear regression models were calculated consecutively with detrended milk yield as a dependent variable and standard deviation of yearly milk output per farm, mean calving interval and replacement rate per farm as independent variables. The method of quantile regression was used to calculate the standard deviations of calving interval and replacement rate between dairy farms as a function of detrended milk yield. Resulting production trait values for different yielding dairy cow production systems are shown in Table 1. Normal distribution was assumed for all considered production traits.

Table 1. Mean and standard deviation (SD) of data input for stochastic modelling of production traits (milk output, calving interval and replacement rate) for model systems yielding 6000, 8000, and 10000 kg milk/cow per year.

System milk yield (kg milk/cow/yr)	Milk yield (kg/cow/farm/yr)		Calving interval (days)		Replacement rate (%)	
	Mean	SD	Mean	SD	Mean	SD
6000	6000	280	405	22	32.6	7.6
8000	8000	342	389	15	36.7	7.6
10000	10000	373	416	17	30.3	6.4

Uncertainty in prices of beef from culled cows and calf prices was incorporated into the modelling when calculating allocation factor of economic allocation method. No parametric distribution for prices was found. Thus a nonparametric approach based on the empirical cumulative probability function of costs and prices over a period of 10 years (2000-2010) was chosen (ZMP, various volumes; AMI, 2011). Greenhouse gas emission inputs parameters were assumed to be independently distributed. Statistically significant correlations between prices were modelled.

Emission factors for GHG emissions from suckler cow beef production were taken from Crosson et al., (2011). In their study Crosson et al., (2011) showed an overview of GHG emissions from beef production systems of different countries and models. Based on the study of Crosson et al., (2011) we included 15 values for GHG emissions of beef from suckler cow production using cumulative probability function. Emission factors per kg beef varied from 15.6 to 37.5 kg CO_{2eq}.

3. Results

Probabilistic simulation was undertaken for all considered parameters simultaneously. Fig. 2 shows cumulative probability of GHG emissions for both scenarios of handling co-products (economic allocation and system expansion). In case of economic allocation the 6000 kg yielding dairy cow system showed highest GHG emissions at each level of probability. Greenhouse gas emissions varied from about 1.1 to 2.4 kg CO_{2eq}/kg milk (Fig. 2a). Probability that the 10000 kg yielding dairy cow system resulted in higher GHG emissions than the 8000 kg yielding dairy cow systems was 77% (Fig. 2a).

The ranking of cumulative probability graphs changed if system boundary was expanded from the dairy farm gate to the whole system of milk and beef production (system expansion). Depending on the amount of beef as a co-product, modelled dairy cow production systems were credited with a certain amount of GHG emissions from suckler cow production (the alternative way producing the same amount of beef). In case of system expansion modelled production systems including 10000 kg yielding dairy cows resulted in highest GHG emissions at each level of probability. Probability that dairy cow production system 6000 had lower GHG emissions than dairy cow production system 8000 was 60%. Total level of GHG emissions decreased considerably for all modelled dairy cow production systems. Greenhouse gas emissions ranged from negative values of minus -0.5 to 1.9 kg CO_{2eq}/kg milk for the 6000 and from 0.2 to 1.7 kg CO_{2eq}/kg milk for the 10000 yielding dairy cow production system.

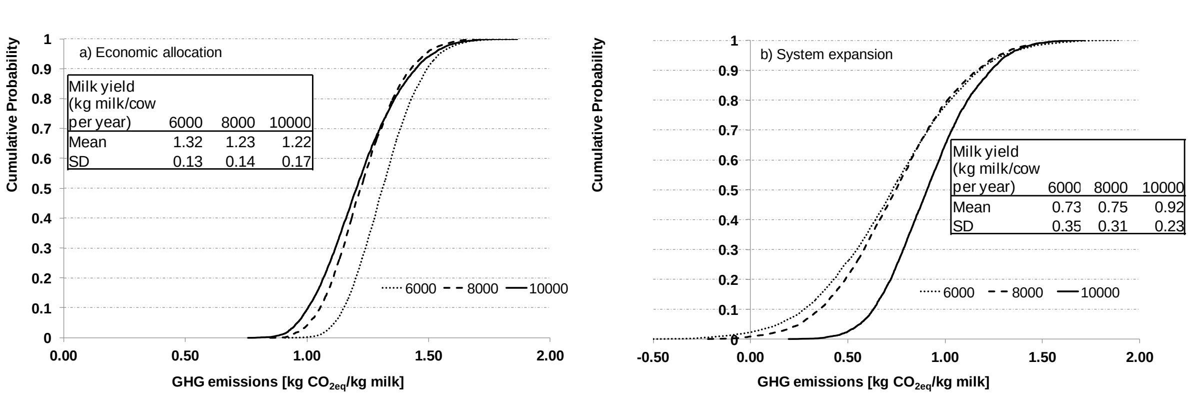


Figure 8. Cumulative probability of GHG emissions considering uncertainty of GHG emission factors, production traits and prices.

Multivariate linear regression was undertaken calculating the impact of each input variable considered in the uncertainty modelling. In the case of uncorrelated input variables squared standardized regression coefficients sum up to r-squared value of the whole model (Murray and Conner, 2009) giving insight into the proportion of total variation of GHG emissions which can be explained by the variation of each variable (Bortz and Weber, 2005). Figure 3 shows the proportion of variance of different input variables for the modelled dairy cow production systems and the investigated allocation methods. In case of economic allocation the impact of emission factors for soybean meal and direct N_2O emissions dominated total variation accounting from 79% for the 6000 kg yielding dairy cow production system to 92% for the 10000 kg yielding dairy cow production system. Furthermore, the variation of yearly milk output had an impact on variation of GHG emissions outcomes especially for the 6000 kg yielding dairy cow production system (13%). The impact of replacement rate on total variance of GHG emissions ranged between 3-2%

In case of system expansion variation of emission factor for beef from suckler cow production had the highest impact on variation of GHG emission outcomes within dual purpose dairy cow production systems (54% for the 6000 and 43% for the 8000 yielding dairy cow production system). Impact of replacement rate could be negated (0.9 - 0.2%). Higher culling rates resulted in higher amount of beef from culled cows per year which reduced the amount of suckler cows needed for beef production. Thus, the effect of reduced GHG emissions due to fewer replacement heifers was reversed.

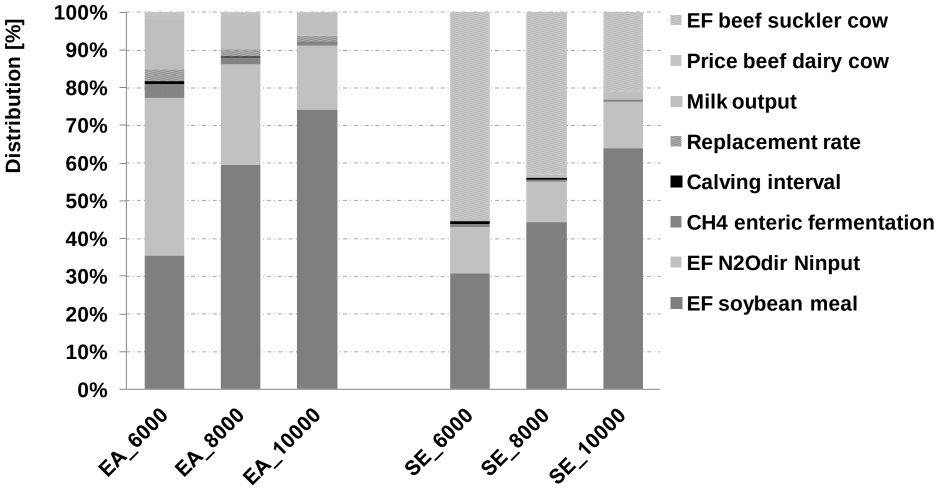


Figure 3. Parameters influencing variation of GHG emission outcomes. EA = economic allocation, SE=system expansion, EF=emission factor

4. Discussion and Conclusions

The main objective of this study was to incorporate uncertainty of main assumptions and parameters from a deterministic model modelling GHG emissions from different dairy cow production systems. Two different methods for handling co-products were used.

In consistence with other studies using deterministic model approaches (Flysjö et al., 2011; Zehetmeier et al., 2012) our study showed that the method for handling co-products had the highest impact on total value of GHG emissions. Mean values decreased up to 56% when system expansion was applied in comparison to economic allocation. Flysjö et al., (2011) discussed different methods for handling co-products comparing New Zealand and Swedish dairy cow production systems. Study results showed that GHG emissions per kg milk decreased 37% when system expansion was applied compared to allocating 100% of impacts to milk. However, in their study different allocation methods did not influence the ranking of modelled systems.

Due to the high uncertainty of emission factor for beef from suckler cow production standard deviation of GHG emissions were higher within system expansion in comparison to economic allocation. Considering uncertainty of emission factor for beef from suckler cow production even negative GHG emissions per kg milk were calculated for the dual purpose dairy cow production systems. This shows that if surplus calves from dairy cow production systems replace calves from suckler cow production systems the GHG emissions from the dairy farm could be reversed. The finding that system expansion could result in negative GHG emissions emphasizes the recommendation that this method is not suitable to calculate e.g carbon footprints of dairy farms. However, despite the high degree of uncertainties the method of system expansion gives insight if changes of GHG emissions at the dairy farm could be reversed by changes in other systems affected.

Stochastic models offer the advantage to give insight on the robustness and probability of model outcomes (Pannell, 1997). This is especially important in case of system expansion where changes of production systems are evaluated. In case of system expansion the stochastic model showed that dairy cow production system 6000 had lower GHG emissions than dairy cow production system system 8000 in only 60% of model runs. In contrary the increase in milk yield ongoing with a change in breed resulted in higher GHG emission at each stage of probability.

In case of economic allocation the main purpose of stochastic modelling was to identify factors which have an important impact on GHG emissions of milk production at the dairy farm. Stochastic models have advantage to give insight into the variation of GHG emissions outcomes and can identify most important factors. Regression analysis showed that uncertainty of soybean meal emission factor had the largest single impact on variation of total GHG emissions especially within high yielding dairy cow production systems. This is consistent with the study of Flysjö et al., (2012), who showed that inclusion of LUC in the emission factor of soybean meal resulted in an increase of 12 - 82% of total GHG emissions for the dairy cow production systems investigated. Thus, the calculation of carbon footprints of dairy products is mostly influenced by the knowledge of production and origin of soybean meal. While the influence of direct LUC (e.g. from soybean meal production) is already included in guidelines for carbon footprint calculations of dairy products (IDF, 2010) the inclusion of indirect LUC in GHG modelling of dairy cow production systems remains to be discussed (Flysjö et al., 2012). This should be focused in further research studies.

Whereas the choice of method for co-product handling depends on the scope of GHG modelling in dairy farming the stochastic model approach gave an insight into robustness and variation of model outcomes within each method for co-product handling. This is of special importance identifying cost-effective GHG abatement options. In the search for cost-effective GHG abatement options further side effects of changes in farming systems as the impact on other environmental and social indicators need to be investigated.

5. References

AMI (Agrarmarkt Informations-Gesellschaft) (Ed) 2011. Markt Bilanz Milch 2011/Getreide Ölsaaten Futtermittel 2011/ Markt Bilanz Vieh und Fleisch. Bonn.

Bortz, J., Weber, R., 2005. Statistik für Human- und Sozialwissenschaftler. Springer Medizin, Heidelberg.

Crosson, P., Shalloo, L., O'Brien, D., Lanigan, G.J., Foley, P.A., Boland, T.M., Kenny, D.A., 2011. A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. Animal Feed Science and Technology 166-67 (SI), 29–45.

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W.A., 2008. LCA of soybean meal. Int J Life Cycle Assess 13, 240-254.

Dämmgen, U., Haenel, H.-D., Rösemann, C., Hutchings, J., Brade, W., Lebzien, P., 2009. Improved national calculation procedures to assess requirements, nitrogen and VS excretions of dairy cows in the German emission model GAS-EM. Landbauforschung 59, 233-252.

Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S., 2011. How does co-product handling affect the carbon footprint of milk? Case study of milk production in New Zealand and Sweden. Int J Life Cycle Assess 16, 420-430.

Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S., 2012. The interaction between milk and beef production and emissions from land use change - critical consideration in life cycle assessment and carbon footprint studies of milk. Journal of Cleaner Production 28, 134-142.

Haenel, H. 2010. Calculations of Emissions from German Agriculture – National Emission Inventory Report (NIR) 2010 for 2008,Sonderheft 334. vTI, Braunschweig.

IDF, 2010. A common carbon footprint approach for dairy. The IDF guide to standard lifecycle assessment methodology for the dairy sector. Bulletin of the International Dairy Federation 445:41.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Volume 4: Agriculture, Forestry and other Land Use. Intergovernmental Panel on Climate Change. Paris, France.

ISO, 2006. Environmental management – life cycle assessment – requirements and guidelines. ISO14044:2006(E). ISO, Geneva.

Jentsch, W., Piatkowski, B., Schweigel, M., Derno, M., 2009. Quantitative results for methane production of cattle in Germany. Archiv für Tirzucht – Archives of Animal Breeding 52, 587-592.

Kirchgeßner, M., Windisch, W., Müller, H.L., 1995. Nutritional Factors for the Quantification of Methane Production, in: Engelhardt, W., Leonhard-Marek, S., Giesecke, D. (Eds.), Ruminant Physiology: Digestion, Metabolism, Growth and Reproduction. Proceedings of the Eighth International Symposium on Ruminant Physiology. Ferdinand Enke Verlag, Berlin.

Lesschen, J.P., van den Berg, M., Westhoek, H.J., Witzke, H.P., Oenema, O., 2011. Greenhouse gas emission profiles of European livestock sectors. Animal Feed Science and Technology 166-67 (SI), 16–28.

Murray, K., Conner, M.M., 2009. Methods to quantify variable importance: implications for the analysis of noisy ecological data. Ecology 90 (5), 1425-1425

Pannell, D.J., 1997. Sensitivity analysis of normative economic models: theoretical framework and practical strategies. Agricultural Economics 16 (2). 139–152.

Walker, W., Harremoes, P., Rotmans, J., van der Sluijs J.P., van Asselt M.B.A., Janssen, P., Krayer Krauss, M. von, 2003. Defining Uncertainty: A Conceptual Basis for Uncertainty Management in Model-Based Decision Support. Integrated Assessment 4 (1), 5–17.

Zehetmeier, M., Baudracco, J., Hoffmann, H., Heißenhuber, A., 2012. Does increasing milk yield reduce greenhouse gas emissions? A system approach. Animal 6, 154-166.

ZMP, various volumes. ZMP Getreide Ölsaaten Futtermittel/ Marktbilanz Milch / Vieh und Fleisch - Deutschland, Europäische Union, Weltmarkt. Bonn.

Towards a sustainable animal production sector: potential and problems of LCA

Imke J.M. de Boer*, Corina E. van Middelaar

Animal Production Systems Group, Wageningen University, The Netherlands
* Corresponding author. E-mail: *Imke.deBoer@wur.nl*

ABSTRACT
A transition to a sustainable animal production sector requires an integrated life cycle assessment of potential innovations. At this moment, however, this transition process is hindered by the complexity and the uncertainty of the combined effect of an innovation on the diverse issues of sustainability. There is an urgent need to develop science-based tools that integrate socio-economic and an environmental impact along the chain to gain insight into the multidimensional and sometimes conflicting consequences of GHG mitigation options.

Keywords: animal production, uncertainty, life cycle sustainability assessment, stakeholder participation

1. Introduction

Animal production is surrounded by concerns around its environmental impact, the health and welfare of animals, the safety of animal products, and its impact on human health. Acknowledging these concerns, Dutch stakeholders along the animal production chain agreed to join forces in the transition towards a competitive sector that produces with respect for animals, humans, and the environment. A lot of research has been directed at feeding, breeding, technological or management innovations to improve sustainability performance of the animal production sector. Moreover, industry partners and other stakeholders start to invest in development of science-based tools to improve and monitor their sustainability performance (www.sustainabilityconsortium.org).

The aim of this paper is to review potentials and problems related to using life cycle assessment (LCA) in the transition to a sustainable animal production sector. We used GHG mitigation as a case study to illustrate the potential and problems of using LCA in the field of animal production.

2. Mitigating greenhouse gas emissions in livestock production

Livestock production is recognized to contribute significantly to emission of greenhouse gases (GHGs, Steinfeld et al., 2006), mainly through emission of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). CO_2 is released from combustion of fossil fuels to power machinery, from burning of biomass, and from microbial decay related to, for example, changes in land use or in crop management (Janzen, 2004). CO_2 can be sequestered also by transforming arable land into permanent grassland. CH_4 is produced when organic matter decomposes in oxygen deprived conditions, for example, during enteric fermentation (especially in ruminants) and storage of manure (Mosier et al., 1998). CH_4 is also inadvertently released during fossil fuel extraction and refining. N_2O is released during microbial transformation of nitrogen in the soil or in manure (i.e. nitrification of NH_4^+ into NO_3^-, and incomplete denitrification of NO_3^- into N_2; (Oenema et al., 2005) as well as during nitrate fertiliser production.

We explored studies that addressed options to mitigate GHG emissions in the animal production chain (De Boer et al., 2011). Mitigation options considered are: reducing enteric methane (CH_4) emission from ruminants, anaerobic manure digestion, and increasing annual milk yield per cow.

2.1 Reducing enteric CH_4 emission from ruminants

Several studies explored the CH_4 reduction potential of feeding strategies at animal level (Ellis et al., 2008; Grainger and Beauchemin, 2011). A feeding strategy with potential to reduce enteric CH_4 emission, for example, is replacing grass silage by maize silage in a cow's diet (Mills et al., 2001; Beauchemin et al., 2008). Dijkstra et al., (2011) showed that replacing 50% of the grass silage for maize silage in a diet containing 30% concentrates and 70% grass silage reduces enteric CH_4 levels with about 5%. Focusing at the animal level, this is a promising strategy to reduce GHG emissions. Literature, however, also shows that dietary manipulation not only changes enteric CH_4 emissions, but also manure composition, and hence N_2O emissions from storage and application of manure (Chianese et al., 2009; Kebreab et al., 2010). Furthermore, replacing grass silage by maize silage might change the farm plan, i.e., part of the grassland will be ploughed for maize land. Ploughing grassland for maize land results in CO_2 and N_2O emissions, due to a change in soil carbon and nitrogen levels (Vellinga et al., 2011; Van Middelaar et al., 2012a). Moreover, cultivating maize instead of grass requires different fertilisation and land management, changing N_2O emissions from crop cultivation and emissions related to production of fertilisers (Schils et al., 2005; Basset-Mens et al., 2009).

To illustrate the importance of life cycle thinking in animal production, we assessed the GHG reduction potential of increasing maize silage at the expense of grass silage in a dairy cow's diet at three interdependent, hierarchical levels, i.e. the animal, farm, and chain level. A mechanistic model to predict enteric CH_4 emission at cow level is combined with a linear programming (LP) model to predict effects of a dietary change at farm level, and with life cycle assessment (LCA) to predict GHG emissions at chain level. The impact of the level of analysis is demonstrated using the case of an average Dutch dairy farm (Van Middelaar et al., 2012b).

Results of this case study showed that per ton of fat-protein-corrected milk (FPCM), with an emission of 955 kg CO_2-e, increasing maize silage with one kg DM per cow per day at the expense of grass silage resulted in an annual emission reduction of 11 kg CO_2-e at animal level, 16 kg CO_2-e at farm level, and 17 kg CO_2-e at chain level. At farm and chain level, however, land use change (e.g. ploughing grassland for maize land) resulted in non-recurrent CO_2 and N_2O emissions of 720 kg CO_2-eq per t FPCM. From an animal perspective, therefore, we would conclude that this feeding strategy offers potential to reduce GHG emissions, whereas from an LCA perspective it takes up to 42 years before annual emission reductions compensate for emissions related to land use change.

This example demonstrates the *potential* of using LCA to assess the GHG reduction potential of an innovation.

2.2 Anaerobic digestions of manure

An important form of renewable energy is bio-energy produced from biomass. Biomass can be converted into biogas, composed of CH_4, CO_2 and some trace gases (e.g., hydrogen gas), by means of anaerobic digestion (AD) (De Vries et al., 2012a; Hamelin et al., 2011). This biogas can be used to produce bio-energy in the form of electricity, heat, or transport fuel. The remaining product after AD, i.e. digestate, can be recycled as organic fertiliser for crop cultivation to substitute mineral fertiliser (Börjesson and Berglund, 2007). Anaerobic digestion of pig manure is expected to reduce the environmental impact of manure management by reducing storage emissions and substituting fossil fuel, but current efficiency of bio-gas production from manure only is low (EU-biogas, 2010). To increase efficiency of bio-gas production, co-substrates, such as maize silage, glycerine or food waste are generally added. De Vries et al., (2012b) compared the life cycle environmental consequences of producing bio-energy by anaerobic digestion of pig manure only (mono-digestion), and co-digestion with maize silage; maize silage and glycerin; beet tails; wheat yeast concentrate (WYC); and roadside grass. They assessed impacts on climate change, terrestrial acidification, marine and freshwater eutrophication, particulate matter formation, land use, and fossil fuel depletion. Results showed that mono-digestion performed well for most impacts, but represents a limited source for bio-energy. Co-digestion with maize silage, beet tails, and WYC (all competing with animal feed), and glycerine increased bio-energy production, but at the expense of increasing climate change (through land use change), marine eutrophication, and land use. Co-digestion with like roadside grass gave the best environmental performance. Hence, technologies that increase efficiency of bio-gas production from animal manure and from organic waste with limited value elsewhere, have most potential to mitigate GHG emissions (De Vries et al., 2012b). This example demonstrates the importance of including the environmental impacts related to production of substitutes to replace initial use of co-substrates in the analysis, or in other words evaluating the full consequences of an innovation.

This example demonstrates the *potential* of LCA to evaluate the environmental consequences (i.e. consequential LCA) of an innovation.

2.3 Increasing annual milk yield per cow

In 2010, the FAO quantified emission of GHGs along the life cycle of milk in many countries across the world (FAO, 2010). From their study you could conclude that GHG emissions per kg milk reduce as annual milk production increases. Research indeed showed that if one is able to use feed more efficiently (i.e. produce more milk with the same amount of feed or use less feed to produce the same amount of milk), GHGs per kg milk produced is reduced (Thomassen et al., 2009). Can we directly compare smallholder systems in which cows produce 500 kg of milk annually with specialised systems in which cows produce 7000 to 8000 kg? Cows in many smallholder systems in developing countries generally are not kept to produce milk or meat, but they have important other functions, such as to provide manure and draught power for crop production, or to function as a capital asset. Bosman et al., (1997) quantified the various functions of livestock in smallholder systems in the developing world in economic terms. If one allocates the total GHG emissions of a smallholder farm to various functions of the animals, based on their relative economic value, GHG emissions per kg of milk produced are not that different between a specialised, intensive production system and a smallholder system.

Zehetmeier et al., (2011) compared the CO_2-e per kg of milk for high-producing Holstein Friesians cows with CO_2-e per kg of milk for moderate-producing Fleckvieh cows. They demonstrated that this comparison was highly affected by the method of co-product handling used. In case of economic allocation, the CO_2-e per kg milk was lower for high-producing Holstein Friesian cows than for moderate-producing Fleckvieh cows, whereas in case of system expansion, the CO_2-e per kg milk was higher for high-producing Holstein Friesian cows than for moderate-producing Fleckvieh cows.

Both studies address the importance of handling multi-functionality while comparing various production systems. Besides handling multi-functionality (or co-product handling), the method used to account for land use change (LUC), such as deforestation for feed production, can have an important impact on comparison of systems. Flysjö et al., (2012) demonstrated that depending on the method of allocation applied, organic milk production in Sweden showed about 50% higher or 40% lower CO_2-e per kg of milk.

The above described studies demonstrate the complexity of animal production systems (multi-functionality of systems, impacts from land-use change due to feed cultivation), and the *problem* that methodological choices have a major impact on the evaluation of innovations. From a scientific point of view, therefore, complete transparency in methods and data are required to enable correct interpretation of results of an LCA study.

2.4 Trade-offs with other issues of sustainability

Most studies found in literature that addressed mitigation options for GHG emissions did not account for the complex interrelated effects on all GHGs, or their relation with other aspects of sustainability, such as eutrophication, animal welfare, land use or food security (De Boer et al., 2011). Genetic selection for increased annual milk production per cow, for example, might not only affect environmental impacts along the chain, but might also negatively affect animal health or fertility (De Vries et al., 2011; Oltenacu and Broom, 2010) or the social acceptance of animal production. Current decisions on GHG mitigation options in animal production are hindered by the complexity and uncertainty of the combined effect of these options on climate change and their relation with other aspects of sustainability.

There is an urgent need to integrate socio-economic impacts along the chain with consequential life cycle modeling to gain insight into the multidimensional and sometimes conflicting consequences of, for example, GHG mitigation options. Assessment of socio-economic impacts along the chain, however, might not necessarily need to follow the same methodology as environmental impact assessment along the chain. Let’s consider the example of dairy cattle welfare and the production of 1 kg of milk ready for consumption. Unlike emission of GHGs, animal welfare is a sustainability concern at the farm, during the transport of calves or cows, and during slaughtering of calves or cows, but not, for example, during the cultivation of feed ingredients or processing of milk. Moreover, assessment of dairy cattle welfare at dairy farms is already a multi-dimensional concept in itself: it requires integration of several types of indicators (De Vries et al., 2011), and defining thresholds of what is acceptable and what not seems to be an even greater challenge. Furthermore, we can wonder if we should aim at summing up welfare along the chain – welfare at the farm, during transport and slaughtering – or, that we need to focus on separate sets of indicators along the chain? More research is required to allow a socio-economic impact assessment of food chains.

3. Science-based harmonised approach

To move towards a sustainable livestock sector, we need to inform stakeholders along the chain about potential improvement options. This requires a science-based integral sustainability assessment along the chain and high-quality data. Stakeholders can contribute to improvement of data quality. Moreover, an active involvement of stakeholders in development in, for example, product specific environmental impact guidelines (e.g. IDF Guidelines) will make them aware of the complexity of environmental impact assessment, and strengthen the support for actual application of innovations. We, however, have to be aware of the fact that stakeholders are eager to defend their own interests. Experiences with stakeholder participation revealed, for example, that different stakeholders preferred different allocation methods because of differences in their interests. For example, beer brewers preferred a physical allocation (resulting in low emissions per unit of beer) for the beer processing stage, whereas feed companies preferred economic allocation (resulting in low emissions per unit of brewer's grain). This stresses the importance of development of science-based harmonisation of guidelines, which is a challenge is itself.

4. Conclusion

Current decisions on GHG mitigation in animal production are hindered by the complexity and uncertainty of the combined effect of GHG mitigation options on climate change and their relation with other aspects of sustainability. There is an urgent need to develop science-based tools that integrate socio-economic and environmental impacts along the chain to gain insight into the multidimensional and sometimes conflicting consequences of GHG mitigation options.

5. References

Basset-Mens, C., Ledgard, S., Boyes, M., 2009. Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecological Economics 68: 1615-1625.

Beauchemin, K.A., Kreuzer, M., O'Mara, F., McAllister, T.A., 2008. Nutritional management for enteric methane abatement: A review. Australian Journal of Experimental Agriculture 48: 21-27.

Börjesson, P., Berglund, M. 2007. Environmental systems analysis of biogas systems-Part II: The environmental impact of replacing various reference systems. Biomass and Bioenergy 31: 326-344.

Bosman, H.G., Moll, H.A.J. and H.M.J. Udo. 1997. Measuring and interpreting the benefits of goat keeping in tropical farm systems. Agricultural systems 53: 349-372.

De Boer I.J.M., Cederberg, C., Eady, S., Gollnow, S., Kristensen, T., Macleod, M., Meul, M., Nemecek, T., Phong, L.T., Thoma, G.,van der Werf, H.M.G., Williams, A.G., Zonderland-Thomassen, M.A., 2011. Greenhouse gas mitigation in animal production: towards an integrated life cycle sustainability assessment. Current Opinion on Environmental Sustainability 3: 423-431.

De Vries M., Bokkers E.A.M., Dijkstra T., Van Schaik G., De Boer I.J.M., 2011. Associations between variables of routine herd data and dairy cattle welfare. Journal of Dairy Science 94:3213-3228.

De Vries, J.W., Groenestein, C.M., De Boer, I.J.M., 2012a. Environmental consequences of processing manure to produce mineral fertilizer and bio-energy. Journal of Environmental Management 102: 173-183.

De Vries, J.W, Vinken T.M.W.J., Hamelin, L., De Boer, I.J.M., 2012b. Comparing environmental consequences of anaerobic mono- and co-digestion of pig manure to produce bio-energy – a life cycle perspective (in preparation).

Chianeese, D.S., Rotz, C.A., Richard, T.L., 2009. Simulation of nitrous oxide emissions from dairy farms to assess greenhouse gas reduction strategies. Transactions of the ASABE, 52: 1325-1335.

Dijkstra, J., Oenema, O., Bannink, A., 2011. Dietary strategies to reducing N excretion from cattle: Implications for methane emissions. Current Opinion in Environmental Sustainability 3: 414-422.

EU-AGRO Biogas. 2010. http://www.eu-agrobiogas.net.

Ellis, J.L., Dijkstra, J., Kebreab, E., Bannink, A., Odongo, N.E., McBride, B.W., France, J., 2008. Aspects of rumen microbiology central to mechanistic modelling of methane production in cattle. Journal of Agricultural Science 146: 213-233.

FAO, 2010. Greenhouse gas emissions from the dairy sector. Animal Production and Health Division, FAO, Rome.

Flysjö, A., Cederberg C, Henriksson M, Ledgard S., 2012. The interaction between milk and beef production and emissions from land use change - critical considerations in life cycle assessment and carbon footprint studies of milk. Journal of Cleaner Production 28: 134-142.

Grainger, C., Beauchemin, K.A., 2011. Can enteric methane emissions from ruminants be lowered without lowering their production? Animal Feed Science and Technology 166–167, 308–320.

Hamelin, L., Wesnæs, M., Wenzel, H., Petersen, B.M., 2011. Environmental Consequences of Future Biogas Technologies Based on Separated Slurry. Environmental Science and Technology 45: 5869-5877

Janzen H.H., 2004. Carbon cycling in earth systems — a soil science perspective. Agriculture Ecosystems and Environment 104:399-417.

Kebreab, E., Strathe, A., Fadel, J., Moraes, L., France, J., 2010. Impact of dietary manipulation on nutrient flows and greenhouse gas emissions in cattle. Revista Brasileira de Zootecnia 39 (SUPPL. 1), pp. 458-464.

Mills, J.A.N., Dijkstra, J., Bannink, A., Cammell, S.B., Kebreab, E., France, J., 2001. A mechanistic model of whole-tract digestion and methanogenesis in the lactating dairy cow: Model development, evaluation, and application. Journal of Animal Science 79: 1584-1597.

Mosier A.R., Duxberry J.M., Freney J.R., Heinemeyer O., Minami K., Johnson D.E., 1998. Mitigating agricultural emissions of methane. Climate Change 40:39-80.

Oenema O., Wrage N., Velthof G.L., Van Groeningen J.W., Dolfing J., Kuikman P.J., 2005. Trends in global nitrous oxide emission from animal production systems. Nutrient Cycling in Agroecosystems 72:51-65.

Oltenacu P.A., Broom D.M., 2010. The impact of genetic selection for increased milk yield on the welfare of dairy cows. Animal Welfare 19:39-49.

Schils, R.L.M., Velthof, G.L., Fraters, B., Willems, W.J., 2005. Limits to the use of manure and mineral fertilizer in grass and silage maize production in the Netherlands, with special reference to the EU Nitrates Directive. Plant Research International B.V., Wageningen, The Netherlands, pp. 1–48.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales M., De Haan, C., 2006. Livestock's Long Shadow: environmental issues and options. FAO, Rome, Italy.

Thomassen M.A., Dolman M.A., Van Calker K.J., De Boer I.J.M.., 2009. Relating life cycle assessment indicators to gross value added for Dutch dairy farms. Ecological Economics 68:2278-2284.

Van Middelaar, C.E. Cederberg, C., Vellinga, T.V., Van der Werf, H.M.G., De Boer, I.J.M., 2012a. Exploring variability in methods and data sensitivity in carbon footprints of feed ingredients. (Submitted).

Van Middelaar, C.E., Berentsen, P.B.M., Dijkstra, J., De Boer, I.J.M., 2012b. Evaluation of a feeding strategy to reduce greenhouse emissions in dairy farming: the level of analysis matters. (Submitted).

Vellinga, T.V., Hoving, I.E., 2011. Maize silage for dairy cows: Mitigation of methane emissions can be offset by land use change. Nutrient Cycling in Agroecosystems 89: 413-426.

Zehetmeier, M., Baudracco, J., Hoffmann, H., Heißenhuber, A., 2012. Does increasing milk yield reduce greenhouse gas emissions? A system approach. Animal 6 (1), 154–166.

Lessons learned from integrated environmental and socioeconomic life cycle assessments

Jean-Michel Couture[1,*], Julie Parent[2], Mia Lafontaine[3], Jean-Pierre Revéret[1,2]

[1] Groupe AGECO, 2014 Cyrille-Duquet suite 307, Quebec, QC, G1N 4N6, Canada
[2] International Chair on Life Cycle, Polytechnique/ESG UQAM; Department of Strategy, Social and Environmental Responsibility, Suite R-3555, École des sciences de la gestion, Montreal, QC, H2X 3X2, Canada
[3] Quantis, 395 Laurier Ouest, Montreal, QC, H2V 2K3, Canada
[*] Corresponding author. E-mail: *jean-michel.couture@groupeageco.ca*

ABSTRACT

An integrated Social and Environmental LCA was conducted to assess the environmental and socioeconomic performance of Canadian milk production. During the project, a number of issues and challenges arose from the integration of the two LCA techniques. This presentation focuses on the key findings and lessons learned from the study regarding the procedural and methodological integration of ELCA and SLCA. It takes stock of the difficulties faced by realisation teams at the main stages of the LCA study and the benefits resulting from such an integrated approach, thus contributing to the reflexion and works already started towards the development of Life Cycle Sustainability Assessment methodology.

Keywords: Life Cycle Sustainability Assessment, social life cycle assessment, environmental life cycle assessment, dairy production, Canada

1. Introduction

Over the years, the Life Cycle Assessment (LCA) approach has become one of the main tools used to provide market actors with the information needed to turn toward more sustainable consumption and production practices. Initially developed to assess the potential environmental impacts of a product through the Environmental Life Cycle Assessment (ELCA), this approach has since evolved to encompass all three pillars of sustainability by combining ELCA with Life Cycle Costing (LCC) – a technique assessing the cost implications of a product's life cycle – and Social Life Cycle Assessment (SLCA) – a more recent technique assessing the social and socioeconomic impacts of a product's life cycle.

However, despite their common conceptual foundations, these LCA tools still lack the integrated framework that would guide a comprehensive assessment of a product's sustainability. Although the UNEP/SETAC Life Cycle Initiative is currently developing a Life Cycle Sustainability Assessment (LCSA) methodology, further developments are admittedly needed, in particular based on findings and lessons learned from cases combining ELCA, LCC and SLCA (UNEP/SETAC 2012, 45).

LCA and its different forms are part of the environmental assessment toolbox, and the question of integration in environmental assessment has been debated for some years about other tools such as Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA) (Revéret 2011). There are many coexisting understandings of what "Integrated Assessment" is (Hacking and Guthrie, 2007). For instance, Lee (2006) identifies three distinct types of integration: 1) Horizontal: bringing together different types or categories of impacts; 2) Vertical: linking separate assessments performed at different levels/stages; and 3) Analytical: integration of assessments into decision-making. Regarding a project's execution, Eggenberger and Partidário (2000) identify three different forms of integration that were also tested and used by Revéret et al., (2000: 1) Substantive: constant consideration of social, environmental and economic (SEE) dimensions in all aspects of the studies; 2) Methodological: integration of concepts and applications; and 3) Procedural: integration of SEE dimensions in the project's planning and management. This last category is of particular interest as it was developed to analyse how integrated other forms of environmental valuation are, namely EIA and SEA.

On the basis of the definition of 'integration' given by Eggenberger and Partidario (2000), this paper discusses the 'integration' of Social and Environmental LCAs of the Canadian dairy sector, a study that we recently carried out. This presentation takes stock of integration issues and challenges encountered in this project, considering both procedural and methodological perspectives. The economic dimension, which could have been assessed with a LCC, is not considered here as it was not identified as a priority by the client.

2. Project overview

As part of the Dairy Cluster research program, the Dairy Farmers of Canada (DFC), in collaboration with Quantis Canada, Group AGÉCO and the CIRAIG (The Interuniversity Research Centre for the Life Cycle of

Products, Processes and Services), conducted an ELCA and SLCA of the Canadian dairy sector. The global objective of this study was to provide a comprehensive assessment of the sustainability of the Canadian milk production sector. More specifically it aimed to:

- Define the profile of environmental and socioeconomic impacts of the Canadian dairy sector over the life cycle of milk production (from cradle to farm gate);
- Identify potential areas for further focus in improving the dairy sector's overall sustainability;
- Provide an overall framework and building blocks to support comparison/benchmark with similar competitive products.

Launched in 2010, the project was completed in September 2012. The main deliverables include an environmental profile of the average kilogram of milk produced in Canada, as well as an evaluation of the socioeconomic performance of the Canadian dairy sector. Overall, the LCA showed the existing commitment of dairy producers towards their supply chain's sustainability, characterised by an overall good performance – both at the environmental and socioeconomic levels. At the international level, the Canadian milk sector positions itself very well, with a relatively low carbon footprint and a water footprint among the best in provinces where there is no irrigation. While there is no benchmark available to compare the social engagement of the sector, the assessment shows that the Canadian dairy farms and their Boards are already socially committed corporate citizens in regards to many social issues.

In addition to the assessment of the dairy sector's performance, the methodological advances derived from this project are also of significant scientific interest, especially in regards to the development of LCSA methodology. Although the project was not meant to be a LCSA, nor named as such, it contributed *de facto* to the development of the LCSA methodology thanks to the integration process – both at procedural and methodological levels – of ELCA and SLCA tools.

3. Procedural integration

An initial dimension of the integration of ELCA and SLCA lies in the procedural approach. It refers to how an LCA project is conducted and managed when it deals with both environmental and social issues. Should it be conducted by one integrated team? In case of more than one team involved, what kind of bond should exist between them? How should the knowledge and expertise be shared between the environmental and social teams, and between the teams and the client? These questions have no straightforward answers given the poor experience of the LCA community in this type of integration, but the lessons we have learned from the Canadian milk sector study provide some answers and highlights useful to other practitioners.

This project was conducted by three teams, two of them part of consulting firms (Groupe AGECO and Quantis) and the third one associated with a university research centre (CIRAIG). The three teams had already conducted joint LCA projects in the past, but none of this size. This project was hence the occasion to come together and overcome new challenges in integrating ELCA and SLCA.

The first challenge was integrating the teams' expertise. While the three teams' members shared a common understanding of ELCA and SLCA approaches, no one mastered both methodological frameworks. This was primarily due to the novelty of the SLCA methodology supported by the still recent (in 2009) UNEP/SETAC Guidelines (UNEP/SETAC, 2009). For this reason, one of the project's underlying objectives was to develop the SLCA methodology. The second reason follows from the first; while CIRAIG and Quantis are specialised in the conduct of ELCAs and hold a vast experience in that field, Groupe AGECO has expertise in the field of socioeconomic studies and surveys applied to the agrifood sector and has developed a growing expertise in SLCA. The team members' profiles were hence different, with the ELCA teams composed mostly of engineers and the SLCA team of agricultural economists.

This situation created a communication challenge, as each team had to become familiar with a new, yet common semantic field and methodology. To overcome this issue, mutual trainings were offered to create a common understanding of each tool's concepts and functioning. Given that SLCA and ELCA teams were located in two distant cities, such exchanges, however, were rare. Discussions on a more regular basis would have benefited both groups.

In contrast, these teams' differing profiles were also a source of significant synergy, as their expertise was complementary in many ways. In particular, Groupe AGECO's knowledge of the agricultural sector in general and of the Canadian dairy sector in particular was crucial. From the ELCA standpoint, this expertise allowed adapting the impact assessment models according to the sector's most specific characteristics. It also gave access to site-specific data, more easily collected through a farm survey supervised by Groupe AGECO. This expertise was also crucial for developing the SCLA framework, given that a good understanding of the sector's characteristics and related socioeconomic issues is essential to identify the relevant issues of concern to document and develop a comprehensive and relevant set of socioeconomic indicators. The socioeconomic background of the SLCA team also facilitated development of this new methodology.

The involvement of the research centre was also highly beneficial to the project. The team contributed to the methodological and theoretical development of both environmental and socioeconomic frameworks. Its participation allowed the two consulting groups to focus on calculating the results, while also being involved in the methodological development, especially for the SLCA assessment framework. In turn, it benefited from the work and expertise of the two other teams.

In performing the assignment, all efforts were made to conduct the socioeconomic and environmental parts simultaneously. Daily work was carried out separately within each team, but a number of tasks were done with close collaboration, such as farm data collection, to ensure that the farmers who contributed would not suffer from "survey fatigue" by receiving too many requests for data that could be gathered at once.

Since the ELCA and SLCA frameworks used in this project were mostly independent from each other (cf. section 4), it was not possible to prevent the three teams from working in silos, especially during the framework development phase. This situation, however, did not impair the results' quality or relevance. Rather, it attests to the limitations encountered during the integration of ELCA and SLCA methodologies.

4. Methodological integration

Although the UNEP/SETAC Guidelines specify that a SLCA must follow the ISO 14040 and 14044 norms developed for ELCA, it does not mean that a SLCA can be readily integrated or even articulated with an ELCA. Several methodological issues arose during the project implementation. Faced with two domains that have evolved separately and are based on very different disciplines, the teams found that some concepts did not have exactly the same meaning and needed further explanation to avoid misunderstandings. This section highlights similarities and differences in the setting of methodological assets of both tools. Many are already identified in the UNEP/SETAC Guidelines for SLCA. However, as methodological choices were made to conduct the project, it is relevant to return to this issue in a case-study context.

4.1. Goal and scope

According to ISO norm 14044 (2006), "the goal and scope of an LCA shall be clearly defined and shall be consistent with the intended application". Therefore, identifying its intended application is the point of departure of an LCA study. All other methodological choices – made in the context of the "scope" definition – hence have to remain consistent with it. This holds true for both the environmental and social dimensions.

One main objective of the project was to identify potential areas of further focus for improving the Canadian dairy sector's sustainability. In the ELCA perspective, this involves identifying hotspots to target mitigation measures and reduce the potential environmental and human health impacts of the product – in this case milk – throughout its life cycle. The same is true in the SLCA perspective. The assessment aims at identifying potential and real social hotspots to be able to provide recommendations for further improvement of the system's overall socioeconomic performance for its stakeholders. While both ELCA and SLCA are intended to identify hotspots, the resulting actions of this identification differ between them. In ELCA, identification of environmental hotspots of milk production is meant to guide changes in agroenvironmental practices and input substitutions, regardless of the suppliers. In SLCA, identification of social hotspots aims to guide improvements in farmers' and suppliers' behaviours, regardless of the nature of the inputs and processes involved.

These two intended applications, while only slightly different, nonetheless had a significant impact on how the two frameworks were developed, starting with their scope. The scope includes several methodological parameters influencing the assessment and, consequently, the results. UNEP/SETAC Guidelines do not mention how the scope of a SLCA should fit that of an ELCA when both are conducted together. It is however acknowledged that given SLCA characteristics, its scope might not necessarily be the same or totally integrated with that of ELCA.

This was notably the case with the product system, which differed between ELCA and SLCA; while the former consisted of technical processes, the latter consisted of the businesses (and their geographic location) responsible for carrying out those processes. This difference was consistent with the intended application of the SLCA more focused on the behaviour of the enterprises involved in the value chain. Indeed, it is broadly recognised in SLCA literature that businesses are the relevant unit process (Dreyer et al., 2006; Macombe et al., 2010; Parent J. Cucuzzella C. Revéret J.P., 2010; Spillemaeckers et al., 2004).

Another parameter to be considered is the choice of the functional unit (FU). As stated in the Guidelines, "specification of the functional unit and the reference flows is essential to build and model the product system" (UNEP/SETAC 2009; 53). In the study, the SLCA referred to the same FU as the ELCA; however, the FU served different purposes in each. In the ELCA, it allowed quantification of the main material assets required to fulfil the primary function of the system, which allowed quantification of all the material and energy flows of the system and, hence, the elementary flows. In the SLCA, it was instead used to list the main inputs and services involved in the milk's life cycle and to identify the businesses providing them. As a consequence, the socioeconomic performance assessed was not necessarily quantitatively related to the FU, as in the ELCA (section 4.3).

4.2. Inventory

In ELCA, the inventory phase includes data collection, validation, relating the inventory to the FU and aggregation. These phases were not readily repeatable in the SLCA. First, as mentioned above, business behaviours were not directly and quantitatively related to the FU. Moreover, aggregation could not take place before the assessment, since the social norms against which business behaviours were benchmarked were site-specific and varied across regions and supplying sectors. Therefore, the SLCA inventory phase included only data collection and validation.

In both cases, site-specific data were used to document processes and behaviours occurring on dairy farms and generic ones for those occurring upstream in the life cycle. But the type of the data collected differed significantly, as did the nature of the sources consulted. In ELCA, specific data are used to get a better idea of the processes occurring at each life cycle stage, and generic databases exist to document those processes. In SLCA, both the behaviours of organisations and the benchmarks against which they are assessed need to be documented. As there are few existing databases covering socioeconomic issues at a business or sectorial level, some site-specific data are hence needed to perform a detailed analysis. Proxies (samples of businesses representative of the sector) can also be used to infer how the businesses actually involved in the life cycle behave. This can be done by using sectorial or national data about the issues of concern assessed. In that case, however, potential rather than actual behaviours of businesses are assessed. Given that site-specific data were only collected at the farm level in the SLCA of Canadian milk, upstream suppliers' behaviours were hence documented using proxies, and this led to the development of two distinct assessment frameworks (section 4.3).

4.3. Impact assessment

Whereas an ELCA assesses environmental impacts created by quantified stressors generated by a FU, assessment of business behaviours in an SLCA refers to "performance reference points" (PRP) to benchmark observed behaviours against recognised or expected social norms. Assessment of business behaviours based on expected behaviours or expected results of behaviours, defined as PRP in the Guidelines (UNEP-SETAC, 2009), is broadly used (Ciroth and Franze, 2011; Foolmaun and Ramjeeawon, 2012; Ugaya, 2012).

More specifically, two types of social assessments were conducted depending on the nature of data collected (site-specific or generic). In the latter, farm supplier behaviours were assessed against internationally recognised social norms to identify potential social hotspots (i.e., the possibility of encountering risky behaviours which might negatively impact stakeholders). This approach was named potential hotspot analysis. In the former, an approach named (social) "specific analysis" was conducted instead by assessing the current behaviour of dairy farmers to evaluate their degree of social responsibility towards their stakeholders. The use of some comprehensive, high-quality data and relevant PRPs allowed the assessment of not only risky behaviours, but also socially committed ones, rated with a four-level semi-quantitative scale (i.e., risky, compliant, proactive, committed behaviour) developed during the study. This approach, allowing semi-quantitative evaluation of the socioeconomic performance of behaviours of businesses involved in the Canadian milk life cycle, differs significantly from that used in ELCA. As a consequence, it has a significant impact on integration and interpretation of results.

4.4. Interpretation

Both the ELCA and SLCA identified hotspots on which the DFC and its members could act. Recommendations were formulated to foster practices minimising both the negative environmental and social impacts and improving the sector's overall environmental and socioeconomic performance. But given the nature of the two techniques, their results and recommendations could not be fully integrated. While some inferences were made, such as that behaving in a more socially responsible manner could decrease dairy farm environmental impacts, no causal effect between socioeconomic and environmental performances could be established. The two frameworks, however, fulfilled their intended application in a comprehensive and meaningful way. Adjusting the SLCA framework to obtain only results that could be integrated with the ELCA, for example using only quantitative data related to the FU, would have significantly reduced the scope and completeness of the assessment.

As the Guidelines point out, “the ultimate objective for conducting a SLCA is to promote improvement of social conditions and of the overall socioeconomic performance of a product throughout its life cycle for all of its stakeholders” (UNEP/SETAC 2009; 50). This was also the project’s objective. Based on our experience, it seems that there are important trade-offs to consider regarding the tools’ intended application to fully integrate the ELCA and SLCA methodologies.

5. Conclusion

Conducting a joint ELCA and SLCA raises interesting challenges and opportunities, as shown by this project. On the procedural side, there were many benefits gained from conducting them together, especially the dialogue that took place between the ELCA and SLCA practitioners and researchers. The fact that the three teams were composed of members of various academic backgrounds holding complementary expertise also contributed to developing a coherent framework adapted to the client’s needs. The biggest procedural challenges were met at the beginning of the project and related to mutual misunderstanding of each tool’s functioning. This issue was overcome by the end of the project, each team now having a strong and lasting understanding of both approaches. While the physical distance between the SCLA and ELCA teams was sometimes challenging and encouraged working in silos, it did not significantly impact the conduct of the project.

The lessons learned on the methodological side are also of significant interest. As the project’s objective was to identify potential areas for further improving the Canadian dairy sector’s sustainability, conducting the SLCA and ELCA together provided the sector with a comprehensive and coherent roadmap to improve its overall environmental and socioeconomic performance. This project displayed that the intended applications of ELCA and SLCA are indeed complementary and contribute, when conducted together, to progress towards sustainability. Our experience shows, however, that the methodological integration required is not simple. To fulfill each tool’s purpose, independent frameworks were needed. Working differently could have impaired one or both assessments. This raises questions about the development of LCSA. While a stronger integration of LCA techniques is needed to progress towards sustainability, such integration must be done in accordance with each technique’s methodological requirements.

6. References

Ciroth, A. and Franze, J., 2011. Conducting a Social LCA. Workshop on Social Aspects of Products Over the Whole Life Cycle, Berlin.

Dreyer, L., Hauschild, M., Schierbeck, J., 2010. Characterisation of social impacts in LCA - Part 1: Development of indicators for labour rights. Inter. J. Life Cycle Ass., 15(3), 247-259.

Foolmaun, R. and Ramjeeawon, T., 2012. Comparative life cycle assessment and social life cycle assessment of used polyethylene terephthalate (PET) bottles in Mauritius. Int. J. Life Cycle Ass., 1-17.

Hacking, T., Guthrie, P., 2007. A framework for clarifying the meaning of Triple Bottom-Line, Integrated, and SustainabilityAssessment. J. Env. Imp. Ass. Rev., 28(2-3), 73-89.

Lee, N. 2006. Bridging the gap between theory and practice in integrated assessment. J. Env. Imp. Ass. Rev., 26(1), 57-78.

ISO (International Organization for Standardization), 2006. ISO 14040, Environmental management - Life cycle assessment - Principles and framework.

ISO (International Organization for Standardization), 2006. ISO 14044, Environmental management - Life cycle assessment - Requirements and guidelines.

Macombe, C., Feschet, P., Garrabé, M., Loeillet, D., 2010. Reporting the social indicators to the functional unit for food product. Theoretical contribution regarding the collection of relevant data. 7th International Conference on Life Cycle Assessment in the Agri-Food Sector. Bari: Italy.

Parent J., C Cucuzella, JP Revéret, 2010. Impact assessment in SLCA: sorting the sLCIA methods according to their outcomes in International Journal of Life Cycle Assessment, Published online 19-01-2010

Parent, J., C. Cucuzzella, J.P. Revéret, 2012. Reflecting on the role and the life cycle perspective in LCA and SLCA used for Sustainable production and consumption. International Journal of Life Cycle Assessment Published online 09-08-2012

Eggenberger, M., Partidário, M.R., 2000. Development of a framework to assist the integration of environmental, social and economic issues in spatial planning. Imp. Ass. And Project Appr., September 2000. 201-207.

Revéret, J.P., Sambo, C., Vincelette, M. Taillon, B., 2000. An integrated SEIA for a large mining project in South East Madagascar: Challenges and approaches. International Association for Impact Assessment Annual Conference, Hong Kong.

Revéret, J.P., 2011. Développement durable et évaluation de la durabilité/soutenabilité, 3ème conférence méthodologique de l'Institut wallon de l'évaluation, de la prospective et de la statistique. December 2011.

Spillemaeckers, S., Vanhoutte, G., 2006. A Product Sustainability Assessment. Management Models for Corporate Social Responsibility, Springer Berlin Heidelberg, 257-264.

Ugaya, C., M., L., 2012. Social Life Cycle Assessment of Natura's cocoa soap. Social LCA Workshop 2012, Berlin.

UNEP/SETAC, 2009. Guidelines for Social Life Cycle Assessment of Products. United Nations Environment Programme. Paris.

UNEP/SETAC, 2012. Toward a Life Cycle Sustainability Assessment: Making Informed Choices on Products. Preliminary Version. United Nations Environment Programme. Paris.

Sustainability assessment of U.S. dairy: environmental, economic and social

Gina Vickery-Niederman[4], Greg Thoma[1*], Jennie Popp[2], Marty Matlock[2], Catherine Benoît-Norris[3], David Shonnard[5,6], Felix Adom[5,6], Eric Cummings[2], Zara Niederman[2]

[1] Ralph E. Martin Department of Chemical Engineering, University of Arkansas, Fayetteville, AR, USA
[2] Center for Agricultural and Rural Sustainability, Division of Agriculture, University of Arkansas, Fayetteville, AR, USA
[3] New Earth, York, ME, USA
[4] Department of Environmental Dynamics, University of Arkansas, Fayetteville, AR, USA
[5] Department of Chemical Engineering, Michigan Technological University, Houghton, MI, USA
[6] Sustainable Futures Institute, Michigan Technological University, Houghton, MI, USA
*Corresponding author. E-mail: *gthoma@uark.edu*

ABSTRACT

This assessment examines US dairy's environmental and social impacts in light of dairy's contribution to the US economy. The study includes a life cycle assessment (LCA) of fluid milk consumed in the US, an economic Input-Output (IO) analysis for US dairy, a social scoping assessment (SSA) of the global dairy supply chain and a literature review of the social impacts of US dairy production. The LCA identifies potential environmental impacts for the farming stages of the fluid milk life cycle, most notably in the categories: land occupation, terrestrial acidification/nitrification and human toxicity. Post-farm gate activities contribute to terrestrial and aquatic ecotoxicity, ozone depletion and mineral extraction. The social analysis identifies areas of social risk related to a consolidating dairy industry. The economic analysis finds that in addition to direct economic contributions, the US dairy industry contributes greatly to employment, labour income and value added in other agricultural and nonagricultural nodes of the US dairy supply chain.

Historically society has placed great emphasis on economic contributions and the personal value of consumption relative to potential environmental and social impacts. More recently, our relative ability to pinpoint environmental impacts for complex life cycles may lead to the over-emphasis of environmental factors relative to socio-economic impacts. Advancements in sustainability assessment would allow us to quantitatively identify relationships between environmental, social and economic impacts of a common product system. Practical tools, such as unit process-level economic and social databases would advance the field of sustainability assessment in this direction.

Keywords: dairy, sustainability, LCSA, Input-Output, Social LCA

1. Introduction

This paper examines the environmental, economic and social impacts of US dairy. Exploring these "three pillars of sustainability" (UN GA, 2005) alongside one another encourages a more comprehensive and balanced view of product sustainability. The three pillar assessment presented below identifies the potential environmental and social concerns of US dairy alongside dairy's contribution to the US economy.

2. Methods

The environmental analysis uses LCA to identify environmental impacts for the cradle-to-grave life cycle of fluid milk. The primary time frame of the assessment was 2007-2008. The LCA was structured following ISO 14040(44):2006 standards and modeled using SimaPro Software (version 7.2) (PRé Consultants, Amersfoot, Netherlands). The functional unit was 1 kg of fluid milk consumed in the US, where a national "average" milk was created as the sales-volume-weighted average of four milk fat content varieties. Product loss in the supply chain, including wasted or spoiled milk by consumers and out-of-date milk at retail, required a reference flow of 1.42 kg milk from the farm to achieve the functional unit. Because every farm produces milk with different fat and protein composition, we normalised on-farm production to a standard milk using the National Research Council approach for fat-protein corrected-milk (FPCM).

A significant quantity of the information for this analysis was generated as part of a GHG emissions study for US dairy (Thoma et al., 2012) and augmented to enable broader LCIA with published data from peer reviewed literature and the USDA National Agricultural Statistics Service. Additional information was obtained through discussions with industry representatives and other experts. The cut-off criterion for the study, generally applied at the scale of an individual life cycle stage, was as follows: flows contributing less than 1% of relevant impacts could be omitted from the model; however, small flows were not omitted if data were readily available. The EcoInvent life cycle inventory database provided data for background unit processes (Frischknecht & Rebitzer, 2005). The life cycle impact assessment methodology, Impact 2002+ (Jolliet et al., 2003), was used to assess environmental impacts.

The economic analysis uses Input-Output (IO) Analysis to examine how dairy production, dairy processing and dairy-related activities contribute to three areas of the US economy: employment, labour income and value-added. Employment measures the number of wage-, salary- and self-employed workers in the sector, labour income includes wages (worker salaries, payments and fringe benefits) and proprietary income (all income received by self-employed individuals). Value added to the US economy is a measure of labour income plus indirect taxes (excise, sales and property taxes, as well as, fees, fines, licenses and permits) and other property-type income (dividends, interest payments, rents and profits). Within these three areas, the economic IO analysis measures the direct and indirect economic impacts of the US dairy sector. Direct impacts stem from dairy production and processing. Indirect economic effects arise from the dairy sector's purchases of goods and services from other sectors.

The economic IO analysis was conducted using the IMPLAN IO software for the year 2008 (MIG, Minnesota, US). The US dairy industry was comprised of five IMPLAN sectors: Dairy cattle and milk production, Fluid milk and butter manufacturing, Cheese manufacturing, Dry-condensed and evaporated dairy products and Ice cream and frozen dessert manufacturing. Dairy's contribution to both the US economy and to overall agricultural economic activity was computed using methods described by Popp et al., (2010).

The social analysis uses SSA to calculate the percentage of worker hours for the supply chain of US dairy products that are attributable to country-specific sectors. The SSA relies on an extension of the Global Trade Analysis Project (GTAP) global IO model. This model maps 2008 worker hours for the supply chain of dairy products according to the sectors and countries involved in production (Benoît-Norris et al., 2012; Benoît et al., 2011). As with the economic IO model, the supply chain analysis is conducted at the sector level. The sector classification scheme for the SSA differs from the scheme used in the economic IO analysis (see Discussion section). The SSA includes the GTAP sectors "Raw Milk" and "Dairy Products". Raw Milk refers to dairy farming and Dairy Products is comprised of manufactured dairy products, specifically the following United Nations Central Product Classification categories: Processed liquid milk, Cream, Milk and cream in solid forms, Milk and cream concentrated or containing added sugar or other sweetening, Yoghurt and other fermented or acidified milk and cream, Dairy products n.e.c., Butter and other fats and oils derived from milk, Cheese and curd, Lactose and lactose syrup, Ice cream and other edible ice, Casein. Results of the SSA identify potential social hotspots throughout the US dairy supply chain. Presenting the share of worker hours according to country-specific sectors is particularly relevant because social impacts often manifest through employment (socialhotspot.org). In addition, a literature review examines social issues of on-farm dairy production in the US but does not include other stages of the dairy life cycle. A review of social impacts for other life cycle stages will be included in future research.

3. Results

Environmental impacts for the life cycle of fluid milk consumed in the US are presented in Figure 1 by IMPACT 2002+ category and life cycle stage. Farming stages of the life cycle, including feed production and dairy farming, contribute significantly (50+%) to several impact categories, including land occupation, terrestrial acidification/nitrification, respiratory organics and respiratory inorganics. The feed production stage is responsible for 99% of land use and ecosystem impacts through land occupation for crop production. Farming stages are responsible for 95% of terrestrial acidification/nitrification, particularly through on-farm release of ammonia, which can be oxidized to nitric acid in the environment.

Approximately 50% of human health impacts, comprised of respiratory organics, respiratory inorganics, and carcinogens, occur by the farm gate. The principal sources of respiratory organics are on-farm releases of volatile organic compounds, primarily ethanol released from the fermentation of silage. Field application of fertilisers (especially urea), manure management, off-farm energy generation and dust from operations contribute to respiratory inorganics. Farming stage fuel use contributes to carcinogens primarily through release of aromatic hydrocarbons associated with deep supply chain natural gas extraction and processing.

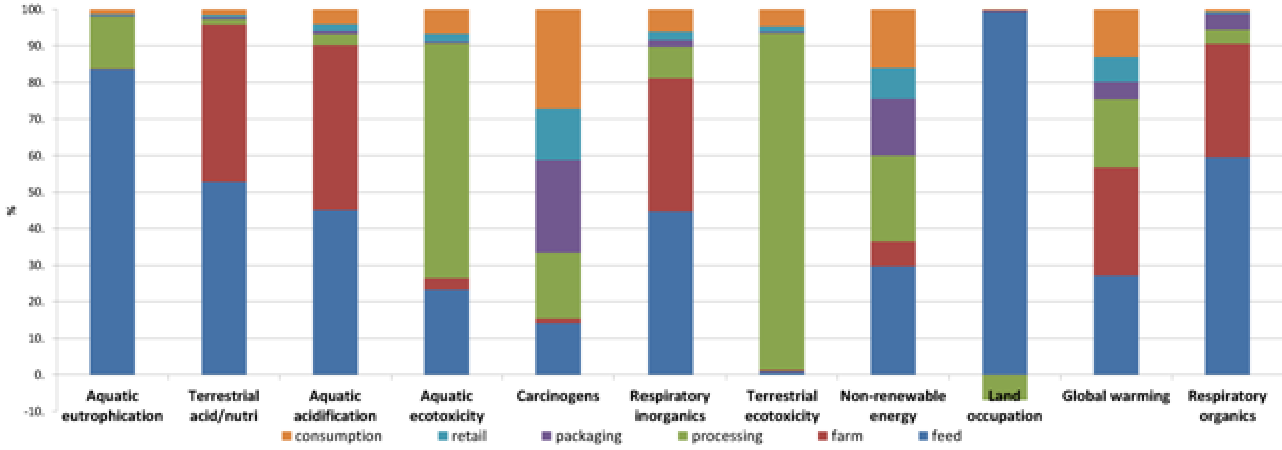


Figure 1. Environmental impact contribution analysis by life cycle stage to IMPACT 2002+ categories

Post-farm gate stages of the fluid milk life cycle also contribute significantly to environmental impacts. The processing stage is the dominant contributor to both terrestrial ecotoxicity (89%**)** and aquatic ecotoxicity **(**65%**)**. This is a direct result of the on-site waste water treatment model chosen for this analysis; in particular, the use of a unit process from Ecoinvent that is specific for whey processing, which includes emissions of phosphorus due to cleaning in place technologies and aluminum, which in the data set apparently arises from the use of alum as a flocculation aid. These emissions are highly uncertain, and may not represent practices at specific locations, therefore these impacts should be considered as warning flags, not assertions of actual impact. The negative contribution to land occupation from processing is the result of out of date milk used as pig feed displacing corn and soymeal in those animal's rations.

Normalisation of environmental impacts (Figure 2), based on 56.7 kg per capita annual fluid milk consumption, with normalisation factors specific to the US (Lautier et al., 2010) suggests that aquatic eutrophication, aquatic ecotoxicity, terrestrial acidification/nitrification, aquatic acidification and carcinogens are important categories on which to focus environmental improvement efforts. The y-axis in the normalisation analysis represents the fractional contribution of each category to total US impacts per category. Phosphorus runoff from land applied manure and fertiliser drives the contribution of aquatic eutrophication.

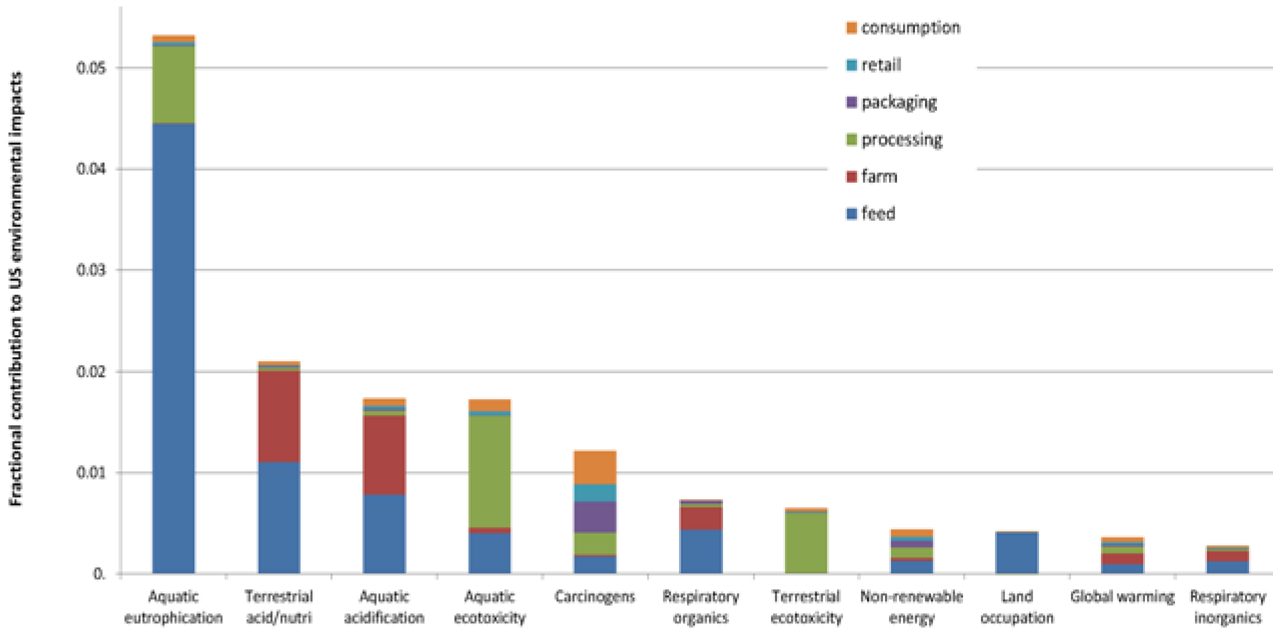


Figure 2. Normalisation results for the US; analysis based on per capita consumption of fluid milk

The economic results, which focus on positive economic contributions, rather than damages, present contributions of the US dairy supply chain to the US economy in terms of 2008 employment, labour income and value added (Table 1). Dairy contributed 407,071 jobs to the US economy, more than 5% of all direct jobs in US agriculture. Dairy Cattle and Milk Production constituted 68% of dairy employment, while the process-

ing sectors accounted for the remaining 32%. Most of dairy's direct processing employment was in Fluid Milk and Butter Manufacturing (44% of processing) and Cheese Manufacturing (30% of processing). Dairy also provided $10.1B of direct labour income (nearly 4% of US agricultural labour income) and $26.3B of direct value added (5.4% of US agricultural value added).
The dairy industry stimulated activities in other sectors. Approximately 38% of dairy's total employment impacts (listed in Table 1) and almost 61% of dairy's labour income and 53% of its value added accrued outside of agricultural sectors, making dairy an important contributor to the overall health of the US economy. Major sectors with significant contributions from US dairy include (non-agricultural) Manufacturing, Wholesale Trade, Transportation and Warehousing.
The social, global IO analysis by Benoît et al., (2012; 2011) found that "Dairy Products, US" was the top country-specific sector (in terms of worker hours) within the global supply chain of US dairy for 2008. "Dairy Products, US" contributes nearly 20% to total worker hours (skilled and unskilled) for the supply chain. "Raw Milk, US" contributes an additional 9% to total worker hours, bringing the contribution of US dairy

Table 1. Contribution of the US dairy industry to employment, labour income and value added[a]

	Employment (No. of Jobs)			Labour Income ($B)			Total Value Added ($B)		
	Dairy	Agricul-ture	% US Agricul-ture	Dair y	Agricul-ture	% US Agricul-ture	Dair y	Agricul-ture	% US Agricul-ture
Direct	407,071	7,443,502	5.47	10.1	270.3	3.73	26.3	488.0	5.39
Production	277,117	2,949,324	9.4	1.8	47.6	3.87	13.3	141.5	9.38
Processing	129,954	3,682,967	3.53	8.3	202.0	4.09	13.0	330.1	3.94
Indirect	470,925	4,338,477		25.4	283.5		44.7	485.8	
Total	877,996	11,781,978		35.5	553.9		70.9	973.8	

[a] Columns may not sum due to rounding

production and manufacturing to nearly 30%. Other country-specific sectors that contributed significantly to total worker hours relate to financial operations, including "Business Services, US" (9.2%), "Retail and Wholesale Operations, US" (9.0%) and "Financial Intermediation, US" (3.4%). The significance of sectors such as "Retail and Wholesale Operations" is in line with the results of the economic analysis. "Oil Seeds, India" is the only country-specific sector outside of the US in the top ten country-specific sectors that contributes to worker hours for the US dairy supply chain. Given that the largest percentage of worker hours for the global supply chain of US dairy products are in US dairy on-farm production and manufacturing, a review of the social impacts of US dairy-related activities is warranted.
The following literature review focuses on the social risks of on-farm US dairy production. Review of the social impacts related to dairy manufacturing and of additional social benefits are areas of future research. As a backdrop to the social sustainability of dairy production, note there are few published studies on the impacts of US dairy on local communities, aside from the social benefits of dairy's economic contributions (Jesse, 2002). The economic results presented above indicate significant socio-economic benefits in the form of employment creation, labour income and value added. To the extent dairy operations purchase local inputs, socio-economic benefits will multiply within rural communities of operation.
Also as a backdrop to social sustainability, there is a notable trend in the US dairy industry toward fewer but larger operations (USDA NASS, 2010). Alongside this trend, Lobao and Sofferahn (2008) note an increasing reliance on hired wage labour, relative to family labour. The increasing demand for wage labour has led to an increasing reliance on immigrant, particularly Latino, labour (Salant et al., 2009). A recent study of Wisconsin dairy operations identified discrepancies between immigrant and US–born dairy workers in terms of wage rates and decision-making power (Harrison et al., 2009). More research is necessary, however, to identify whether immigrant and US-born laborers were provided equal opportunities for advancement.
Lobao and Stofferahn (2008) synthesized research on the impacts of industrialised farming on socioeconomic well-being and local communities. Their review defines industrialised farms according to scale and operating attributes (see Lobao and Stofferahn, 2008). Given the trend towards dairy consolidation, impacts of industrialised farming are particularly relevant. Impacts of industrialised dairy operations in this review include a modest reduction in reliance on local inputs (Foltz et al., 2002). In addition, operators of industrialised dairies tend to know their neighbors less well and are more likely to field neighbor complaints; however, these operators tend to participate more in community organisations (Jackson-Smith & Gillespie, 2005). Lobao and Stofferahn point out that, due to differences in study location and methodologies, findings are not

always consistent; for example, Foltz and Zueli (2005) determined that large operations do not, in general, reduce reliance on local inputs in a study that included local supply considerations.

4. Discussion

The LCA identified detailed aspects of the fluid milk life cycle as contributors to specific environmental impacts. The LCA also used normalisation analysis to prioritize environmental improvement efforts by identifying impact categories where improvements would be most meaningful in a national context, in this case highlighting the importance of nutrient management in the farm stages and understanding the potential emissions from wastewater treatment at the processing stage. Together these tools provide organisations and policymakers with detailed information that can be used to develop management strategies and environmental policies.

The economic and social analyses presented here do not have the same level of process-related granularity as LCA. These analyses indicate sector-level economic contributions and social areas of risk, respectively. The sector-level focus of the IO analyses is a function of the data that populate the models. Note also that while indirect economic contributions are included, they may overestimate the actual indirect contributions of the dairy supply chain; because of allocation methods, the indirect economic contributions presented here may only be in part attributable to dairy. For example, a worker in a feed mill that supplies feed to both dairy farms and cattle farms will be counted in the indirect impacts of dairy and cattle. Thus caution must be taken as indirect contributions may overstate contributions from the US dairy sector. Another important difference between the economic and social IO analyses is that the economic IO model study region is the US only and the social IO model is global.

Economic IO analysis is not meant to inform product comparisons or identify opportunities for more sustainable practices; rather, it presents an indication of the economic contributions of the dairy industry to the overall US economy. The SSA identifies country-specific sectors along the dairy supply chain with a predominance of worker hours in order to identify areas of social risk. Not all social impacts, however, are tied to workers hours. For example, the National Dairy Animal Well-Being Initiative (2008) acknowledged that "ethical obligations associated with dairy production include a strong emphasis on animal well-being." The Initiative published management principles and guidelines in areas such as nutrition; herd health; housing and facilities; and handling, movement and transportation. Finally, the social literature review identifies potential social risks in the areas of on-farm US dairy production. Note that the studies cited are often limited to case studies or surveys in individual communities, and while they raise flags, they are not necessarily generalizable. While together these analyses provide valuable information on the individual areas of sustainability, there is limited ability to draw conclusions about the relationship between the environmental and social risks of US dairy and its economic contribution.

5. Conclusion

Overall, the LCA identified significant sources of environmental impacts throughout the life cycle of fluid milk. Farming stages of the life cycle most significantly affected land occupation, terrestrial acidification/nitrification and human health impacts. On-farm releases of ammonia drove terrestrial acidification/nitrification. On-farm releases of volatile organic compounds, manure management and the use of fertilisers and fuel drove human toxicity. Post-farm gate stages of the fluid milk life cycle also contributed to terrestrial and aquatic ecotoxicity. The normalisation analysis showed that aquatic eutrophication is by far the most significant environmental impact in terms of the fluid milk life cycle's relative contributions to environmental impacts in the US.

The economic analysis found that in addition to directly providing jobs, income and added value, the US dairy industry contributes greatly to employment, labour income and value added in other agricultural and nonagricultural nodes of the US dairy supply chain. The economic significance of these indirect impacts highlights the importance of dairy to the strength of the greater economy. Describing relationships between the economic contributions of specific supply chain activities alongside the environmental impacts of these activities would enhance organisational and policy-based decision-making.

According to the SSA a significant share of worker hours for the total US dairy supply chain are found in US dairy production and manufacturing. In the social domain, literature on dairy's impacts is sparse and limited in scope, and the literature review lacks the rigor of a LCA framework. Some evidence suggesting that a consolidating dairy industry brings new challenges to social welfare is a call for more rigorous research.

The use of process-level, life cycle-based tools to identify impacts within each of the three pillars of sustainability would improve the decision-making value of this three part assessment. To encourage life cycle-based assessments of environmental, economic and social impacts, a Task Force of the UNEP-SETAC Life Cycle Initiative has published a framework for Life Cycle Sustainability Assessment (UNEP-SETAC, 2011). This is a future area of research with potential to produce consistent and more comprehensive sustainability assessments. One critical need, noted by the UNEP-SETAC framework yet lacking in this assessment, is to define a common product system for the environmental, economic and social analysis. Use of a common product system would be more feasible if there were tools to support process-level life cycle assessment in the areas of social and environmental sustainability. As noted above, IO analysis operates on a sector-level, and in this case, the IO analyses could not be linked to the fluid milk life cycle (as a subset of the dairy sector) examined in the environmental analysis. Socio-economic databases that provide process-level background data for an agreed-upon set of socio-economic indicators is a critical research need. Compiling databases of process-level economic and social background data would be a feasible, practical step in this direction.

6. References

Benoît-Norris, C., Aulisio, D., Norris, G. (2012) Identifying social impacts in product supply chains: overview and application of the social hotspot database. Sustainability 4, 1946-1965.

Benoît-Norris, C., Aulisio, D., Hallisey-Kepka, C., Norris, G.A., 2011. TSC – Prototype social scoping report: Product category 7: Strawberry yogurt. New Earth &The Sustainability Consortium (www.socialhotspot.org).

Broom, D., Dantzer, R., Willeberg, P., Mepham, B., Noordhuizen-Stassen, E., 1999. Report on animal welfare aspects of the use of bovine somatotrophin. The Scientific Committee on Animal Health and Animal Welfare, European Union.

Capper, J.L., Castañeda-Gutiérrez, E., Cady, R.A., Bauman, D.E., 2008. The environmental impact of recombinant bovine somatotropin (rbST) use in dairy production. Proc. Natl. Acad. Sci. 105(28), 9668 -9673.

Capper, J. L., & Cady, R. A. (2012). A comparison of the environmental impact of Jersey compared with Holstein milk for cheese production. J. Dairy Sci., 95(1), 165-76. Elsevier. doi:10.3168/jds.2011-4360

Dohoo, I.C., DesCôteaux, L., Leslie, K., Fredeen, A., Shewfelt, W., Preston, A., Dowling, P., 2003. A meta-analysis review of the effects of recombinant bovine somatotropin: 2. Effects on animal health, reproductive performance, and culling. The Can. J. Vet. Res. 67, 252-264.

Foltz, J., Jackson-Smith, D., Chen, L., 2002. Do purchasing patterns differ between large and small dairy farms? Economic evidence from three Wisconsin communities. Agric. and Resour. Econ. 31(1), 28-32.

Foltz, J., Zueli, K., 2005. The role of community and farm characteristics in farm input purchasing patterns. Rev. Agric. Econ. 27(4), 508-525.

Frischknecht, R., Rebitzer, G., 2005. The ecoinvent database system: A comprehensive web-based LCA database. J. Clean. Prod. 13(13-14), 1337 – 1343.

Harrison, J., Lloyd, S., O'Kane, T., 2009. Dairy workers in Wisconsin: Tasks, shifts, wages, and benefits (Briefing No. 3). Changing hands: Hired labour on Wisconsin dairy farms. Madison, WI, Program on Agricultural Studies, University of Wisconsin-Madison.

Heijungs, R., Goedkoop, M.J., Strujis, J., Effting, S., Sevenster, M. Huppes, G. (2010). Towards a life cycle impact assessment method which comprises category indicators at the midpoint and the endpoint level, Report of the first project phase: Design of the new method. Retrieved from www.lcia-recipe.net.

Jackson-Smith, D., Gillespie Jr., G.W., 2005. Impacts of farm structural change on farmers' social ties. Soc. Nat. Resour., 18(3), 215-240.

Jesse, E. (2002) Rethinking dairyland: Background for decisions about Wisconsin's dairy industry. Marketing and Policy Briefing Paper No. 78A, May 2002, University of Wisconsin.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., Rosenbaum, R., 2003. IMPACT 2002+: A new life cycle impact assessment methodology. Int. J. LCA, 8(6), 324-330.

Lautier, A., Rosenbaum, R. K., Margni, M., Bare, J. C., Roy, P.-O., & Deschênes, L. (2010). Development of normalization factors for Canada and the United States and comparison with European factors. Sci. Tot. Env., 409(1), 33-42. Elsevier B.V. doi:10.1016/j.scitotenv.2010.09.016

Lobao, L., Stofferahn, C., 2008. The community effects of industrialized farming: Social science research and challenges to corporate farming laws. Agric. Hum. Val. 25, 219-240.

National Dairy Animal Well-Being Initiative, 2008. Principles & Guidelines for Dairy Animal Well-Being. Retrieved from www.dairywellbeing.org/guidelines.php.

Popp, J., Kemper, N., Miller, W., McGraw, K., Karr, K., 2010. The Economic Contribution of the Agricultural Sector to the Arkansas Economy in 2008, Research Report No. 989. Little Rock, AR.

Salant, P.,Wulfhorst, J.D., Kane, S., Dearien, C., 2009. Community level impacts of Idaho's changing dairy industry. Moscow, ID: University of Idaho, College of Agricultural and Life Sciences.

Thoma, G., Popp, J., Shonnard, D., Nutter, D., Ulrich, R., Matlock, M., Kim, D., Neiderman, Z., East, C., Adom, F., Kemper, N., Mayes, A., (2012). Greenhouse gas emissions from milk production and consumption in the United States: A cradle to grave life cycle assessment circa 2008. Int. Dairy J, submitted.

UN GA (2005) 2005 World Summit Outcome. Resolution A/60/1, adopted by the United Nations General Assembly on 15 September 2005

UNEP/SETAC, 2011. Towards a life cycle sustainability assessment: Making informed choices on products. Paris, France: UNEP/SETAC Life Cycle Initiative.

USDA NASS, 2010. Overview of the United States dairy industry. Retrieved from http://usda.mannlib.cornell.edu/usda/current/USDairyIndus/USDairyIndus-09-22-2010.pdf

Bioenergy production from perennial energy crops: a consequential LCA of 12 bioenergy chains including land use changes

Lorie Hamelin[1*], Davide Tonini[2], Thomas Astrup[2], Henrik Wenzel[1]

[1] Institute of Chemical Engineering, Biotechnology and Environmental technology (KBM), University of Southern Denmark, Campusvej 55, 5230 Odense M., Denmark

[2] Department of Environmental Engineering, Technical University of Denmark, Miljøvej, Building 113, 2800 Lyngby, Denmark.

[*] Corresponding author. E-mail: *loha@kbm.sdu.dk*

ABSTRACT

In the endeavour of optimizing the sustainability of bioenergy production in Denmark, this consequential life cycle assessment (LCA) evaluated the environmental impacts associated with the production of heat and electricity from one hectare of Danish arable land cultivated with three perennial crops: ryegrass, willow and *Miscanthus*. For each, four conversion pathways were assessed against a fossil fuel reference: I) anaerobic co-digestion with manure, II) gasification, III) combustion in small-to-medium scale biomass combined heat and power (CHP) plants and IV) co-firing in large scale coal-fired CHP plants. Soil carbon changes, direct and indirect land use changes as well as uncertainty analysis (sensitivity, MonteCarlo) were included in the LCA. Results showed that global warming was the bottleneck impact, where only two scenarios, namely willow and *Miscanthus* co-firing, allowed for an improvement as compared to the reference (-82 and -45 t CO_2-eq. ha^{-1}, respectively).

Keywords: perennial crops, combustion, land use changes, gasification, anaerobic digestion

1. Introduction

The ambition of the energy policy in Denmark is to reach a 100% renewable energy system by 2050 (Lund et al., 2011). Several studies have been conducted to design and optimize such a system, and these all highlight the indispensability of a biomass potential of around 35%–50% of the overall energy consumption (Lund et al., 2011; Energinet.dk, 2010; Klimakommissionen, 2010). Though biomass is a renewable energy source, it is not unlimited in supply, and does involve considerable environmental costs. One of the most critical costs of bioenergy relates to its incidence on land use changes (LUC) (Searchinger et al., 2008; Edwards et al., 2010), i.e. the conversion of land from one use (e.g. forest, grassland or food/feed crop cultivation) to another use (e.g. energy crop cultivation). One way to minimize these LUC impacts could be through favouring the cultivation of perennial energy crops (e.g. perennial ryegrass, willow and *Miscanthus*) instead of annual crops (e.g. maize, barley, wheat, sugar beet). In fact, it is acknowledged that perennial energy crops nowadays represent the most efficient and sustainable feedstock available for bioenergy production in temperate regions (Bessou et al., 2011).

The goal of this study is to assess the environmental impacts associated with the production of bioenergy (heat and electricity) from 1 hectare (ha) of Danish arable land cultivated with ryegrass, willow and *Miscanthus*, considering four different biomass-to-energy (BtE) conversion pathways: i) anaerobic co-digestion with manure, ii) gasification, iii) combustion in small-to-medium scale biomass combined heat and power (CHP) plants and iv) co-firing in large scale coal-fired CHP plants.

2. Methods

2.1. Life cycle assessment model

The environmental assessment presented in this study was performed using consequential life cycle assessment (LCA). The functional unit upon which all input and output flows were expressed is 1 ha of agricultural land used to grow the selected energy crops. The geographical scope considered for the LCA was Denmark, i.e. the data inventory for crops cultivation and BtE plants were specific for Danish conditions. Similarly, the legislative context of Denmark (e.g. fertilisation) was considered. The temporal scope considered was 20 years, i.e. all assessed systems were operated for a 20 years duration. The life cycle impact assessment was carried out according to the Danish EDIP 2003 method (Hauschild and Potting, 2005), to which one impact category, “Phosphorous as resource”, was added based on the Impact 2002+ method (Jolliet et al., 2003). Background LCA data were based on the Ecoinvent v.2.2 database, and the assessment was facilitated with the LCA software SimaPro 7.3.3. Foreground LCA data essentially included Danish-specific data for agricultural and energy conversion processes, and the impacts associated with capital goods (foreground data only) as well as those related to transportation of the residues (i.e. ash and digestate) have been excluded. The systems assessed considered three perennial crops (ryegrass, willow and *Miscanthus*) and four BtE conversion technologies (anaerobic co-digestion, gasification, combustion in small-to-medium scale biomass CHP plants and co-firing in large scale coal-fired CHP plants). A total of 12 scenarios have therefore been assessed. The system and boundary conditions are illustrated in Figure 1.

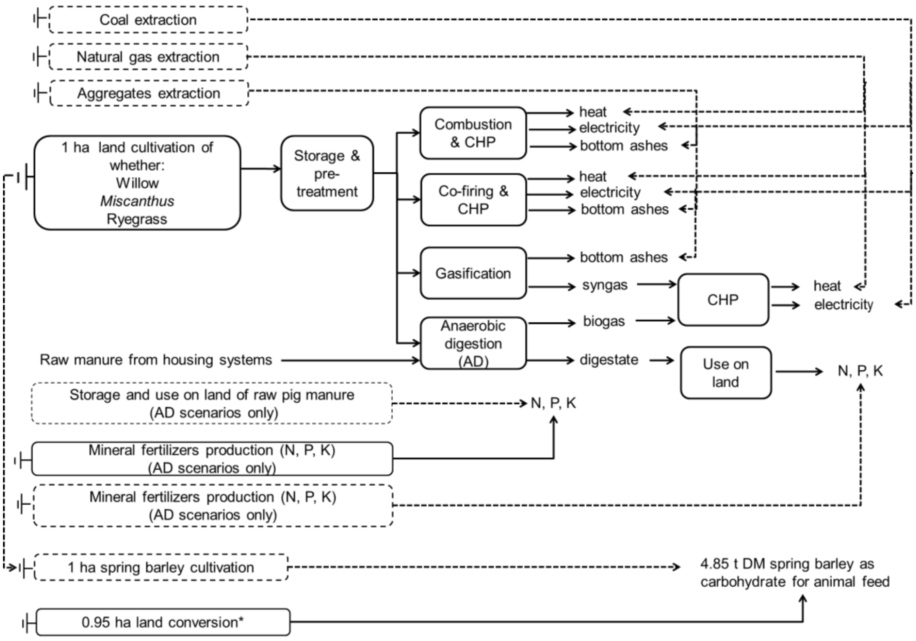


Figure 1. Illustration of the system boundary considered for all scenarios. Dotted lines indicate avoided processes, and full lines indicate processes induced by the scenarios. (*) Not all the converted land is to be cultivated in barley, and not all the Danish barley displaced is replaced, due to various market mechanisms.

For all BtE technologies, the energy produced was considered to be used for CHP production, thereby substituting the production of marginal heat and power. In the present study, the marginal electricity source was assumed to be from coal-fired power plants, and the marginal heat from natural gas based domestic boiler. As illustrated in Figure 1, the digestate produced from anaerobic digestion was used as a fertiliser (for N, P and K), which avoided marginal mineral N, P and K fertilisers to be produced and used, based on the content of N, P and K of the digestate. The marginal N, P and K fertilisers considered were calcium ammonium nitrate, diammonium phosphate and potassium chloride, respectively, conformingly with Hamelin et al., (2012). Further, based on the model from Hamelin et al., (2011), it was considered that the manure portion used for co-digestion would have otherwise been stored and applied on land, without digestion.

The three thermal bioenergy scenarios (i.e. gasification, combustion and co-firing) implied negligible residual unconverted carbon that is found in the bottom ashes, fly ashes and eventual waste water. The bottom ashes were assumed to be used for road construction, substituting for natural aggregates, while the fly ashes were assumed to be utilised for backfilling of old salt mines with negligible environmental impacts (not illustrated in Figure 1). Treatment of waste water was not included.

All bioenergy scenarios involved the use of Danish agricultural land in order to grow the energy crops. Based on Weidema (2003), it is considered that the (Danish) land needed to grow the energy crops will be taken from land under spring barley cultivation. Based on the consequential LCA logic, as well as on recent studies (Searchinger et al., 2008; Kløverpris, 2008), this resulting drop in supply of Danish spring barley will cause a relative increase in agricultural prices, which then provide incentives to increase the production elsewhere. Such increased crop production may stem from both increased yield and land conversion to cropland, the latter being also referred to as indirect land use change (iLUC) (Searchinger et al., 2008; Kløverpris, 2008). As illustrated in Fig. 1, and as in recent iLUC studies (e.g. Laborde et al., 2011; Tyner et al., 2010), this study included the environmental impacts of the latter only.

2.2. Life cycle inventory (LCI) for crops and BtE technologies

The LCI of all crops was based on a recent Danish consequential LCI (Hamelin et al., 2012), which comprises all processes involved during the cultivation stage, up to harvest, and include soil carbon changes. For all crops, the fertilisation operations were performed in conformity with Danish regulations involving an upper limit for the amount of N to be applied on the field, both as mineral fertiliser and animal slurry. Based on Hamelin et al., (2012), the life-cycle considered for perennial ryegrass, willow and *Miscanthus* plantations were respectively 2y, 21y (6 cuts; 3 years harvest cycle; 1 year establishment; 1 year preparation before planting) and 20y (18 cuts; 1 year establishment: 1 year preparation before planting). Further, it was considered that *Miscanthus* was harvested during the spring season.

Anaerobic digestion was modelled as mesophilic co-digestion of the respective energy crops with raw pig manure. The modelled methane yield for ryegrass, willow, *Miscanthus* and raw pig manure was, respectively, 358, 243, 253 and 319 Nm^3 t^{-1} VS. Based on Hamelin et al., (2011), the mixture of crop and raw pig manure was calculated in order to ensure a biomass mixture input having a dry matter (DM) content of 10% after the first digestion step. The resulting ratio manure:crop (fresh weight basis) for co-digestion of ryegrass, willow and *Miscanthus* equaled 1.6, 3 and 3.1, yielding respectively 153.3, 162 and 130 MJ CH_4 ha^{-1}. Consumption of electricity and heat was modelled according to Hamelin et al., (2012).

Gasification was modelled as fluidized bed gasification with a cold gas and carbon conversion efficiency (CGE and CCE) of 70% (±15%) and 95% (±4%), respectively. Consumption of electricity and heat was based on Jungbluth et al., (2007). Both biogas and syngas were assumed utilised in a gas engine with an average electricity efficiency of 38% (±4%) (of the LHV of the input-gas).

Combustion was modelled as direct biomass combustion in small-to-medium scale biomass CHP plants, considering electricity and heat efficiencies of 27% (±2%) and 63% (±7%), respectively. For co-firing in large scale coal-fired CHP plant, electricity and heat efficiencies of 38% (±3%) and 52% (±8%), respectively, were considered. The air emissions from biogas and syngas combustion in gas engines and from biomass combustion in CHP plants were based on NERI (2010).

Pre-treatments included on-field drying (ryegrass, for all BtE conversion technologies) and natural drying (willow, for gasification and co-firing), size comminution (all crops, for all BtE conversion technologies except direct combustion) as well as steam pre-treatment for breaking the lignocellulosic structures of *Miscanthus* and willow undergoing anaerobic digestion. Additional processes modelled in the LCA were: crops and digestate storage, use on land (UOL) of the digestate, treatment of residues from thermal BtE technologies and transportation.

2.3 Direct and indirect land use changes

The LCA system established in this study considers that the land used for cultivating the energy crops would have otherwise been used for cultivating spring barley (with straw incorporation) for the food/feed market (Figure 1). The direct land use change (dLUC) consequence of this translates into the environmental impacts of cultivating the selected energy crops instead of spring barley (Figure 1). The environmental impacts from spring barley cultivation have been included on the basis of the data from Hamelin et al., (2012).

The iLUC consequence corresponds to the environmental impact of converting land nowadays not used for crop cultivation to cropland, as a result of the induced demand for the displaced spring barley. To quantify this impact, it is necessary to identify i) how much land is converted and where; and ii) which types of land are converted (biome types). So far, most studies attempting to quantify the magnitude of iLUC used economical modeling approaches to this end, (e.g. Searchinger et al., 2008; Edwards et al., 2010; Kløverpris, 2008; Tyner et al., 2010; Laborde, 2011), but most of them focused on biofuel mandates. In Kløverpris (2008), however, the iLUC consequences in terms of points i) and ii) above are identified, for a marginal increase in wheat consumption in 4 different countries, including Denmark. In the present study, the results of Kløverpris (2008) for Denmark have been used as a proxy to estimate how much land is converted (due to the increased spring barley demand) and where. However, the CO_2 impact of land conversion is not estimated in Kløverpris. In order to do so, the soil and vegetation C data from the Woods Hole Research Centre, as published in Searchinger et al., (2008), have been used, and the CO_2 emitted due to land conversion was calculated based on the methodology published in Müller-Wenk and Brandão (2010). Based on this methodology, it was considered that 25% of the C in the soil was converted to CO_2 for all types of land use conversion, except when forests were converted to grassland, where 0% was converted. Further, it was considered that 100% of the C in vegetation was converted to CO_2 for all forest types as well as for tropical grassland conversions, while 0% was converted for the remaining biome types (e.g. shrub land, non-tropical grassland, chaparral).

2.4 Sensitivity analyses

A number of sensitivity analyses were performed for the 12 scenarios, with focus on GW. These included: a) variation (min-max) of the iLUC impacts with respect to CO_2 emissions (vs. mean value assumed as baseline); b) winter wheat as the marginal crop for Denmark (vs. spring barley as baseline); c) coal-based heat production as the marginal technology for heat generation (vs. natural gas-based as baseline); d) natural gas power plant as the marginal technology for electricity generation (vs. condensing coal power plant as baseline); e) mono-digestion of the crops (vs. co-digestion with manure as baseline); f) pre-treatment of pelletisation before co-firing (vs. 'no pelletisation' as baseline). Each of these changes was tested individually to assess the influence of the individual change on the overall LCA results.

Furthermore, the influence of the uncertainty of the several data collected in the LCA model on the ranking provided by the LCA results was tested by means of Monte Carlo analysis (number of simulations: 1000).

3. Results and Discussion

Among the selected impact categories, global warming appears critical as only two scenarios indicate overall savings for this category as compared with the fossil fuel reference (Table 1). Only co-firing of willow and *Miscanthus* indicated net overall savings, i.e. these were the only two scenarios for which an environmental benefit, GHG-wise, was identified in relation to using 1 ha of land for bioenergy. However, the magnitude of the global warming impacts found in this study (between -82 and 268 t CO_2-eq ha^{-1} over 20 years) was much higher than values published in literature, where results from -207 t CO_2-eq ha^{-1} over 20y (Brandão et al., 2010) to -700 t CO_2-eq ha^{-1} over 20 y (Styles and Jones, 2007) are reported for different bioenergy systems based on willow and *Miscanthus*. The reason for these differences is that this study, as opposed to the previous, considered iLUC, which has tremendous significance on the overall GHG balance.

The iLUC impacts of the studied bioenergy systems were the same for all scenarios (Table 1), as they all had the same "point of origin": the conversion of 1 ha of Danish land (cultivated with spring barley) to energy crops. As shown in Table 1, iLUC impacts were estimated to 309 t CO_2-eq. ha^{-1} (± 168 t CO_2-eq. ha^{-1}). The impacts were assumed to occur over a period of 20 years in accordance with IPCC (IPCC, 2006), corresponding to about 15.5 t CO_2-eq. $ha^{-1}y^{-1}$.

Co-firing and combustion provided the smallest global warming impacts for all crops. The environmental performance of co-firing was directly related to the higher electricity efficiency of these plants (about 38% of the LHV of the fuel, wet basis), and consequently to the larger amount of marginal coal electricity substituted. Co-firing of willow provided the largest savings, mostly because of the beneficial dLUC, higher yield and minimal pre-treatment required (Table 1). Similarly, the environmental performance of combustion was due to the high overall energy recovery as heat and electricity (about 90% of the LHV of the fuel, wet basis). As opposed to combustion and co-firing, anaerobic digestion and gasification involved a conversion to gas before energy generation, thereby inducing losses. Therefore, less electricity and heat were produced and substituted, resulting in larger net GW impact from these technologies. Further, UOL of the digestate contributed with a GW impact comparable to the one of iLUC, i.e. ranging between 248 (ryegrass) and 371 (willow) t CO_2-eq. ha^{-1}, primarily connected to the release of biogenic carbon not entering the soil C pool. Co-digestion also resulted in GHG savings associated with avoiding raw manure management, which would otherwise be stored and applied on land without digestion. These savings depended on the amount of manure co-digested (per hectare), i.e. the more manure co-digested (to meet the 10% DM in the input mixture), the larger the savings were. This also applied for aquatic N-eutrophication, where the impacts were much higher for ryegrass because less raw manure was involved as compared with the other crops.

Table 1 highlights the significance of dLUC for all scenarios and impact categories, where changing from spring barley to perennials generally resulted in environmental benefits. For global warming, this reflects that less C (whether soil native C, or C from above/below ground biomass) is lost as CO_2 emissions to the atmosphere during the cultivation stage for the perennials, as compared with spring barley. For the other impact categories, the dLUC results for ryegrass differed from those of *Miscanthus* and willow. Table 1 for example reflects the high load of N fertilisers applied in the ryegrass system, which resulted in much higher N leaching than in the reference system (barley cultivation), while willow and *Miscanthus* systems resulted in a dLUC improvement. On the other hand, as half of the N fertilisers used during cultivation came from animal slurries (from the inventory of Hamelin et al., 2012), which also contain P, no mineral P fertilisers needed to be applied for ryegrass, as opposed to all other crop systems, which explains the greater P savings for this crop in connection with dLUC (Table 1). For the P-related categories, the category "others" mostly reflects the avoided P fertilisers from the digestate use, which are relatively important in the anaerobic digestion scenarios.

The results of the sensitivity analyses highlighted that the variation of the iLUC impacts played the most important role for GW; with minimum iLUC impacts (7 t CO_2-eq. $ha^{-1}y^{-1}$) all bioenergy scenarios for willow and *Miscanthus* as well as co-firing of ryegrass achieved environmental savings on GW. In all other analyses, the individual changes in assumptions did not alter the conclusions relative to the baseline. However, the different assumptions made regarding marginal energy and crop decreased or increased the magnitude of the impacts or savings in all scenarios. In the case of mono-digestion, GW impacts were significantly increased as compared to their levels in the co-digestion scenarios (increase between 112 and 150 t CO_2-eq. ha^{-1}), reflecting the tremendous benefits obtained when avoiding storage and application of raw manure. The sensitivity analysis also demonstrated that additional pelletisation and milling of the biomass in the co-firing scenarios would decrease the GW performance of these scenarios to a level very close to direct biomass combustion. The results of the MonteCarlo simulation for GW supported the ranking of the bioenergy scenarios found with the baseline scenarios, demonstrating that despite of the significant uncertainties, the results obtained were robust.

Table 1. Characterised results for the selected environmental impact categories

Crop	**Ryegrass**				**Willow**				***Miscanthus***			
BtE technologies[a]	**AD**	**GA**	**CO**	**CF**	**AD**	**GA**	**CO**	**CF**	**AD**	**GA**	**CO**	**CF**
Global Warming (kg CO_2 eq/ha of perennial crop)												
dLUC	-167	-167	-167	-167	-249	-249	-249	-249	-211	-211	-211	-211
iLUC	309	309	309	309	309	309	309	309	309	309	309	309
Crop pre-treatment	118	118	118	118	46	28	28	28	42	24	24	27
Energy production	305	343	351	351	316	426	440	440	252	322	336	335
Energy substitution	-408	-339	-416	-482	-448	-433	-521	-612	-355	-330	-432	-504
Use on land (digestate)	248	0	0	0	371	0	0	0	278	0	0	0
Raw manure management[b]	-155	0	0	0	-286	0	0	0	-222	0	0	0
Other	18	2	2	2	32	2	2	2	29	1	2	-1
Net[c]	**268**	**266**	**197**	**131**	**91**	**83**	**9**	**-82**	**123**	**115**	**28**	**-45**
Aquatic Eutrophication (N) (kg N/ha of perennial crop)												
dLUC	434	434	434	434	-569	-569	-569	-569	-550	-550	-550	-550
iLUC	0	0	0	0	0	0	0	0	0	0	0	0
Crop pre-treatment	3	0	0	0	2	0,08	0,08	0,08	2	0,06	0,06	0,06
Energy production	59	43	41	41	63	52	34	34	50	41	54	43
Energy substitution	-35	-29	-34	-41	-38	-36	-43	-52	-29	-28	-35	-43
Use on land (digestate)	2960	0	0	0	2840	0	0	0	2130	0	0	0
Raw manure management[b]	-1251	0	0	0	-2300	0	0	0	-1801	0	0	0
Other	40	4	3	4	38	0,9	0,9	1	28	0,9	1	0,5
Net[c]	**2210**	**452**	**444**	**438**	**36,3**	**-552**	**-577**	**-586**	**-170**	**-536**	**-530**	**-550**
Aquatic Eutrophication (P) (kg P/ha of perennial crop)												
dLUC	-10	-10	-10	-10	-2	-2	-2	-2	-3	-3	-3	-3
iLUC	0	0	0	0	0	0	0	0	0	0	0	0
Crop pre-treatment	0	0	0	0	0	0	0	0	0	0,07	0	0
Energy production	0,01	0,4	0,02	0,03	0,01	0,3	0,03	0,03	0,03	0,3	0,03	0,03
Energy substitution	-0,2	-0,2	-0,3	-0,4	-0,2	-0,3	-0,4	-0,4	-0,2	-0,2	-0,3	-0,4
UOL (digestate)	98	0	0	0	99	0	0	0	115	0	0	0
Raw manure management[b]	11	0	0	0	5	0	0	0	8	0	0	0
Other	-17	0,1	0,1	0,2	-17	0,1	0,1	0,1	-17	-4	-3	-3
Net[c]	**82**	**-9**	**-10**	**-10**	**85**	**-2**	**-2**	**-2**	**103**	**-6**	**-7**	**-7**
P as a ressource (kg P/ha of perennial crop)												
dLUC	-157	-157	-157	-157	48	48	48	48	0	0	0	0
iLUC	0	0	0	0	0	0	0	0	0	0	0	0
Crop pre-treatment	0	0	0	0	0	0	0	0	0	0	0	0
Energy production	0	0	0	0	0	0	0	0	0	0	0	0
Energy substitution	0	0	0	0	0	0	0	0	0	0	0	0
UOL (digestate)	0	0	0	0	0	0	0	0	0	0	0	0
Raw manure management[b]	560	0	0	0	550	0	0	0	556	0	0	0
Other	-515	-1	0	0	-524	-1	-1	-1	-519	0	0	0
Net[c]	**-112**	**-158**	**-158**	**-158**	**74**	**47**	**47**	**47**	**36**	**0**	**0**	**0**

[a] AD: Anaerobic digestion; GA: Gasification; CO: Combustion; CF: Co-firing.
[b] Raw manure storage (avoided), application on land (avoided) and fertilisers (induced) because the raw manure is no longer a fertiliser.
[c] Eventual inconsistencies due to rounding.

4. Conclusion

Overall, co-firing of *Miscanthus* and willow were the options presenting the best environmental performances. These performances can probably be increased in the future as yields for these crops undergo further research and development, especially *Miscanthus*, a C4 crop. It should however be realised that a main driver for future utilisation of biomass may be to balance electricity generation from fluctuating energy sources, such as wind and solar power. Not all biomass combustion technologies may be suited for this, especially when co-generation of heat is important, as such plants can have a fixed production ratio between electricity and heat. Anaerobic co-digestion as well as gasification of biomass, on the other hand, may be operated more flexible without similar constraints. Additionally, syngas or biogas offers the flexibility of storage. On this basis, improving the environmental performance of these BtE conversion technologies would be desirable. For anaerobic co-digestion, a solution may be to favour manure-based biogas together with co-substrates not involving iLUC (e.g. straw, organic municipal household waste, garden waste) as well as in boosting the digestion process by other means (e.g. digestion in series, addition of hydrogen, etc.).

6. References

Bessou, C., Ferchaud, F., Gabrielle, B., Mary, B., 2011. Biofuels, greenhouse gases and climate change. A review. Agron. Sustain. Dev. 31, 1-79.

Brandão, M., Milà i Canals, L., Clift, R., 2010. Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. Biomass Bioenerg. 35(6), 2323-2336.

Edwards, R., Mulligan, D., Marelli, L., 2010. Indirect land use change from increased biofuels demand. Comparison of models and results for marginal biofuels production from different feedstocks. European Commission Joint Research Centre, Luxembourg, Luxembourg, European Union.

Energinet.dk, 2010. Energi 2050. Udviklingsspor for energisystemet. Energinet.dk, Fredericia, Denmark.

Hamelin, L., Jørgensen, U., Petersen, B.M., Olesen, J.E., Wenzel, H., 2012. Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark: a consequential life cycle inventory. GCB Bioenergy, DOI: 10.1111/j.1757-1707.2012.01174.x

Hamelin, L., Wesnaes, M., Wenzel, H., Petersen, B.M., 2011. Environmental Consequences of Future Biogas Technologies Based on Separated Slurry. Environ. Sci. Technol. 45, 5869-5877.

Hauschild M., Potting, J., 2005. Spatial differentiation in life cycle impact assessment – The EDIP 2003 methodology. Environmental news no. 80, Danish Ministry of the Environment, Environmental Protection Agency, Copenhagen, Denmark.

IPCC, 2006. Chapter 2: generic Methodologies applicable to multiple land-use categories, in: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tabane, K. (eds), Volume 4: Agriculture, Firestry and Other land use. 2006 IPCC guidelines for national greenhouse gas inventories, IGES, Japan, pp. 2.1-2.59.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., Rosenbaum, R., 2003. IMPACT 2002+: A new life cycle impact assessment methodology. Int. J. Life Cycle Ass. 8, 324-330.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., Spielmann, M., Stettler, C., Sutter, J., 2007. Life cycle inventory of bioenergy, ecoinvent Report no. 17. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Klimakommissionen, 2010. Danish Commission on Climate Change Policy. Green energy - the road to a Danish energy system without fossil fuels. Summary of the work, results and recommendations of the Danish Commission on Climate Change Policy, Copenhagen, Denmark

Kløverpris, J.H., 2008. Consequential life cycle inventory modelling of land use induced by crop consumption. PhD thesis: Department of Management Engineering, Technical University of Denmark (DTU), Lyngby, Denmark, ISBN: 978-87-90855-69-7.

Laborde, D., 2011. Assessing the Land Use Change Consequences of European Biofuel Policies. International Food Policy Institute (IFPRI): Washington, DC, USA.

Lund, H., Hvelplund, F., Mathiesen, B.V., Alberg Østergaard, P., Christensen, P., Connolly, D., Schaltz, E., Pillay, J.R., Nielsen, M.P., Felby, C., Bentsen, N.S., Tonini, D., Astrup, T., Meyer, N.I., Heussen, K., Lind, M., Morthorst, P.E., Andersen, F.M., Münster, M., Hansen, L-L.P., Wenzel, H., Hamelin, L., Munskgaard, J., Karnøe, P., 2011. Coherent energy and environmental system analysis, a strategic research project financed by the Danish Council for strategic research, programme commission on sustainable energy and environment. Department of Development and Planning, Aalborg University, Aalborg, Denmark.

Müller-Wenk, R., Brandão, M., 2010. Climatic impact of land use in LCA - carbon transfers between vegetation/soil and air. Int. J. Life Cycle Ass. 15, 172-182.

NERI, 2010. Emissions from decentralised CHP plants 2007. Environmental project no. 07/1882. Project report 5–Emission factors and emission inventory for decentralised CHP production. National Environmental Research Institute: Aarhus University, Aahrus, Denmark

Searchinger T., Heimlich R., Houghton R.A., Dong F., Elobeid A., Fabiosa J., Tokgoz S., Hayes D., Yu T.-H, 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319, 1238-1240.

Styles, D, Jones, M.B., 2007. Energy crops in Ireland: Quantifying the potential life-cycle greenhouse gas reductions of energy-crop electricity. Biomass Bioenerg. 31, 759-772.

Tyner, W.E., Taheripour, F., Zhuang, Q., Birur, D., Baldos, U., 2010. Land use changes and consequent CO_2 emissions due to US corn ethanol production: A comprehensive analysis. Department of Agricultural Economics: Purdue University, Indiana, USA.

Weidema, B., 2003. Market information in life cycle assessment. Environmental Project no. 863, Danish Ministry of the Environment, Environmental Protection Agency, Copenhagen, Denmark.

A model of indirect land use change

Jannick H Schmidt[1,*], Jürgen Reinhard[2], Bo P Weidema[3]

[1] 2.-0 LCA consultants, Skibbrogade 5, 1, 9000 Aalborg, Denmark
[2] EMPA, Technology and Society Lab, Überlandstrasse 126, Dübendorf, Switzerland
[3] Aalborg University, The Danish Centre for Environmental Assessment, Skibbrogade 5, 1, 9000 Aalborg, Denmark
[*] Corresponding author. E-mail: *js@lca-net.com*

ABSTRACT

Around 9% of global carbon emissions in 2010 originated from deforestation. Often, these emissions are not appropriately addressed in life cycle assessment (LCA). The link between demand for land in one region and deforestation in other regions is referred to as indirect land use changes (iLUC). Existing models for iLUC most often operate with arbitrary amortisation periods to allocate deforestation emissions over time, and the causal link between land occupation and deforestation is generally weekly established. This paper presents an iLUC model where amortisation is avoided by use of IPCC's global warming potential. The causal link between demand for land and land use changes is well established through markets for land. The land use effects include changes in land cover, i.e. deforestation, as well as intensification of land already in use. The presented iLUC model is applicable to all types of land, all crops and in all regions of the world.

Keywords: indirect land use changes, iLUC, consequential, attributional, land use change transition matrix

1. Introduction

According to Peters et al., (2012), around 9% of global carbon emissions in 2010 originated from deforestation. Often, these emissions are not addressed in life cycle assessment (LCA) because the causal link between the use of land and deforestation is not well described and because there is a missing consensus on how to establish this link. Further, several studies suggest that effects from intensification of cropland may be caused by changes in demand for land. In the following a new model and data for establishing the causal link between the use of land and the effects on land use changes and intensification are presented. Here, this link is referred to as indirect land use changes (iLUC).

iLUC is defined as the upstream consequences of the occupation of land, regardless of what you do to it. Whereas indirect land use changes are upstream life cycle impacts of an activity which induces the land use change, direct land use changes take place only in the land transforming activity. It should be noticed that the upstream effects, i.e. deforestation and intensification, of occupying land in one region of the world are likely to take place in other parts of the world.

The model is based on the assumption that current demand for land causes current land use changes. The market for land is defined as a service that supplies capacity for production of biomass. This market has inputs from different suppliers, e.g. land already in use, expansion of land (which may cause deforestation) and intensification. The presented model is applicable to all regions in the world and to all types of land use. The standard reference flow for the use of land, 'land tenure', is the land's potential production capacity, measured as the potential net primary production, NPP_0 (in unit kg carbon). This can easily be converted to occupation in units of hectare years (ha yr), e.g. by use of data in Haberl et al., (2007). The concept is illustrated in Figure 2.

The starting point of the model is an inventory of the total global observed land use changes. The accounting framework for this is a land use change transition matrix, see Table 2. The land use change transition matrix is mainly based on FAO's Global Forest Resources Assessment (FAO 2010). Distinction is made between different markets for land; land suitable for arable cropping, land suitable for forestry, land suitable for rangeland, and other land (barren, deserts, ice caps etc.). The land tenure market activities have four types of inputs: land already in use, expansion, intensification and crop displacement. The emissions related to deforestation and intensification are based on IPCC (2006) and Schmidt (2008) respectively. The inventory framework allows for the application of consequential and attributional modelling assumptions. Time related effects are addressed by use of a fate function of CO_2 pulse emissions in the atmosphere opposed to the amortisation approach which is typically used in iLUC models.

2. Methods

2.1 Goal and scope

The purpose of the presented iLUC model is to provide a life cycle inventory of the upstream consequences of land occupation. These consequences include land transformation as well as intensification. The model is applicable to all types of land in all regions of the world for all types of crops regardless of what they are used for; food, biofuel, fiber or other.

The model is applicable to so-called small scale changes, i.e. changes that do not change overall market trends (Weidema et al., 2009). An example of a large scale change would be if the change under study required a considerable share of the remaining land available for arable cropping so that the ratio between expansion and intensification would be affected. It should be noted that a large scale change would require really large changes in demand for land; more than most current biofuel policies imply. Large scale changes needs to be analysed specifically by use of scenario modelling.

The model only includes long-term changes in supply caused by a change in demand. Hence, short-term effects on prices and subsequent price-elasticity effects are not included. This implies that reduced food consumption as a result of increased demand for land is not included.

2.2 Functional unit

Activities which include occupation of land obviously need a specified area in a specified period of time. This can be measured in hectare years (ha yr). An LCA market activity is defined in order to model this. This activity is called ‘Market for land tenure’. It is the inputs and outputs of the market for land tenure that is the modelling of iLUC. An obvious option for a reference flow of a land tenure activity would be occupation of land (ha yr). However, this approach does not take into account that the potential production on 1 ha yr land in e.g. a dry temperate climate is very different from the potential in wet tropical climate. This could be overcome by operating with a kind of productivity weighted occupation of land. Another option would be the potential net primary production (NPP_0), measured in unit of kg carbon. Since the latter provides a simple way to include land with different productivities, this option is chosen. Thus, the reference flow of the LCA activities that supply land tenure is defined as the contribution to the production capacity. This can be interpreted in parallel to a production facility’s capacity to produce a product. It is up to the current production facility to utilise the capacity, like e.g. a 400 MW power plant has the capacity to produce 400 MW, but most often the installed capacity is not fully utilised. In the case of a land-using activity, it is up to the land-using activity to utilise the land’s capacity to produce biomass. Hence, the functional unit (or reference flow) of the land tenure market activity is defined as of Table 1.

It should be noted that the current iLUC model covers all regions in the world. Some areas have very low or even zero potential net primary production. This is valid for barren land, deserts, ice caps, high mountains etc. Since the productive function of these lands cannot be production of plant material, NPP_0 is not a feasible reference flow. Instead, the reference flow is measured in units of area*time. The environmental impacts from transforming these lands into e.g. urban area or mining sites is not associated with changes on below and above ground carbon (which causes GHG-emissions) and the effect on biodiversity, measured as the absolute number of species affected, is very limited.

Table 1. Functional unit of the land tenure market.

Functional unit
Productive land (plant material): The functional unit is defined in terms of potential net primary production (NPP_0) as 1 kg C. The reference flow of the land tenure market activity is ‘biomass production capacity, measured in kg NPP_0’. Data on potential net primary production (NPP_0) in different parts of the world can be obtained from Haberl et al., (2007, SI figure 2).
Other land: Another functional unit has been used for the land tenure market ‘other land’, which includes barren land, deserts, ice caps, high mountains etc. The functional unit is defined as 1 ha yr, and the reference flow is measured in units of area*time.

2.3 System boundaries, causalities and time perspectives

To understand the concept of iLUC, it must be realised that there are two types of land use change, i.e. direct land use changes (dLUC) and indirect land use changes (iLUC). dLUC is defined as the consequences of what you do to the land that you occupy. These effects take place within the same LCA activity that uses (occupy) the land. iLUC is defined as the upstream consequences of the occupation of land, regardless of what you do to it. These effects have previously been called the competition effect (Lindeijer et al., 2001). The iLUC and dLUC are illustrated in Figure 1. In Figure 1, the area “a” is occupied by the product under study. Fig. 1 shows two situations; left illustrates the situation when the product under study is not demanded/produced, and right illustrates the situation where the product is demanded/produced. The situation in the right side obviously requires some land occupation ("a") in the country where the product is produced. This is illustrated as a_{after}. The difference between a_{before} and a_{after} illustrates the direct land use effects. Often the dLUC only involves replanting of existing arable land, or other shifts between land use types which are already in use. The land use change related emissions of this are most often insignificant. However, drainage of organic soils which is included in dLUC is significant in terms of GHG-emissions. This paper does not focus on dLUC. When a_{before} is transformed to a_{after} the net output of the land in the affected country is reduced from 10 t to 9 t (quantities are just for illustration). To compensate for this reduction, new production capacity is needed. This is illustrated as the transformation from b_{before} to b_{after}. b_{before} represents a situation where the land is not in use. The transformation of the area b often involves deforestation, and this is what is referred to as iLUC. Notice that iLUC is illustrated as transformation of land only; as it is described later in this paper, iLUC also include intensification and crop displacement.

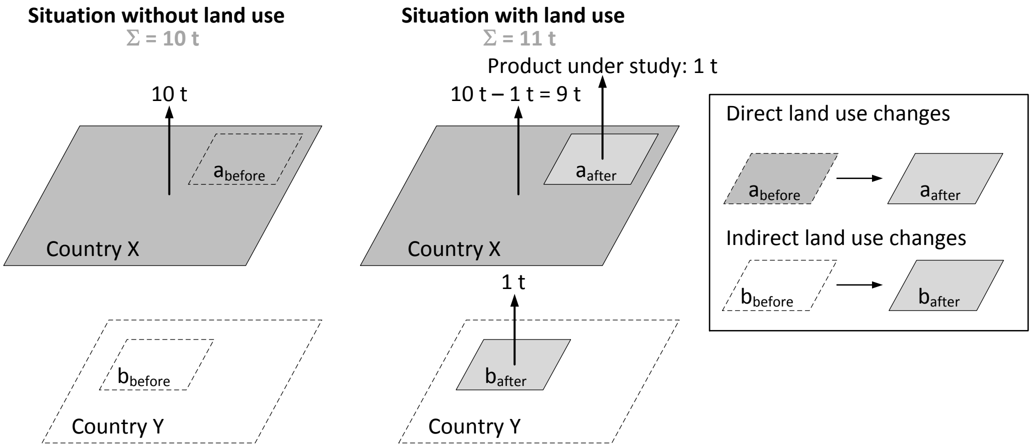


Figure. 1: Illustration of direct and indirect land use changes. The land under study is the area "a". To fully account for land use changes, the sum of direct and indirect land use changes shall be included.

The overall concept of the model is that it is assumed that the current use of land reflects the current demand for land, and that land use changes are caused by changes in demand for land. This concept is equivalent to all other modelling in life cycle inventory, i.e. the demand for a product determines the production volume. The market for land is defined as a service that supplies capacity for production of biomass. This capacity can be obtained from the market, which has inputs from different suppliers, e.g. expansion of land (such as deforestation) and intensification.

The link between LCA activities that occupy land and the land use changes is established via the markets for land, and indirect land use changes are modelled as upstream inputs to the product system. The upstream effects (indirect land use changes) related to the occupation of land are illustrated in Figure 2. Notice that all inflows to the land tenure LCA activity are measured in kg NPP_0 (as kg C). This is explained further in section 2.5.

When the occupation of land causes deforestation, a critical point is often to decide the period of time, the deforestation emissions should be 'amortised' over. The current model does not operate with amortisation. If only expansion is considered, occupation of 1 ha in 1 year will cause 1 ha deforestation. After the duration of 1 yr, the land is given over to other crops, which can then be grown without deforestation. Hence, the occupation of 1 ha yr is modelled as 1 ha deforestation in year 0 and -1 ha deforestation in year 1. The Bern Cycle, which represents the fate of a CO_2 pulse emission and which is used to calculate the global warming potential (GWP100), is used to calculate the GWP100 associated with speeding up deforestation as referred to above.

The model enables for operating with consequential and attributional modelling assumptions. The difference between the two approaches is here defined as; attributional modelling includes the average of all suppliers to a market for land whereas consequential modelling excludes constrained suppliers. Therefore, the difference is that the consequential inventory only includes inputs of expansion and intensification (see Figure 2), whereas the attributional inventory also includes inputs from land already in use.

2.4 Segmentation of the land tenure market, five land markets

The relevant functions of land to be modelled is the land's ability to produce products, i.e. crops, grass and wood, and the function to provide area for human structures, i.e. buildings, infrastructure and production facilities such as mines. When forests and human structures occupy land suitable for agriculture, it will have similar land use related effects as when crops are grown, because it is related to the acquisition of the same type of land. The following five distinct markets for land are considered (All land tenure markets can be used for urban, industrial or infrastructure area):

- Extensive forest land: not fit for more intensive forestry (e.g. clear cutting and reforestation), e.g. because it is too hilly, too remote, or it is growing on very infertile land making intensive forestry uneconomic. Forests grown on extensive forestland are typically harvest after natural regrowth with mixed species.
- Intensive forest land: fit for intensive forestry (e.g. clear cutting, reforestation, species control etc.), but not fit for arable cultivation because the soil cannot be treated for arable cultivation, e.g. because the soil is too rocky. Forests grown on intensive forestland may be managed as intensive or extensive forestry. Intensive forest land may also be used for other land use, e.g. livestock grassing and extensive forestry.
- Arable land: fit for arable cultivation (annual crops and perennial crops). Arable land may be used for cultivation of annual or perennial crops, for intensive or extensive forestry, and pasture.
- Rangeland: too dry for forestry and arable cultivation. Therefore, when in use, rangeland is most often used for livestock grassing.
- other land: not fit for biomass production; barren land, deserts, ice caps, high mountains etc.

It should be noted that the above mentioned land markets are regarded as separate markets for land, i.e. acquisition for a specific type of land is not associated with upstream effects on the other land use types. This is also the reason that the term 'not fit for agriculture' is added to forest land; if forest is grown on agricultural land then the upstream effects will be caused by the effect on the crop cultivation on this land.

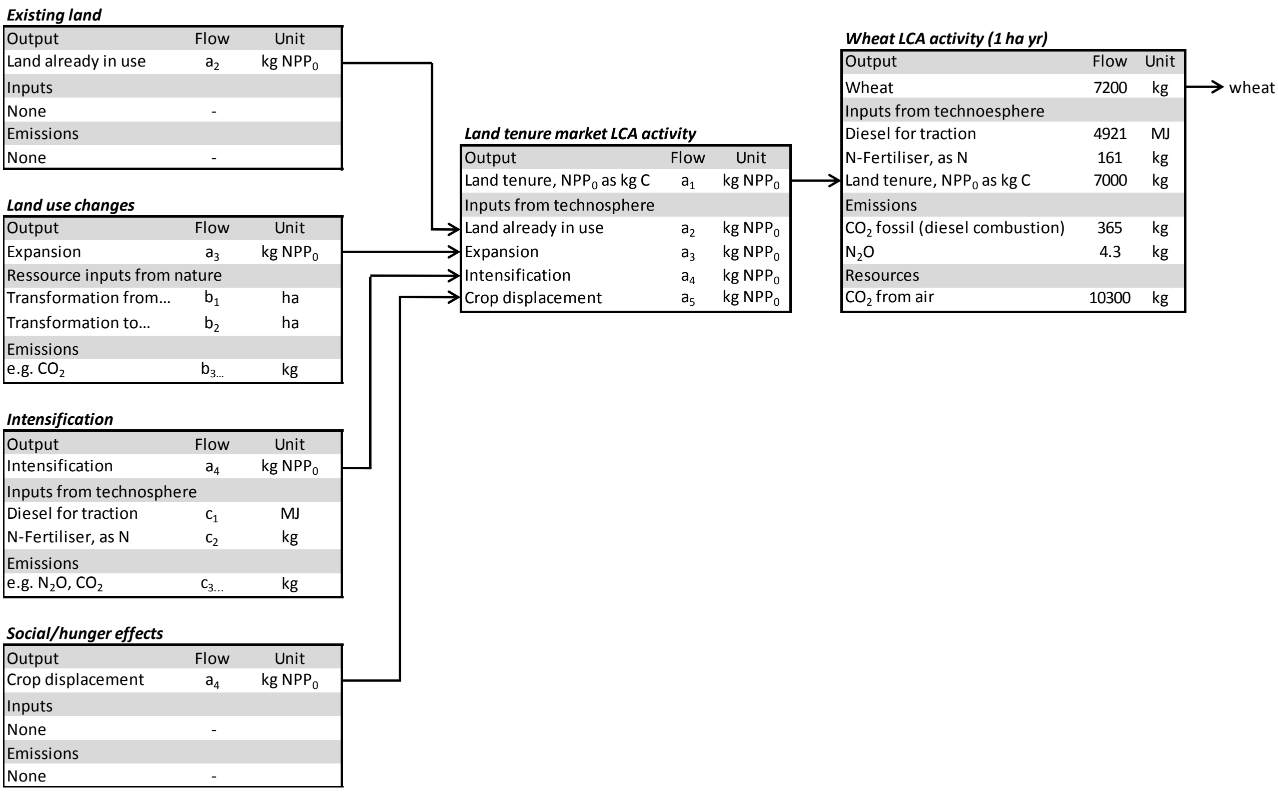


Figure 2. Illustration of the land tenure market activity and it's inputs and outputs. An agricultural activity, here wheat, has inputs from the land tenure market activity. The land tenure market activity has inputs from four different supplies of biomass production capacity. Each of these suppliers is associated with emissions. The sum of these emissions is referred to as iLUC emissions.

2.5 Life cycle inventory data

As illustrated in Figure 1, the land tenure market LCA activity has inputs of land already in use, expansion, intensification, and crop displacement. The LCA activity 'Land already in use' is not associated with any inputs or emissions because this land's production capacity is already established, and because maintenance of this capacity is inventoried as emissions belonging to the agricultural activity that occupies the land. The activity 'Expansion' has emissions of CO_2 and N_2O associated with land transformation (most often deforestation). The emissions are determined based on the land use change transition matrix presented in Table 2 combined with data on above and below ground carbon stocks in IPCC (2006, chapter 4, 5 and 6) and N_2O model in IPCC (2006, chapter 11). By-products as timber and emissions related to diesel etc. in the deforestation activity are not included in the current version of the iLUC model. The activity 'Intensification' has inputs of fertilisers and traction, and emissions associated with the fertiliser. The inventory data of intensification are described further in Schmidt (2008). The activity 'Crop displacement' (not included in the iLUC model) has no inputs or emissions. If the activity was included, inventory items related to social impacts should be considered.

All inflows to the land market tenure activity are measured in kg NPP_0 (as kg carbon). The NPP_0 from land already in use and expansion are determined based on general NPP_0 per ha yr figures (Haberl et al., 2007) and figures on total area of land already in use and annual deforestation (FAO 2010; FAOSTAT 2011). The NPP_0 from intensification is calculated as the carbon in crop produced via intensification during one year. The intensification is determined based on crop dose-response figures for fertiliser input (Schmidt 2008) combined with information on which crops and where intensification takes place (data from FAOSTAT 2011) and current fertiliser levels for these crops (IFA 2011).

Table 2. Land use change matrix, unit: million ha. The global land use transition matrix is established for an average year in 2000-2010. The top column headings divide the total land into land not in use and land in use. For the land in use there are four land tenure markets. The growth of these markets, which involves deforestation and land degradation, can be seen as inputs in the rows. (Schmidt et al., 2012)

Transformation to:	**Non use**			**Markets**				
	Primary forest	Secondary forest	Other (grassland, wetland and scrubland)	Extensive forest land	Intensive forest land	Arable land	Rangeland	**Total land use ref. year**
Transformation from:								
Primary forest	1,102	0	0	1.09	0.084	3.02	0	**1,106**
Secondary forest	0.34	1,798	0	0	4.85	9.98	0	**1,813**
Other (grassland, wetland and scrubland)	0	1.30	3,769	0	0	0.60	1.88	**3,773**
Extensive forest	0	0	0	930	0	0	0	**930**
Intensive forest	0	0	0	0	196	0	0	**196**
Arable	0	0	0	0	0	1,624	0	**1,624**
Range	0	0	0	0	0	0	3,569	**3,569**
Total land use ref. year + 1	**1,102**	**1,799**	**3,769**	**931**	**201**	**1,638**	**3,571**	**13,012**

3. Results

Table 3 summarizes some results calculated with the iLUC model. The first column shows the affected region/country, the second column converts 1 ha yr to NPP_0. The last three columns presents the results divided into contributions from expansion and intensification.

Table 3. Results for the occupation of 1 ha yr of different land tenure markets (for global average). Further, in the lower part of the table example results are shown for arable land and rangeland in different countries in the world. Results are shown when applying the consequential modelling assumption.

Affected land market	NPP_0, kg C ha-1 yr-1	Result, consequential Expansion, kg CO_2-eq.	Intensification, kg CO_2-eq.	Total, kg CO_2-eq.
Global averages for 1 ha yr				
Arable	6,110	1,010	6,820	**7,830**
Forest int.	7,220	3,340	-	**3,340**
Forest ext.	7,220	1,670	-	**1,670**
Range	4,860	1,160	-	**1,160**
Examples of country specific results for 1 ha yr				
Arable, Malaysia	12,000	2,000	13,400	**15,400**
Arable, Denmark	7,000	1,170	7,800	**8,970**
Arable, Brazil Amazonas	9,000	1,500	10,000	**11,500**
Range, Brazil Cerrado savannah	5,000	1,190	-	**1,190**

Table 3 only shows results using the consequential modelling assumption. If the attributional modelling assumption is applied, the results for arable land will be around 4% of the results for consequential. Similarly, the results for forestland and rangeland are significantly lower. The reason for this is that in the attributional modelling a large market share of the inputs to the land tenure market is comprised by 'land already in use' which is not associated with any emissions.

4. Discussion and conclusion

This paper presents a model of iLUC. The model has the following characteristics; it includes effects on deforestation and intensification, allocation of deforestation emissions using a chosen amortisation period is avoided, the model is applicable to any land use type in any location, and the model can handle both consequential and attributional modelling assumptions.

The results show significant iLUC effects on global warming; crops with yields 5-7 t/ha yr grown on average yielding land will be associated with iLUC effects at 1 to 1.5 kg CO_2-eq./kg crop. For many arable crops this means a factor 2-4 increase in GHG-emissions. For arable land the effect from intensification comprises around 87% of the total iLUC where the remaining is related to deforestation. The iLUC from forestry and rangeland are also significant. Here no intensification is considered. For forestland, the emissions are caused by transformation of natural forest to cultivated forest, and for rangeland, the emissions originate from transformation of natural savannah/scrub to grassland.

The major uncertainties in the model are a) the market for land is assumed to be global, b) the determination of NPP_0 as of Haberl et al., (2007) is relative coarse grained, so 10-15% uncertainty is associated with reading maps which only show intervals, c) the determination of the relative distribution of inputs of expansion and intensification to the land tenure market is associated with uncertainties since it is based on indirect estimation methods, d) the modelling of intensification as of Schmidt (2008) only considers additional NPK fertiliser as an affected mean of intensification, and e) also the dose-response effects in Schmidt (2008) of crop-yields relative to additional fertiliser are associated with uncertainties.

Despite the uncertainties listed above, the model results are regarded as being within reasonable uncertainty intervals. It should be noticed, that the model concept is open for improved data input at almost any level. This means that the presented model can be seen as a proposal for solving the conceptual challenges with the modelling of iLUC, and that future improvements of the model will focus on improving input data.

6. References

FAOSTAT, 2011. FAOSTAT, Food and agriculture organization of the United Nations. Accessed November 2011: http://faostat.fao.org/

FAO, 2010. Global Forest Resources Assessment 2010 – main report. FAO, Rome

Haberl H., Heinz Erb K., Krausmann F., Gaube V., Bondeau A., Plutzar C., Gingrich S., Lucht W. and Fischer-Kowalski M., 2007. Quantifying and mapping the human appropriation of net primary production in earths terrestrial ecosystems. PNAS, vol. 104, no. 31, pp 12942–12947

IFA, 2011. Production and trade statistics. International Fertilizer Industry Association (IFA). Accessed November 2011: http://www.fertilizer.org/ifa/Home-Page/STATISTICS

IPCC, 2006. 2006 IPCC Guidelines for national greenhouse gas inventories, Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H S et al., (eds). IGES, Japan.

Peters G. P., Marland G., Le Quéré C., Boden T., Canadell J. G., Raupach M. R. (2011). Rapid growth in CO_2 emissions after the 2008-2009 global financial crisis. Nature Climate Change 2, 2–4

Lindeijer E., Müller-Wenk R., Steen B., with written contributions from, Baitz M., Broers J., Finnveden G., ten Houten M., Köllner T., May J., Mila i Canals L., Renner I. and Weidema B., 2001. Impact Assessment of resources and land use. SETAC WIA-2 taskforce on resources and land

Schmidt J. H., 2008. System delimitation in agricultural consequential LCA, outline of methodology and illustrative case study of wheat in Denmark. Int J Life Cycle Assess 13(4):350–364

Schmidt J. H., Reinhard J., and Weidema B. P., 2012. Modelling of indirect land use changes in LCA, report v3. 2.-0 LCA consultants, Aalborg, http://www.lca-net.com/projects/iLUC_model/

Weidema B. P., Ekvall T., Heijungs R., 2009. Guidelines for applications of deepened and broadened LCA. Deliverable D18 of work package 5 of the CALCAS project.
http://fr1.estis.net/includes/file.asp?site=calcas&file=7F2938F9-09CD-409F-9D70-767169EC8AA9

Weidema B. P., Bauer C., Hischier R., Mutel C., Nemecek T., Vadenbo C. O., Wernet G. (2011). Overview and methodology. Data quality guideline for the ecoinvent database version 3. Ecoinvent Report 1(v3). St. Gallen: The ecoinvent Centre

Land use change - GHG emissions from food and feedstuffs

Stefan Hörtenhuber[1,2,*], Michaela Theurl[1], Thomas Lindenthal[1,3],Werner Zollitsch[2]

[1] Research Institute of Organic Agriculture (FiBL) Austria
[2] University of Natural Resources and Life Sciences Vienna, Department of Sustainable Agricultural Systems
[3] University of Natural Resources and Life Sciences Vienna, Centre for Global Change and Sustainability
[*] Corresponding author. E-mail: *stefan.hoertenhuber@fibl.org*

ABSTRACT

Land use change (LUC) is assumed to be a major contributor to global CO_2 emissions. Despite its great impact on global greenhouse gas (GHG) emissions, LUC is hardly incorporated into GHG estimations for food and feedstuffs for various reasons related to methodological limitations. This contribution outlines a method which can be used to derive direct and indirect emissions from LUC on the national level for specific crops within a given accounting period for allocation. LUC-related emissions are restricted to physically occurring fluxes; ten or 20 years are identified as suitable accounting periods for overall LUC-related emissions. Primarily, this contribution compares effects of different methods on the impact of emissions from LUC on the overall results for products' GHG emissions. As a result, we suggest the use of methods which allow for a direct, product-related allocation as this is probably the only way to quantify mitigation strategies for LUC-related GHG emissions.

Keywords: land use change, greenhouse gas emission, GHG, food, feedstuffs

1. Introduction

Land use change (LUC) and – to a much lesser extent – land use (LU) are assumed to be one of the major contributors to global CO_2 emissions, contributing 23% to the increase in atmospheric CO_2 concentration during the last 250 years (Hörtenhuber et al., 2011a). Accordingly, studies which quantify emissions from the production of food and feedstuffs or bio-energy will underestimate the increase of the CO_2 concentration in the atmosphere by more than 20% on average unless they account for relevant emissions occurring from LUC and LU (LULUC). Despite the great impact on global greenhouse gas (GHG) emissions and thus on global warming, LULUC is hardly incorporated into estimations of the global warming potential (GWP) in life cycle assessments and into current carbon footprints (CF) in previous studies dealing with the production of food and feedstuffs. Various methodological problems are among the most important reasons for this. Literature sources link methodological limitations to lacking information on aspects such as regions of origin for imported food and feedstuffs, the nature of land before conversion, the onset of LUC-related emissions and their temporal progression as well as debatable accounting periods for depreciation of LUC-emissions (see e.g. Dalgaard et al., 2008).

This paper aims at contributing to closing methodological gaps and particularly to deriving reasonable accounting periods in order to enable a strictly product-related inclusion of emissions from LULUC Primarily, results for product-related GHG emissions from LULUC are compared to results derived with other methods and to a basic variant, which does not account for LUC-related emissions.

2. Methods

For the estimation of GHG emissions, information on characteristics of supply chains and on specified regions of origin of raw materials is often available on a regional scale. Therefore, a product- and region-related approach should be applicable for emissions occurring from the supply chains of most products.

Concerning a methodology which enables a product-related inclusion of emissions from LUC (see Hörtenhuber et al., 2011a), information was derived from modelling and a literature review. It was concluded that system boundaries should be defined broadly in the estimation of GHG-emissions from agricultural production. CO_2-neutrality for emissions from LUC exists only theoretically, as the storage of released carbon occurs over substantial time periods and not necessarily in spatial proximity. Hence, besides fossil CO_2 emissions, CO_2 from the degradation of above-ground biomass plus soil organic carbon from LUC and LU (including below-ground biomass from cleared vegetation) is assessed herein.

Hörtenhuber et al., (2011a) suggested that GHG emissions which CF and GWP accounted for should be restricted to physically occurring fluxes of GHGs which are related to a specific product. Consequently, we exclude hypothetical or prospective fluxes such as a 'loss of sink function' (Kool et al., 2009), which may be imposed to quantify a farmlands' effect of reduced sink capacity as compared to natural vegetation. Likewise, we do not apply a concept of land use impact to our product-specific calculation of LUC emissions, as was described in Milà i Canals et al., (2007) and Müller-Wenk and Brandão (2010). The main reason for this is that this concept also uses hypothetical pathways for emission fluxes from soil organic carbon (SOC). The method applied herein and described in Hörtenhuber et al., (2011a) includes estimated emissions from LUC which were directly allocated to a specific crop, based on the area additionally cultivated with the respective crop relative to the overall expansion of agricultural land. It also includes estimated indirect LUC effects for the replaced crops with an allocation based on expansion rates of replaced crops relative to the respective national agricultural area (see also Ponsioen and Blonk, 2012).

Accounting (depreciation) periods for LUC can be either derived from isolated areas with LUC-emission or from a global carbon view. For derivation of accounting periods, information on soil carbon cycling was collected during a literature review; in order to obtain sound accounting periods which reflect the dynamics in atmospheric carbon cycles, Hörtenhuber et al., (2011a) simulated the development of CO_2 concentration by using elements of an established method (Bern Carbon cycle model, Bern2.5CC; Joos et al., 2001). Hörtenhuber et al., (2011a) suggested that 10 or 20 years may be used as suitable default values for accounting periods for overall LUC-related emissions. The 10 year period includes the majority of LUC-related emissions released from biomass and soils as well as of emissions remaining in the atmosphere, especially for tropical regions with quicker processes. Twenty years are more feasible for temperate conditions with their lower rates of CO_2-release (Hörtenhuber et al., 2011a).

The productive period of farmland which originated from LUC is usually greater than one year, probably at least within the magnitude of the duration which is needed to find a new equilibrium state of soil organic carbon. Consequently and for the sake of simplification, we allocate CO_2 which is rapidly released from cleared biomass not only to the crops of the first year following LUC, but to the same time period accounted for CO_2 emissions from soil after LUC (see Hörtenhuber et al., 2011a). This constitutes an exemption to our definition of restricting GHG accounting to physical fluxes. This is nevertheless justifiable, as it does not create a new accounting period and is also in line with most CF standards (e.g. BSI, 2011). Thereby, punctually emitted fluxes from above-ground biomass are distributed over a representative period of land use by allocating the sum of emissions from cleared above- and below-ground biomass together with SOC losses to the overall emitting period.

For exemplary model calculations following different approaches concerning LUC (see Fig. 1), a 20 years period was used. GHG emission factors for the applied 20 years accounting period are derived from Hörtenhuber et al., (2011b; Table 1). Emission from LU are estimated according to Hörtenhuber et al., (2010) for Austrian average barley and based on Hörtenhuber et al., (2011a) for an average of Brazilian soybeans (no specifical value for certified soybeans; for certification criteria see ProForest, 2004).

For classification and comparison of our product-specific LUC results, we use average global LUC emissions according to Vellinga et al., (2011) and values for LUC (land transformation) and LU (land occupation) derived from the land use impact method by Müller-Wenk and Brandão (2010), both presented in Table 1.

Vellinga et al., (2011) calculated global CO_2-eq emission from LUC (5.8 Gt per year) per total agricultural land used ($4.9 * 10^9$ hectare). This translates into 1,180 kg CO_2-eq per ha and per year or 1.18 kg CO_2-eq per kg crop if yield is 1,000 kg per ha.

For comparison of the method presented herein with the method of Müller-Wenk and Brandão (2010), we introduced a few assumptions to the latter, which are based on Hörtenhuber et al., (2011b): emissions from land transformation (LUC) are evaluated only for areas converted within the last 20 years with 39% and 10% from tropical forest, 9% and 2% from savannah for average conventional and for certified Brazilian soybeans, respectively (Hörtenhuber et al., 2011b). For Austrian barley this land transformation can be neglected (Hörtenhuber et al., 2010, 2011b). Emissions from land occupation (defined as land use, which prevents atmospheric CO_2 to be sequestrated again) are fully counted as described in Müller-Wenk and Brandão (2010). The land occupation results following Müller-Wenk and Brandão (2010) in Table 1 take into account that soybean production is partially taking place on previous tropical forest land and partially in previous grassland areas (we assumed three quarters of overall farmland for soybeans coming from savannahs/grasslands, one quarter from tropical forests, but no difference between average Brazilian and certified soybeans). For the case of previous (Brazilian) tropical forest areas converted to cropland, Müller-Wenk and Brandão (2010) calculated 46.2 tons of C emitted from land transformation (LUC) over the whole time series, resulting in 0.961 kg C per kg dry matter (DM) soybeans (with the assumption of a 20 years allocation period). Analogous, for (Brazilian) crop land converted from tropical savannahs, they reported 18.0 tons of C from LUC (0.374 kg C per kg DM soybean, assuming a 20 years period). Additionally, Müller-Wenk and Brandão (2010) reported 1.48 and 0.37 tons C for land occupation (LU) per ha and year from previous tropical forest and grassland, respectively.

Table 1. GHG emissions from LUC and LU assuming either (i) average global emissions (Vellinga et al., 2011), (ii) specific conditions for specific products and regions (based on Hörtenhuber et al., 2010, 2011b) or as (iii) emissions from a land use impact method derived from Müller-Wenk and Brandão (2010).

kg CO_2-eq per kg DM feedstuff	Austrian barley	Certified Brazilian solvent-extracted soybean meal	Average Brazilian solvent-extracted soybean meal
(i) global average LU/LUC emissions related to grain yields	0.246	0.389	0.389
(ii) regionally and product specified GHG emissions from LUC	0	1.245	4.150
(ii) regionally and product specified GHG emissions from LU	0.030	0.119	0.119
(iii) land use impact emissions related to land transformation (LUC)	0	0.380	1.498
(iii) land use impact emissions related to land occupation (LU)	0.378	0.988	0.988

3. Results

To quantify the impact of LUC-related emissions on overall global warming potential for food and feedstuffs, emissions from LUC were included into emissions for other categories, such as use of fuels, industrial processing, production of mineral fertilisers and pesticides as well as direct and indirect N_2O emissions from soil (Hörtenhuber et al., 2011b). This has been done for three different cases, for which results are presented in Figure 1. Austrian barley, certified Brazilian extracted soybean meal (SBME) and an Austrian national average of SBME imported mainly from Brazil. Based on that, GHG emissions from land use change are partially included: (a) does not include any LULUC-related GHG emissions; (b) accounts for LULUC-emissions as derived from average global emissions (Vellinga et al., 2011; see (i) in Table 1); (c) accounts for product-related LULUC emissions (see (ii) in Table 1); (d) represents emissions from a land use impact method (Müller-Wenk and Brandão, 2010; see (iii) in Table 1).

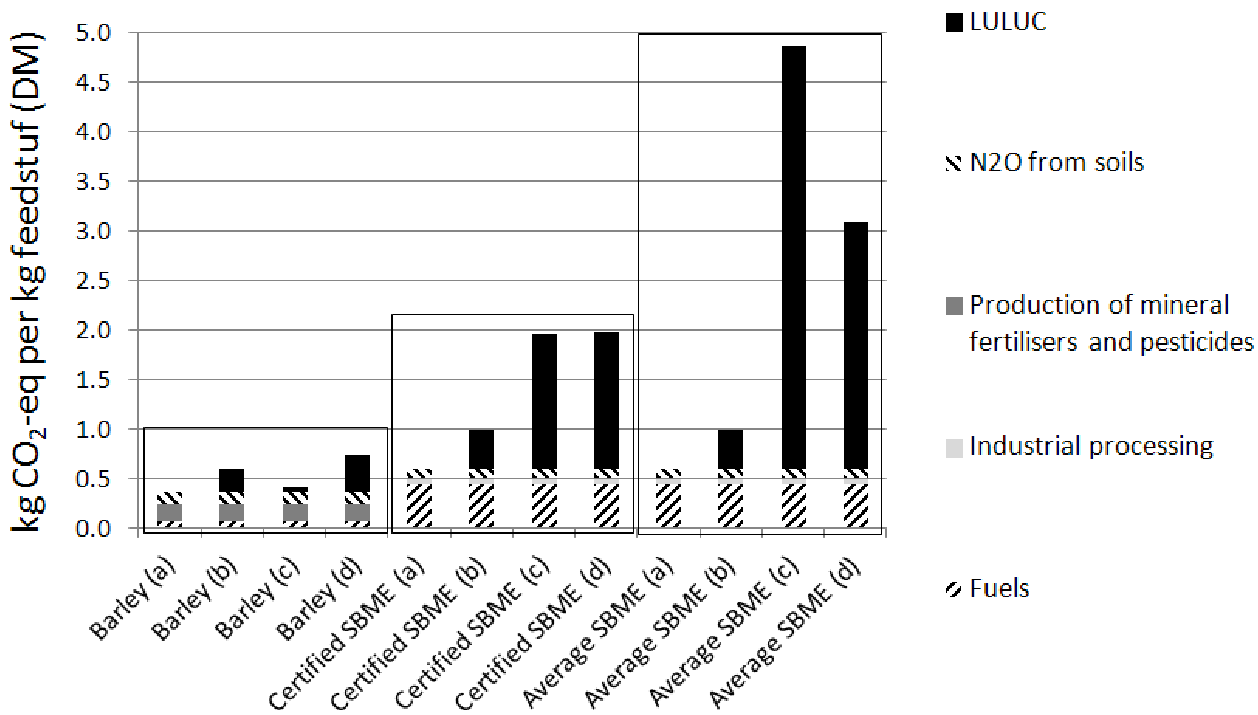


Figure 1. Overall GHG emissions for feed ingredients: (a) without any LULUC-related GHG emissions; (b) LULUC-emissions derived from average global emissions; (c) product-related LULUC emissions: (d) emissions following a land use impact method.

4. Discussion

In reality, physically occurring carbon fluxes follow a logarithmic function with high initial rates of losses which are levelling off after a certain number of years after LUC. Although in Figure 1 the case of certified

SBME shows similar results for the variants (c) 'product-related LUC emissions' and (d) 'emissions from a land use impact method', the results show great differences, when differentiated for LU and LUC. The reason for the differences is that in variant (d) according to Müller-Wenk and Brandão (2010) low LUC-related emissions go along with high rates of carbon losses from LU. From our point of view, this constitutes an underestimation of emissions from land transformation (LUC), while overestimating the emission effect of occupied land by assuming that agricultural land use prevents atmospheric carbon from being sequestrated in terrestrial sinks again. In fact, a notable part of the huge amount of CO_2 emitted after a LUC is sequestrated in natural ('residual') terrestrial sinks, such as natural tropical forests (Denman et al., 2007) and does not stay in the atmosphere until the occupation of the land is abandoned.

According to Vellinga et al., (2011), emissions from LUC should be derived from average global emissions. The authors describe LUC emissions from single crops as useless and that direct and indirect LUC cannot be separated, as total human consumption of all commodities is the driver for LUC. Contrarily, our approach differentiates direct and indirect emissions, as direct and national indirect LUC emissions should be applied to specific crops, which expanded in specific regions. A comparison of our product-related LUC emissions to global average LUC emissions (Vellinga et al., 2011) shows a difference in the same order of magnitude as are described for other GHG emissions (from soil, fuels, etc.). On the one hand, for the case of European grains (e.g. barley in Fig. 1) the inclusion of globally derived LUC values leads to an overestimation of GHG emissions from LUC on average and especially in areas, where LUC-related CO_2-fluxes had occurred long ago. For this case, on-going changes in SOC, i.e. emission or sequestration due to LU, should be reported, but in reality hardly any great losses of SOC can be reported as they usually occur only after LUC. On the other hand, the global average LUC emissions could underestimate LUC-related CO_2 emissions if huge amounts of CO_2 are emitted after a LUC and before a new equilibrium in SOC has been established.

Furthermore, Vellinga et al., (2011) characterise the land use per unit of crop harvested as a key item and promote their method as being simple and robust while avoiding problems of double counting and displacement. They refer to an adequate description of their method's results concerning efficiency (i.e. low yields or bad feed conversion show high LUC emission). From a global point of view, efficiency of land use (described by yield per ha) is certainly partially indirectly related to LUC, but on the other side the land demand of a specific product from a specific region occupying an above-average area of land per product unit is not automatically related to LUC within a relevant period. In contrast to Vellinga et al., (2011), we assume that indirect LUC is not only connected to a low productivity per unit of land used, but also to market dynamics which influence land demand, as do changes in consumption habits and the growing demand for bioenergy and additional drivers.

The most debated concern about the use of direct LUC emission methods (e.g. applied in BSI, 20011 is the derivation of a suitable accounting period (also termed depreciation or amortisation period; see Ponsioen and Blonk, 2012). Herein, we describe an answer to this controversial issue, which is based on soil-related and atmospheric carbon cycles. Hörtenhuber et al., (2011a) suggested two default periods for LUC-related accounting: 10 and 20 years. For the Brazilian soybeans used as feedstuff, LULUC-related emissions will decrease by about 23% if 10 instead of 20 years are implemented, mainly due to a reduced prevalence of soybean production on newly deforested land. Short accounting periods would lead to the phenomenon that converted areas are relieved from LUC-related emissions after a short time period. This would lead to indicating a 'cleaner' production after LUC-related emissions had peaked markedly, although the majority of the released CO_2 still remained in the atmosphere. The estimation of product-specific emissions from LULUC which includes substantially longer accounting periods than used herein, would result in lower LUC-related emission loads for e.g. feedstuffs imported from Latin-America or from other regions with a high share of land recently converted.

Due to its high global relevance, emissions from LUC need to be included in the estimation of CF and GWP, whenever great quantities of (LU)LUC-burdened inputs, such as extracted soybean meal, are used. This can be illustrated for European meat production systems, which rely on high quantities of imported LULUC-burdened inputs; for example, the substitution of feedstuffs loaded with LUC-related emissions (particularly extracted soybean meal) would lead to an estimated 50% less CO_2-eq per kg of broiler carcass (Hörtenhuber et al., 2011a).

5. Conclusion

We suggest using methods for the estimation of LUC-related emissions which allow for a direct, product-related allocation. This is probably the only way to quantify mitigation strategies for LUC-related GHG emissions, even if they do not integrate indirect transnational LUC-effects. To our opinion methods for calculating LUC emissions should be based on physically occurring fluxes. Despite our approach of estimating

LULUC-related GHG emissions only if they can be allocated directly to a product and thereby neglecting indirect LUC, the difficult task of including transnational iLUC should be addressed in future studies.

6. References

BSI, 2011. PAS 2050:2011. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standard Institute, London, UK..

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue W.A, 2008. LCA of Soybean Meal. International Journal of LCA 13 (3):240-254.

Denman, K.L., Brasseur, G., Chidhaisong, A., Ciais, P., Cox, P.M., Dickinson, R.E., Hauglustaine, D., Heinze, C., Holland, E., Jacob, D., Lohmann, U., Ramachandran, S., da Silva Dias, P.L., Wofsy, S.C., Zhang, X., 2007. Couplings Between Changes in the Climate System and Biogeochemistry. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. and Miller, H.L., (ed.), Climate Change 2007: The Physical Science Basis, Contribution of the Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK and New York, USA.

Hörtenhuber, S., Lindenthal, T., Amon, B., Markut, T., Kirner, L., Zollitsch, W., 2010. Greenhouse gas emissions from selected Austrian dairy production systems: model calculations considering the effects of land use change. Renewable Agriculture and Food Systems 25 (4), 316-329.

Hörtenhuber S., 2011: Greenhouse gas emissions from dairy production – assessment and effects of important drivers. Dissertation thesis. University of Natural Resources and Life Sciences, Vienna, Austria.

Hörtenhuber, S., Lindenthal, T., Zollitsch, W., 2011b. Reduction of greenhouse gas emissions from feed supply chains by utilizing regionally produced protein sources - the case of Austrian dairy production. Journal of the Science of Food and Agriculture 91, 1118-1127.

Joos, F., Prentice, C., Sitch, S., Meyer, R., Hooss, G., Plattner, G.-K., Gerber, S., Hasselmann, K., 2001. Global warming feedbacks on terrestrial carbon uptake under the Intergovernmental Panel on Climate Change (IPCC) emission scenarios. Global Biogeochemical Cycles 15 (4), 891-907.

Kool, A., Blonk, H., Ponsioen, T., Sukkel, W., Vermeer, H., de Vries, J., Hoste, R., 2009. Carbon Footprints of conventional and organic pork: Assessment of typical production systems in the Netherlands, Denmark, England and Germany. Blonk Milieu Advies BV, Gouda, The Netherlands.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth-Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007. Key elements in a framework for land use impact assessment in LCA. International Journal of LCA 12 (1), 5-15.

Müller-Wenk, R., Brandão M., 2010. Climatic impact of land use in LCA – carbon transfers between vegetation/soil and air. International Journal of LCA 15 (2), 172-182.

Ponsioen, T.C., Blonk, T.J., 2012. Calculating land use change in carbon footprints of agricultural products as an impact of current land use. Journal of Cleaner Production 28, 120-126.

ProForest, 2004. The Basel Criteria for Responsible Soy Production. Available at http://www.proforest.net/objects/publications/basel-criteria-august-04.pdf, 2012-05-25.

Vellinga, T., Blonk, H., 2011. Feed database for livestock systems - An improvement of CFP calculations for livestock systems. [http://ebookbrowse.com/theun-vellinga-fao-pdf-d92065125, 2012-02-08]

LCA of sunflower oil addressing alternative land use change scenarios and practices

Filipa Figueiredo, Érica Geraldes Castanheira, Fausto Freire*

ADAI-LAETA, Department of Mechanical Engineering, University of Coimbra, Pólo II Campus, Rua Luís Reis Santos, 3030-788 Coimbra, Portugal
* Corresponding author. E-mail: *fausto.freire@dem.uc.pt*

ABSTRACT

Sunflower oil is one of the leading food oils, recently also used for biodiesel production, mainly in southern European countries. This paper presents a Life-Cycle Assessment of sunflower oil produced in Portugal. Two alternative agriculture practices (irrigated and non-irrigated cultivation) were assessed. Twenty-eight alternative land use change (LUC) scenarios were studied (combining four actual land use types and seven scenarios for the reference land use). Life-cycle Impact Assessment results were calculated (ReCiPe method) for six impact categories. Sunflower cultivated on non-irrigated land had higher environmental impacts in 4 categories because of the low productivity, while sunflower cultivated in irrigated land had higher impacts in only 2 categories (due to the use of fertilisers). Cultivation is the main contributor to the life-cycle impacts in all categories. A huge variation in greenhouse gas intensity for sunflower oil was calculated (0.3-20.9 kg CO_{2eq}/kg_{oil}). The results show the importance of LUC and cultivation practices in the environmental performance of sunflower oil.

Keywords: agriculture practices, greenhouse gas, irrigation, land use change (LUC), sunflower oil

1. Introduction

Sunflower is one of the leading oilseed crops used for the production of oil for human consumption. It has also been considered an important crop for biodiesel production in southern European countries (Kallivrossis, 2002). Life-cycle Assessment (LCA) was employed to assess sunflower oil; most of the studies were in the scope of biodiesel production (JEC, 2008, Kallivrossis, 2002, Requena et al., 2010, Tsoutsos et al., 2010). Other LCA studies focused on sunflower cultivation (Cotana et al., 2010) and the use of sunflower oil in agricultural tractors on Greek farms (Balafoutis et al., 2010). However, only one of these LCA studies (Iriarte et al., 2010) addressed land use change (LUC).

LUC is an emergent topic with important implications in terms of the greenhouse gas (GHG) balance of food and bioenergy crops, as demonstrated by several LCA studies for vegetable oils and oil-based biodiesel systems, which have concluded that the GHG intensity is sensitive to the type of LUC. For example, Castanheira and Freire (2011a;b) evaluated LUC for soybean produced in Latin America and palm oil produced in Colombia, calculating a large variation in GHG intensity between different LUC scenarios. Lechon et al., (2011) studied alternative biofuel feedstocks (soybean, palm, rapeseed, sunflower oil), concluding that when LUC impacts are considered the benefits of biofuels are significantly reduced and can even be negative. Malça and Freire (2011) assessed the implications of LUC for bioethanol produced from wheat, and Malça and Freire (2010) assessed rapeseed oil, both produced in Europe, concluding that GHG emissions due to LUC dominate results and have high uncertainty.

Increasing prices of food products together with the expansion of biodiesel produced from vegetable oils in Europe may lead to an increase in the production of sunflower in Portugal, which can be achieved by the expansion of sunflower plantation area (extensification) or by an increase in the productivity (intensification). This motivates assessing the environmental impacts of sunflower oil produced in Portugal, including the carbon-stock changes caused by alternative LUC scenarios. The main objective of this paper is to present an LCA of sunflower oil produced in Portugal.

2. Life-Cycle Model and Inventory

A life-cycle model and inventory for sunflower oil produced in Portugal were developed and implemented. The life-cycle model includes the land use conversion necessary to establish sunflower cultivation, cultivation, transportation, oil extraction and treatment (Fig. 1). For sunflower cultivation, two systems were considered: irrigated and non-irrigated. The infrastructure for facilities, machines and vehicles was included (even that not presented in Fig. 1). The functional unit chosen was 1 kg of oil. The oil chain is multifunctional, with one co-product (sunflower meal). According to ISO 14044 (2006), whenever several allocation approaches seem applicable, a sensitivity analysis shall be conducted to illustrate how different methods change the results. In this study, three allocation methods were analysed: mass, energy and economic.

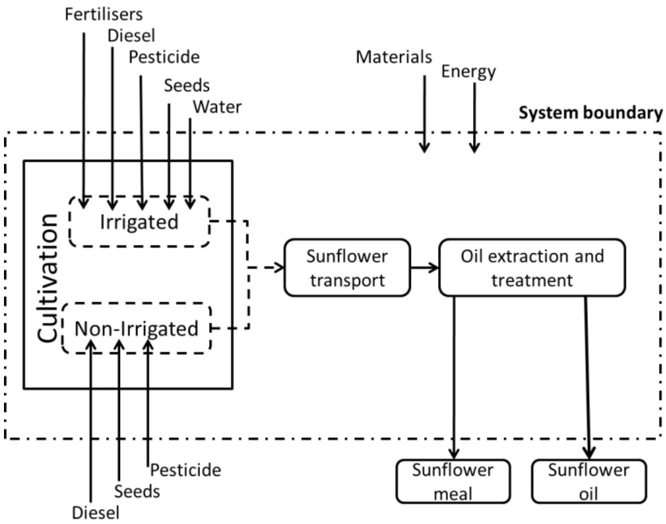


Figure 1. Sunflower oil chain: main processes and system boundaries.

Table 1 shows the main inventory data (average) for sunflower cultivation (a); oil extraction and treatment (b). Two alternative agriculture practices (irrigated and non-irrigated cultivation) were modelled. The main difference between these practices is that irrigated cultivation requires 3000-6000 m^3/ha/year of water (Toureiro et al., 2005) and fertilisers (Table 1), while non-irrigated cultivation does not use either. The average productivity of irrigated cultivation is 3000 kg/ha/year, while non-irrigated cultivation is 650 kg/ha/year (Gírio et al., 2010). Data for the oil extraction process, using hexane as a solvent, was based on Jungbluth et al., (2007). After extraction, the oil was separated from the solvent by a distillation process.

Table 1. Main inventory data: a) sunflower cultivation (1 kg sunflower); b) extraction and treatment of sunflower oil (1 kg, no allocation).

a)

Main Inputs		Irrigated	Non-Irrigated	Unit
Fertilisers	N	0.007	-	kg
	K_2O	0.021	-	kg
	P_2O_5	0.021	-	kg
Pesticide (atrazine)		0.001	0.0023	kg
Seeds for planting		0.0023	0.0046	kg
Diesel		0.0523	0.1539	L
Water		1.5	_-	m^3
Product		**Irrigated**	**Non-Irrigated**	**Unit**
Sunflower seeds		1	1	kg

b)

Main Inputs	Value	Unit
Sunflower seeds	2.29	kg
Natural Gas	1.63	MJ
Bentonite	$5.38x10^{-3}$	kg
Hexane	$2.53x10^{-3}$	kg
Phosphoric acid	$8.16x10^{-4}$	kg
Electricity	$9.66x10^{-2}$	kWh
Co-products	**Value**	**Unit**
Sunflower oil	1	kg
Sunflower meal	1.29	kg

3. Land use change scenarios and carbon calculations

Twenty-eight LUC scenarios were assessed based on a combination of seven reference land use types (grassland (R1-R3) and perennial crops (R4-R7)) and four actual land uses (irrigated sunflower (I) (A1 and A2) and non-irrigated sunflower (NI) (A3 and A4)) (Table 2). The emissions from carbon-stock changes caused by LUC (e_1, kg CO_{2eq}/kg_{oil}) were calculated using IPCC Tier 1, (IPCC, 2006) and adapting the following equation from the Renewable Energy Directive (RED, EC, 2009):

$$e_1 = (C_{SR}-C_{SA}) \text{ x } 3.664 \text{ x } 1/20 \text{ x } 1/P \qquad \text{Eq. 1}$$

in which:
C_{SR} is the carbon stock associated with the each reference LU (kg CO_{2eq}/ha);
C_{SA} is the carbon stock associated with the actual LU (sunflower oil plantation) (kg CO_{2eq}/ha);
P is the sunflower oil productivity (kg oil/ha)

Based on the Portuguese climate region (warm temperate dry) and soil type (high activity clay soils), a standard value of 38 t C/ha was obtained from EC (2010) for soil organic carbon (SOC_{ST}). To calculate the reference and actual land use soil organic carbon (SOC_R and SOC_A), appropriate values for the factors reflecting the difference in SOC associated with type of land use (F_{LU}), management practice (F_{MG}) and different levels of carbon input to soil (F_I) compared to the SOC_{ST} were selected from EC (2010) (Table 2). Above and below-ground vegetation carbon stock (C_{VEG}) also came from EC (2010) (Table 2).

Table 2. Calculated soil organic carbon (SOC), above and below-ground vegetation carbon stock in living biomass (C_{VEG}), and total values (C_S) for reference ($_R$ subscript) and actual ($_A$ subscript) land use.

Actual Land Use		$SOC_i = (SOC_{ST}*F_{LU}*F_{MG}*F_I)$				SOC_i (t C/ha)	C_{VEGi} (t C/ha)	$CS_i = SOC_i + C_{VEGi}$ (t C/ha)
		SOC_{ST} (t C/ha)	F_{LU}	F_{MG}	F_I			
Sunflower cultivation (irrigated, RT, medium input)	A1	38	0.8	1.02	1.00	31.01	0	31.01
Sunflower cultivation (irrigated, NT, medium input)	A2	38	0.8	1.10	1.00	33.44	0	33.44
Sunflower cultivation (non-irrigated, RT, low input)	A3	38	0.8	1.02	0.95	29.46	0	29.46
Sunflower cultivation (non-irrigated, NT, low input)	A4	38	0.8	1.10	0.95	31.77	0	31.77
Reference Land Use								
Grassland (improved, medium input)	R1	38	1.0	1.14	1.00	43.32	3.1	46.42
Grassland (improved, high input)	R2	38	1.0	1.14	1.11	48.09	3.1	51.19
Grassland (severely degraded, medium input)	R3	38	1.0	0.70	1.00	26.60	3.1	29.70
Perennial crop (RT, high input, with manure)	R4	38	1.0	1.02	1.37	53.10	43.2	96.30
Perennial crop (RT, high input, without manure)	R5	38	1.0	1.02	1.04	40.31	43.2	83.50
Perennial crop (NT, high input, with manure)	R6	38	1.0	1.10	1.37	57.27	43.2	100.40
Perennial crop (NT, high input, without manure)	R7	38	1.0	1.10	1.04	43.47	43.2	86.67

NT: no tillage; RT: reduced tillage; F_{LU}: type of land use; F_{MG}: management practice; F_I: different levels of carbon input to soil.

4. Results and Discussion

Life-cycle Impact Assessment (LCIA) results were calculated using the ReCiPe method (midpoint level and hierarchical perspective; Goedkoop et al., 2010). We selected the following six impact categories: climate change (CC), ozone depletion (OD), terrestrial acidification (TA), freshwater eutrophication (FWE), marine eutrophication (ME) and fossil depletion (FD). The allocation method had an important influence in the results. In this study, three allocation methods were analysed: mass (43% oil, 57% meal), energy (65% oil, 35% meal) and economic (77% oil, 23% meal). The highest impacts occurred for economic allocation. Below we present only results for mass-based allocation.

4.1. LCIA results excluding LUC

Sunflower cultivated on non-irrigated land had higher environmental impacts in the categories of CC, ME, FD and OD because of the low productivity per ha (650 kg/ha/year) (Table 3). On the other hand, sunflower cultivated on irrigated land (3000 kg/ha/year) had higher impacts for TA and FWE, due to the use of fertilisers. The life-cycle phase of sunflower oil with the highest environmental impacts was cultivation for all categories (70%-99%). The main contributors for the impacts in sunflower cultivated on irrigated land were fertilisers (10%-99%, for all impact categories) and diesel for agricultural processes (30%-45%, for all categories except FWE). The main contributor in non-irrigated cultivation was diesel (64%-84%, for all categories).

Table 3. LCIA results (1 kg sunflower oil, mass allocation, no land use change).

	CC (kg $CO_{2\,eq}$) $x10^{-1}$		OD (kg CFC^{-11}) $x10^{-7}$		TA (kg $SO_{2\,eq}$) $x10^{-3}$		FWE (kgP $_{eq}$) $x10^{-4}$		ME (kg N $_{eq}$) $x10^{-3}$		FD (kg oil $_{eq}$) $x10^{-1}$	
	I	NI	I	NI	I	NI	I	NI	I	NI	I	NI
Cultivation	6.08	8.22	0.72	1.05	4.34	3.94	3.56	0.085	1.42	2.12	1.87	2.53
Oil Extraction	0.81		0.10		0.28		0.025		0.047		0.32	
Total	6.89	9.03	0.82	1.15	4.62	4.22	3.59	0.11	1.47	2.17	2.19	2.85

I – Irrigated; NI – Non-Irrigated; CC – Climate change; OD – Ozone layer depletion; TA – Terrestrial acidification; FWE – Freshwater eutrophication; ME – Marine eutrophication; FD – Fossil depletion.

Normalised results (using global values for the Europe Union (EU_{25+3}) year 2000, as reference) were calculated. Normalisation is an optional step in LCA and relates the magnitude of the impacts to reference values (Clift et al., 2000). It places LCIA indicator results into a broader context and adjust the results to have common dimensions. Fig. 2 shows that normalised results had similar magnitude for all categories (0.6 $x10^{-4}$ - 1.7 $x10^{-4}$), except for FWE and OD. FWE had the highest normalised impacts for irrigated cultivation due to the use of P_2O_5 fertiliser (Fig. 2). These impacts are about 30 times higher than those for non-irrigated cultivation (in which there was no fertiliser input), meaning that the use of fertiliser dominates the impacts in FWE.

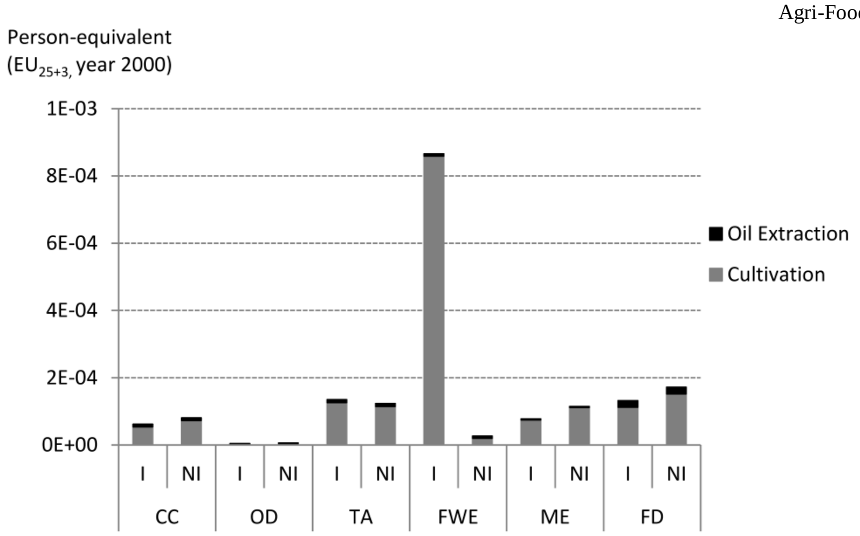


Figure 2. Normalised LCIA results (1 kg sunflower oil, mass allocation, no land use change).

4.2. Climate change impact: LUC scenario analysis

Sunflower oil GHG intensity greatly depends on the LUC scenario and varies greatly (0.3-20.9 kg CO_{2eq}/kg_{oil}) (Fig. 3). The lowest values were obtained when sunflower was cultivated on severely degraded grassland (R3), for which there was an increase in SOC (negative GHG emission in Fig.3). Highest values occurred when perennial crops (R4 to R7) were converted into sunflower cultivation, due to an important loss of above and below-ground vegetation carbon stock (C_{VEG}) from the previous perennial crop land.

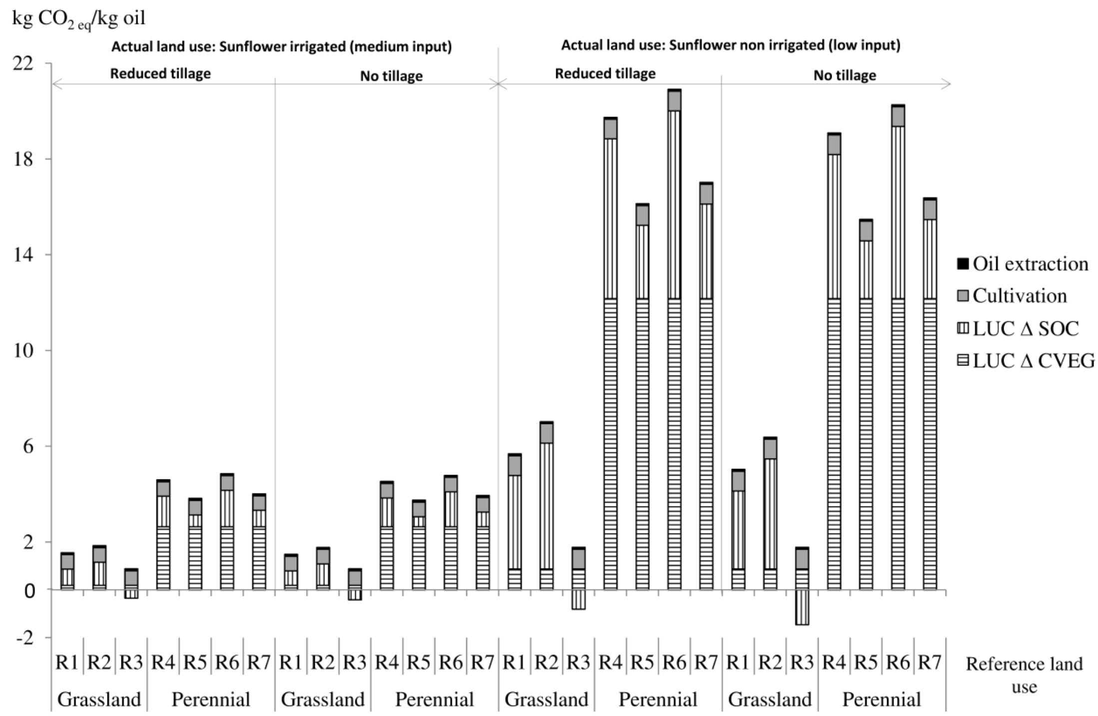


Figure 3. Greenhouse gas intensity of sunflower oil: land use change (LUC) and agriculture-practice scenarios (mass allocation).

5. Conclusion

Sunflower cultivated on non-irrigated land had higher environmental impacts in 4 categories (CC, ME, FD and OD) because of the low productivity per ha (650 kg/ha/year), while sunflower cultivated in irrigated land (3000 kg/ha/year)) had higher impacts in only 2 categories (TA and FWE) due to the use of fertilisers. The FWE impacts were about 30 times higher for irrigated cultivation relatively to non-irrigated cultivation (in which there was no fertiliser input). Cultivation contributed 70%-99% to the life-cycle impacts in all categories, mainly due to fertilisers and diesel (agricultural processes in irrigated cultivation). Normalised results had similar magnitude for all categories, except for FWE and OD.

A huge variation in GHG intensity for sunflower oil in Portugal was calculated (0.3-20.9 kg CO_{2eq}/kg_{oil}, mass allocation), demonstrating that agricultural practices and LUC scenarios have an important influence on GHG intensity. To assure low GHG intensity, sunflower should preferably be cultivated in severely degraded grassland. Cultivation on previous perennial crop land should not be used.

6. Acknowledgment

The authors gratefully acknowledge the project ECODEEP (Eco-efficiency and Eco-management in the Agro Industrial sector, nº FCOMP – 05 – 0128 – FEDER – 018643) and the Portuguese Science and Technology Foundation (FCT) projects: PTDC/SEN-TRA/117251/2010 (Extended “Well-to-Wheels” Assessment of Biodiesel for Heavy Transport Vehicles) and MIT/SET/0014/2009 (Capturing Uncertainty in Biofuels for Transportation. Resolving Environmental Performance and Enabling Improved Use).

7. References

Balafoutis, A., Freire, F., Papadakis, G. (2010). Life-cycle Assessment of Sunflower oil produced in Greece and used as fuel in tractor diesel engines. Proceedings of the International Conference of Agricultural Engineering (AgEng2010). 1-9.

Castanheira É.G.,Freire F. (2011a). “Environmental performance of palm oil biodiesel in a life-cycle perspective". IEEE International Symposium on Sustainable Systems and Technology. Chicago.

Castanheira É.G., Freire F. (2011b). “LC GHG Assessment of soybeans.". LCM 2011 - Towards Life-cycle Sustainability Management. Berlin.

Clift, R., Doig, A., Finnveden, G. (2000). The application of life-cycle assessment to integrated solid waste management. Part 1 – Methodology. Institution of Chemical Engineers. Trans IChemE, Vol 78, part B.

Cotana, F., Barbarena, M., Fantozzi, F., Ciriaf, C. N. (2010). Life cycle assessment of sunflower and rapeseed. 10° Congresso Nazionale CIRIAF – Atti (Perugia 9/10 April 2010). Sviluppo Sostenibile, Tutela dell’Ambiente e della Salute Umana.

EC, (2010). Directive 2009/28/EC of the European Parliament and of the council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. European Commission, Official Journal of the European Union, L140/16 of 5.6.2009.

EC, (2011). Commission Decision of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC. European Commission. Official Journal of the European Union, L151/19 of 17.6.2010.

Gírio, F.M., Matos, C.T., Oliveira, C.A., Silva, L., Bogel-Lukasik, R., Aguiar, R. (2010). Typical greenhouse gas emissions in the cultivation phase of raw materials for the production of biofuels. Report from Portugal. Agriculture. http://ec.europa.eu/energy/renewables/transparency_platform/emissions_en.htm

Goedkoop, M.J., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., Van Zelm R. (2008). A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level; First edition Report I.

IPCC (2006). Guidelines for national greenhouse gas inventories. In: Intergovernmental Panel on Climate Change, Agriculture, Forestry and other Land Use, vol. 4. Iriarte, A., Rieradevall, J., Gabarrell, X. (2010). Life cycle assessment of sunflower and rapeseed as energy crops under Chilean conditions. Journal of Cleaner Production, 18(4), 336-345. Elsevier Ltd.

ISO 14044, (2006): Environmental management—life cycle assessment—requirements and guidelines; Genève, Switzerland.

Kallivroussis, L. (2002). Rural Development The Energy Balance of Sunflower Production for Biodiesel in Greece. Biosystems Engineering, 81(3), 347-354.

JEC, (2008). Concawe JRC EUCAR Description and detailed energy and GHG balance of individual pathways, Well-to-Wheels analysis of future automotive fuels and powertrains in the European context WELL-TO-TANK Report Version 3.0 November 2008 WTT.

Jungbluth N., Chudacoff M., Dauriat A., Dinkel F., Doka G., Faist Emmenegger M., Gnansounou E., Kljun N., Spielmann M., Stettler C., Sutter J. (2007). Life Cycle Inventories of Bioenergy. Final report ecoinvent data v2.0. Volume: 17. Swiss Centre for LCI, ESU. Duebendorf and Uster, CH.

Lechon, Y., Cabal, H., Sáez, R. (2011). Life cycle greenhouse gas emissions impacts of the adoption of the EU Directive on biofuels in Spain. Effect of the import of raw materials and land use changes. Biomass and Bioenergy, 35(6), 2374-2384.

Malça, J., Freire, F. (2010). Uncertainty Analysis in Biofuel Systems: An Application to the Life Cycle of Rapeseed Oil. J Ind Ecol, vol. 14, pp. 322-334, 2010.

Malça, J., Freire, F. (2011). Life-cycle studies of biodiesel in Europe: A review addressing the variability of results and modeling issues. Renewable and Sustainable Energy Reviews, 15(1), 338-351.

Requena, J. F.S., Guimaraes, a C., Quirós Alpera, S., Relea Gangas, E., Hernandez-Navarro, S., Navas Gracia, L. M. (2010). Life Cycle Assessment (LCA) of the biofuel production process from sunflower oil, rapeseed oil and soybean oil. Fuel Processing Technology, 92(2), 190-199

Toureiro, C. M., Serralheiro, R. P., Oliveira, M. R. (2005). Resposta das culturas do girassol e do milho a diferentes cenários de rega deficitária. Revista de ciências agrárias, 35-47.

Tsoutsos, T., Kouloumpis, V., Zafiris, T., Foteinis, S. (2010). Life Cycle Assessment for biodiesel production under Greek climate conditions. Journal of Cleaner Production, 18(4), 328-335.

System expansion and allocation in the life-cycle GHG assessment of soybean oil

Érica Geraldes Castanheira*, Fausto Freire

ADAI-LAETA, Department of Mechanical Engineering, University of Coimbra, Pólo II Campus, Rua Luís Reis Santos, 3030-788 Coimbra, Portugal
* Corresponding author. E-mail: *erica@dem.uc.pt*

ABSTRACT

This paper presents a life-cycle (LC) greenhouse gas (GHG) assessment of soybean oil addressing critical issues in the modelling and results. Different methods for handling co-production in the oil extraction process are compared based on a sensitivity analysis to illustrate the influence on the results of using different allocation approaches and scenarios for system expansion. The implications of alternative land use change (LUC) scenarios associated with the expansion of different soybean cultivation systems have also been evaluated. The LC GHG emissions of soybean oil vary widely for the alternative methods for handling co-products adopted demonstrating, based on the soybean oil case, the importance of assessing different approaches to deal with multifunctionality in agri-food chains. Concerning the LUC scenarios, the results show that LUC dominates soybean oil LC GHG emissions, but significant GHG variation was observed between the alternative scenarios.

Keywords: life-cycle GHG assessment, multifunctionality, soybean meal, soybean oil, substitution

1. Introduction

Despite the significant growth in the number of published studies addressing the life-cycle (LC) greenhouse gas (GHG) intensity of soybean oil for food and biodiesel production purposes (Buratti et al., 2010; Castanheira and Freire, 2012; Hou et al., 2011, 2009; Kim and Dale, 2009; Panicheli et al., 2009; Reinhard and Zah, 2009), disagreement and controversies exist regarding the modelling assumptions or choices of these studies. The approach selected to attribute the emissions among the co-products in the soybean oil chain has been one of this most critical issues, since it may significantly influence or even determine the result of the assessment (Weidema, 2003). In some of the soybean oil LC GHG assessment studies, only one type of multifunctionality approach is applied (Buratti et al., 2010) and even for the same approach the results vary widely (Castanheira and Freire, 2012; Huo et al., 2009). Since multifunctionality is a critical in the soybean oil chain, a sensitivity analysis shall be conducted to illustrate how different methods change the results (ISO, 2006).

Another important aspect that has not captured sufficient attention yet is the GHG emissions due to the carbon stock changes from land use change (LUC) related with different soybean cultivation systems expansion. The main goal of this paper is to assess and discuss the implications of adopting different allocation methods (mass, energy and marked price based allocation) and different scenarios of system expansion on the LC GHG emissions of soybean oil. The implications of alternative LUC scenarios were also evaluated.

2. LC modeling and inventory

A LC model and inventory for the various stages of the soybean oil chain was developed and implemented. The model includes land use change (LUC), soybean cultivation in Brazil and soybean transport to Portugal, where soybean oil extraction, together with soybean meal production, takes place. A comprehensive evaluation of 20 scenarios, resulting from a combination of alternative LUC scenarios (conversion of managed forest plantation, perennial crop plantation, savannah and grassland) and different cultivation systems (tillage and no-tillage) was performed to analyse the effect on soybean oil GHG intensity. Table 1 shows the main inputs for 2 inventories of alternative soybean cultivation that were selected from transparent studies: no-tillage system (Cavalett and Ortega, 2009, 2010) and tillage system (Jungbluth *et al*., 2007). It should be noted that these studies are independent and use different data sources for the 2 inventories presented in Table 1. Thus, the differences between them are not necessarily caused by the tillage system. In fact, it can be seen that no-tillage requires more fertiliser than tillage and that both systems use the same amount of diesel. This does not seem very logical since no-tillage requires less machinery (diesel) work and fertilisers. GHG emissions arise from fertiliser application and biological nitrogen fixation (N_2O), diesel combustion from agricultural operations and from the production of inputs (Frischknecht et al., 2007; IPCC, 2006; Jungbluth, 2007; Kellenberger et al., 2007; Nemecek et al., 2007; Patyk et al., 1997).

Table 1. Soybean cultivation systems: main inputs and yields (values per ha and year).

	No-tillage (Cavalett and Ortega, 2009, 2010)	Tillage (Jungbluth et al., 2007)
Pesticides	8.0 kg	1.47 kg[a]
Limestone	375 kg	-
Fertilisers	33.8 kg P and 65.4 kg K	30 kg P_2O_5[b] and 30 kg K_2O
Diesel	65 L	65 L
Electricity	122 MJ	-
Soybean production (yield)	2830 kg	2544 kg

[a] 2,4-D (51%), glyphosate (37%), monocrotofos(8%) and endosulfan (4%).
[b] Diammonium phosphate (45%), single super phosphate (29%), triple super phosphate (16%), phosphate rock (5%) and ammonium nitrate phosphate (5%).

Two climate regions (tropical moist and warm temperate moist) and a low activity clay soil were considered for LUC emissions calculations. GHG emissions from carbon stock changes caused by LUC were calculated based on the carbon stock associated with each **R**eference land use, **CS_R** (previous land use types) and the carbon stock associated with the soybean production systems considered, **CS_A** (**A**ctual land use), following IPCC Tier 1 and European Union Directive on Renewable Energy (EC, 2009, 2010; IPCC, 2006). Soil organic carbon (SOC) calculated and the above and below ground vegetation carbon stock in living biomass and in dead organic matter (C_{VEG}) adopted from EC (2010) are presented in Fig. 1 for the various reference land use (SOC_R and C_{VEGR}) and for the soybean plantations (SOC_A and C_{VEGA}).

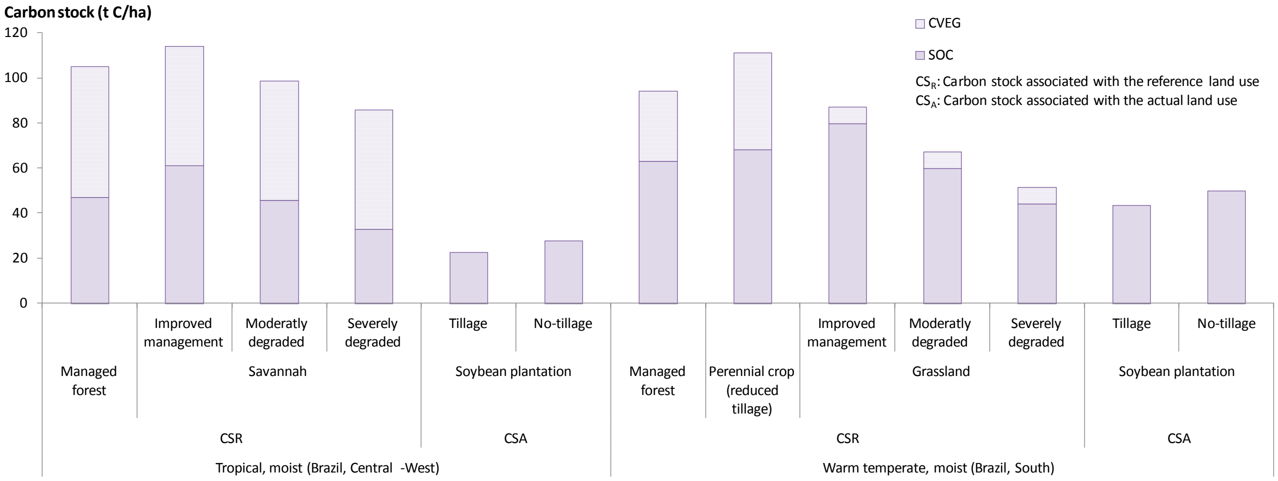


Figure 1. Carbon stocks of previous land use (SOC_R and C_{VEGR}) and of soybean plantation (SOC_A and C_{VEGA}) for 2 climate regions in Brazil.

Soybean produced in Brazil is transported from plantation to the Paranaguá port by road (897 km) and by transoceanic freight ship over 8393 km to Lisbon port (near the oil extraction plant). GHG emissions related with the soybean transportation were calculated based on these average distances and the factors given by Spielmann et al., (2007). Regarding the soybean oil extraction process (co-produced with soybean meal), a LC inventory and modelling was implemented based on average data collected from Portuguese industry. Electricity and heat requirements are 0.16 MWh/t oil and 3292 MJ/t oil, respectively. Natural gas and heavy fuel oil are used to produce heat. Electricity is obtained from the grid and produced onsite from a natural gas combined heat and power (cogeneration) plant. GHG emissions from electricity and heat production were calculated based on EC (2009), Faist Emmenegger et al., (2007), Frischknecht et al., (2007) and Jungbluth et al., (2007). The GHG emissions from hexane production have also been considered based on the quantity consumed (7.9 kg/t oil) and the GHG emission factor (Jungbluth et al., 2007). Since a valuable co-product is obtained from the soybean oil extraction system, the soybean meal, there is a multifunctional issue that should be solved. Several scenarios based on alternative allocation and substitution approach were used for dealing with this co-production, as detailed in the next sub-section.

2.1. Multifunctionality in the soybean oil process: allocation and substitution scenarios

The ISO standards provide a hierarchical approach for handling co-products (ISO, 2006): avoiding allocation (by dividing the unit process to be allocated into sub-processes or expanding the product system to include the additional functions related to the co-products) and allocation based on physical (e.g. mass, energy) or other (market price) relationships between the co-products.

In the substitution method, the system is expanded with "avoided" processes to remove additional functions related to the soybean meal (Guinee et al., 2009). Two alternative scenarios for the substitution of soybean meal were implemented. Soybean meal is mainly used as animal feed and these scenarios took into account two product systems that are currently displaced by soybean meal co-produced with soybean oil in Portugal according to experts from the soybean oil industry.

- Substitution scenario "*Imported Soybean meal" (ISM)* describes the case in which soybean meal co-produced with soybean oil in Portugal displaces direct soybean meal imports from Argentina for animal feed purposes. Substitution credits for GHG emissions were quantified based on the GHG intensity of soybean meal produced in Argentina given by Dalgaard et al., (2008): 721 g CO_2eq/kg meal (no LUC).
- Substitution scenario "*Imported Soybean" (IS)* describes the case in which soybean meal co-produced with soybean oil in Portugal displaces direct soybean imports from Brazil (also used as animal feed in Portugal). The substitution credits for GHG emissions were quantified based on the GHG intensity of soybean produced in Brazil (calculated in this paper: 432 g CO_2eq/kg soybean, no LUC and no-tillage) and assuming a displacement ratio of 0.85 kg soybean meal per kg of soybean, based on an equivalent protein content.

In allocation, the multifunctional process is split up into two single functional processes (oil + meal) based on specific relationships. In this paper, a sensitivity analysis of allocation based on mass, energy (lower heating value, LHV) and market prices was performed. Table 2 shows the mass, energy and economic data (based in 2010 monthly market prices from IMF, 2011) used to calculate the allocation factors for soybean oil and meal. Three alternative economic allocation factors were calculated based on: the average annual prices for oil and meal ("Av"); the monthly prices for which the price ratio between oil and meal is the lowest ("Min") and it is the highest ("Max"). It should be noted that soybean oil and meal have different characteristics, being used for completely different purposes (e.g. energy purposes, food and feed production), which complicates the selection of a particular allocation method.

Table 2. Soybean oil and meal mass ratios, LHV, market prices and corresponding allocation factors.

	Mass		**Energy**		**Market price**					
	Mass ratio (t/t oil)	**Allocation factor**	**LHV (MJ/kg)**	**Allocation factor**	**Prices[a] (US$/t)**			**Allocation factor**		
					Min	**Max**	**Av**	**Min**	**Max**	**Av**
Oil	1.0	20%	36.6	36%	837	1208	925	38%	43%	41%
Meal	4.1	80%	16.3	64%	335	388	331	62%	57%	59%

[a] IMF (2011)

3. Results and discussion

Fig. 2 presents the results of a sensitivity analysis conducted to illustrate the implications of different multifunctionality approaches on the GHG balance of soybean oil (excluding LUC). It can be observed that the results obtained for different approaches present significant variation: between -0.36 and 1.12 kg CO_2eq/kg. The lowest value was obtained when the substitution scenario "*Imported Soybean meal" (ISM)* is adopted. In contrast, soybean oil had higher emissions for market-price-based allocation. This large variation in results demonstrates the critical influence on the results of the method selected for handling co-production. This also justifies the need to perform a sensitivity analysis, as recommended by ISO 14044:2006 (ISO, 2006).

Regarding allocation approaches, the GHG emissions calculated based on energy or market price allocations are almost double compared results obtained using mass allocation. The results do not significantly vary in consequence of annual market price variations, since soybean meal has a relatively high mass share of the extraction process, and its price (mostly driven by the livestock feed industry) does not vary as much as the oil price (Fig. 2). Furthermore, it can be said that it is soybean meal demand that determines the production volume of the soybean oil extraction process.

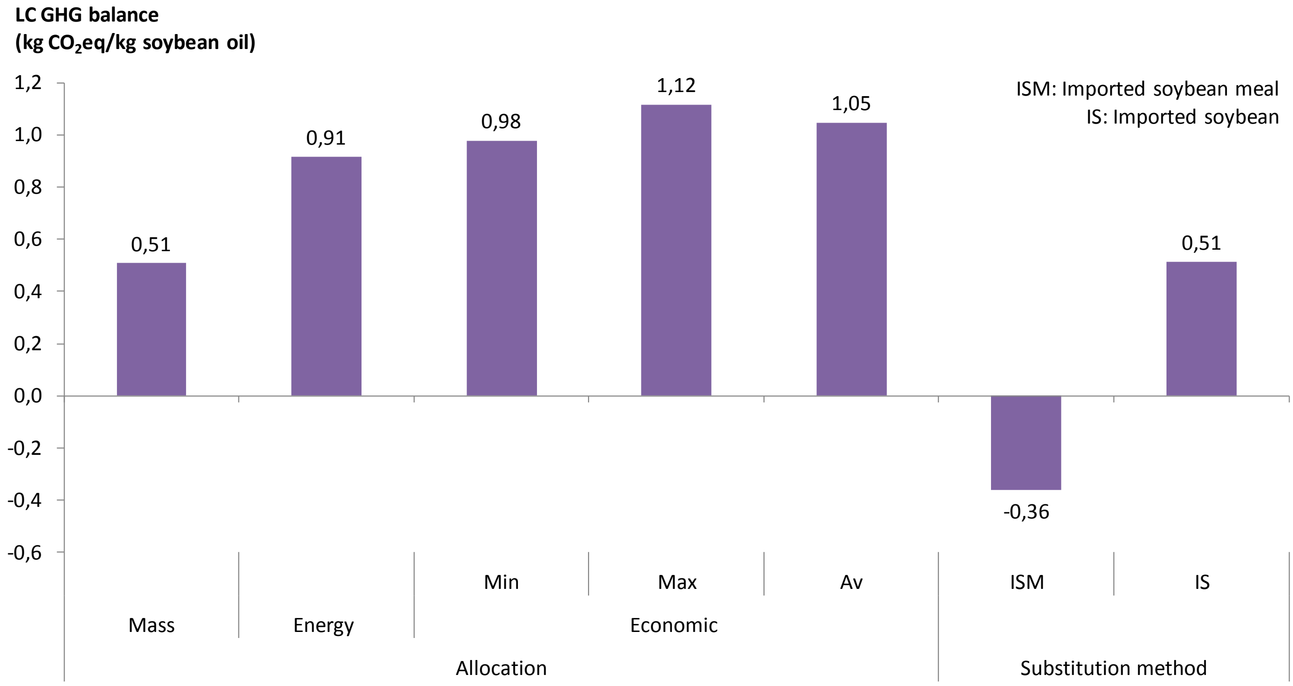


Figure 2. LC GHG emissions of soybean oil (no LUC, no-tillage): different methods for handling co-products.

Fig. 3 presents soybean oil life-cycle GHG emissions (energy allocation) from soybean produced in 2 climate regions in Brazil. The results for the alternative LUC scenarios are comparatively assessed. The contribution of each LC phase (LUC, cultivation, transport and oil extraction) is also shown. The results show a huge variation in GHG emissions: between 0.9 kg CO_2eq/kg (no LUC) and 12.9 kg CO_2eq/kg (previous improved management savannah, tropical moist region). LUC dominates the results. GHG emissions due to LUC represent more than 69% in all scenarios, except for the scenarios where soybean is cultivated in previous severely degraded grassland in warm temperate region. The GHG emissions of soybean oil associated with cultivation, transport and oil extraction are also compared in Fig. 3. Concerning the scenarios with no LUC, process contributions to the GHG balance are as follows: cultivation (3-44%), transport (3-44%) and oil extraction (1-14%).

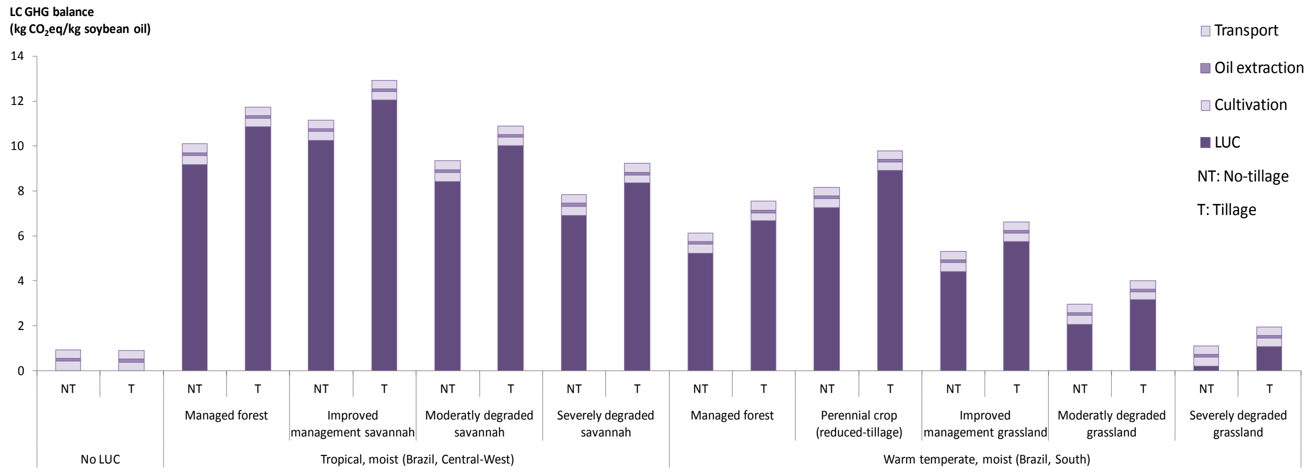


Figure 3. LC GHG emissions of soybean oil (energy allocation): LUC scenarios and phase contribution.

4. Conclusions

The main goal of this paper is to assess and discuss the implications of adopting different allocation methods (mass, energy and marked price based allocation) and different scenarios of system expansion on the LC GHG emissions of soybean oil. The sensitivity analysis conducted to illustrate the consequences of different multifunctionality approaches shows that results are very sensitive to the approach adopted. The lowest GHG emissions were obtained when substitution scenario *"Imported Soybean meal" (ISM)* is adopted and the highest emissions when market price based allocation is applied. The implications of alternative LUC scenarios were also evaluated, showing that the original land choice is a critical issue. It was not possible to compare results from alternative cultivation systems (tillage and non-tillage) due to limitations of inventory data. The results presented here suggest future work developing specific LC inventory and modeling for tillage and no-tillage soybean plantation to allow a comparison between cultivation systems.

To assure the lowest LC GHG balance of soybean oil, severely degraded grassland should be preferably used for soybean cultivation. The results demonstrate the importance of LUC on soybean oil GHG emissions; however, large variations on the GHG balance were calculated between the various LUC scenarios.

5. Acknowledgments

The research presented in this paper was supported by ECODEEP project (Eco-efficiency and Eco-management in Agro Industrial sector, nº FCOMP – 05 – 0128 – FEDER – 018643) and the Portuguese Science and Technology Foundation (FCT) projects: PTDC/SEN-TRA/117251/2010 (Extended "well-to-wheels" assessment of biodiesel for heavy transport vehicles) and MIT/SET/0014/2009 (Capturing Uncertainty in Biofuels for Transportation. Resolving Environmental Performance and Enabling Improved Use). Furthermore, Érica Geraldes Castanheira gratefully acknowledges support from FCT, through grant SFRH/BD/60328/2009 and the Energy for Sustainability Initiative at the University of Coimbra (www.uc.pt/efs).

6. References

Buratti, C., Moretti, E., Fantozzi, F., 2010. Assessing the GHG emissions of rapeseed and soybean biodiesel in compliance to the EU Renewable Energy Directive methodology for biofuels. 18th European Biomass Conference and Exhibition, 3-7 May 2010, Lyon, France.

Castanheira, É., Freire, F., 2012. Life-cycle GHG assessment of soybean biodiesel. Proceedings of the International Symposium on Sustainable Systems and Technology (IEEE ISSST). 5 pages. Boston, MA, USA.

Cavalett, O., Ortega, E., 2010. Integrated environmental assessment of biodiesel production from soybean in Brazil. Journal of Cleaner Production 18, 55-70.

Cavalett, O., Ortega, E., 2009. Emergy, nutrients balance, and economic assessment of soybean production and industrialization in Brazil. Journal of Cleaner Production 17, 762-771.

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W. A., 2008. LCA of Soybean Meal. Int. J. Life Cycle Assess. 3, 240-254.

European Commission, 2010. Commission Decision of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC. Official Journal of the European Union, L151/19 of 17.6.2010.

European Commission, 2009. Directive 2009/28/EC of the European Parliament and of the council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union, L140/16 of 5.6.2009.

Faist Emmenegger, M., Heck, T., Jungbluth, N., 2007. Erdgas. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent data v2.0. Editors: Dones R.. Volume: 6. Swiss Centre for LCI, PSI. Dübendorf and Villigen, CH.

Frischknecht, R., Tuchschmid, M., Faist Emmenegger, M., Bauer, C., Dones, R., 2007. Strommix und Stromnetz. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent data v2.0. Editors: Dones R. Volume: 6. Swiss Centre for LCI, PSI. Dübendorf and Villigen, CH.

Guinée, J.B., Heijungs, R., van der Voet, E., 2009. A greenhouse gas indicator for bioenergy: some theoretical issues with practical implications. Int J LCA 14(4), 328-339.

Hou, J., Zhang, P., Yuan, X., Zheng, Y., 2011. Life cycle assessment of biodiesel from soybean, jatropha and microalgae in China conditions. Renewable and Sustainable Energy Reviews 15, 5081–5091.

Huo, H., Wang, M., Bloyd, C., Putsche, V., 2009. Life-Cycle Assessment of Energy Use and Greenhouse Gas Emissions of Soybean-Derived Biodiesel and Renewable Fuels. Environ. Sci. Technol. 43, 750-756.

IMF, 2011. Historical commodity prices - International Monetary Fund. Available at IndexMundi website (http://www.indexmundi.com/commodities/), 11.07.2011.

IPCC, 2006. IPCC guidelines for national greenhouse gas inventories. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.). Hayama, Japan: Institute for Global Environmental Strategies.

ISO, 2006. ISO 14044: environmental management. Life cycle assessment e requirements and guidelines. ISO-International Organization for Standardization, Genève, Switzerland.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Spielmann, M., Stettler, C., Sutter, J., 2007. Life Cycle Inventories of Bioenergy. Final report ecoinvent data v2.0. Volume: 17. Swiss Centre for LCI, ESU. Duebendorf and Uster, CH.

Jungbluth, N., 2007. Erdöl. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent data v2.0. Editors: Dones R. Volume: 6. Swiss Centre for LCI, PSI. Dübendorf and Villigen, CH.

Kellenberger, D., Althaus, H.-J., Jungbluth, N., Künniger, T., 2007. Life Cycle Inventories of Building Products. Final report ecoinvent data v2.0, Vol. 7, Swiss Centre for LCI, Empa – TSL, Dübendorf.

Kim, S., Dale, B. E., 2009. Regional variations in greenhouse gas emissions of biobased products in the United States—corn-based ethanol and soybean oil. Int J Life Cycle Assess,14, 540–546.

Nemecek, T., Kägi, T., Blaser, S., 2007. Life Cycle Inventories of Agricultural Production Systems, Ecoinvent report version 2.0, Vol. 15, Swiss Centre for LCI, ART, Duebendorf and Zurich.

Patyk, A., Reinhardt, G., 1997. Düngemittel - Energie- und Stoffstromsbilanzen. Vieweg. Umweltvissenschaften. Friedr. Vieweg & Sohn Verlagsgesellschaft mbH, Braunschweig/Wiesbaden, Germany.

Panichelli, L., Dauriat, A., Gnansounou, E., 2009. Life cycle assessment of soybean-based biodiesel in Argentina for export. Int. J. Life Cycle Assess. 14, 144-159.

Reinhard, J., Zah, R., 2009. Global environmental consequences of increased biodiesel consumption in Switzerland: consequential life cycle assessment. Journal of Cleaner Production 17, S46–S56.

Spielmann, M., Dones, R., Bauer, C., 2007. Life Cycle Inventories of Transport Services. Final report ecoinvent Data v2.0, Vol.: 14, Issue: 0, Swiss Centre for LCI, PSI, Dübendorf and Villigen.

Weidema, B., 2003. Market information in life cycle assessment, Environmental Project No. 863. Available at: http://www.norlca.org

Considering land use change and soil carbon dynamics in an LCA of French agricultural products

Thibault Salou[1,2,*], Etienne Mathias[3], Aurélie Tailleur[4], Anne Paillier[5], Hayo M.G. van der Werf[1,2]

[1]INRA, UMR1069 Sol, Agro and hydroSystem, 35000 Rennes, France
[2]Agrocampus Ouest, 35000 Rennes, France
[3]CITEPA, 7 Cité Paradis – 75010 Paris, France
[4]ARVALIS – Institut du Végétal, 75016 Paris, France
[5]ADEME, 20 Avenue du Grésillé - 49 000 Angers, France
[*]Corresponding author. E-mail: *thibault.salou@rennes.inra.fr*

ABSTRACT

Agri-BALYSE is a programme to provide LCIs of agricultural products at the farm gate, to bring reliable and complete environmental information on "product plus packaging" to consumers. The choice of appropriate methods for the estimation of resource use and emissions of pollutants is a major challenge for Agri-BALYSE.
Two methods to estimate soil C dynamics from direct Land Use Change (LUC) in mainland France were developed. These methods are based on IPCC (2003) recommendations. Data from the French Soil Survey Network (Martin et al., 2011) and the French United Nations Framework Convention on Climate Change National Inventory Report (UNFCCC NIR) (CITEPA, 2012) allowed us to determine: i) soil C stocks, ii) areas involved in LUC for both grasslands and annual crops, for each year of our reference period (2005-2009). Emission factors were determined for permanent grassland and annual crops in the simplified method; for permanent grassland and for each annual crop individually, taking in consideration spatial repartition, in the more accurate method.

Keywords: soil carbon sequestration, land-use change, direct field emission, grassland

1. Introduction

In 2009 two French laws, Grenelle 1 and 2, on the provision of reliable and complete environmental information on "product plus packaging" to consumers were passed. The Life Cycle Assessment (LCA) method was chosen to assess environmental impacts of products. ADEME, the French Environment and Energy Management Agency, was mandated to set up a Life Cycle Inventory (LCI) database to support this policy. In early 2010 ADEME commissioned the Agri-BALYSE programme involving: i) ART (Switzerland) and INRA as project co-leaders; ii) CIRAD, ACTA and 10 technical agricultural institutes for data collection, elaboration of methods and implementation in practice. The aims of Agri-BALYSE are: i) to provide LCIs of agricultural products at the farm gate; ii) to establish a common methodological framework to create LCIs of agricultural goods.

The choice of appropriate methods for the estimation of resource use and emissions of pollutants associated with agricultural products is a major challenge for Agri-BALYSE. In this respect the impact of land-use change (LUC) on GreenHouse Gas (GHG) emissions is an important issue. Several studies propose methodologies to consider this issue: IPCC (2003) applied to France in the French United Nations Framework Convention on Climate Change National Inventory Report (UNFCCC NIR) (CITEPA, 2012), Milà i Canals et al., (2012).

The aim of Agri-BALYSE is to provide LCIs of French agricultural products at the farm gate. To ensure consistency between the various products of the data base, a general methodological framework for the programme was defined. It was decided that the methodologies used and the deliverables must be consistent with ILCD recommendations (JRC and EIS, 2010). This guide recommends "for CO_2 release caused by land use and land transformation, the use of the most recent IPCC CO_2 emission factor [...], unless more accurate, specific data is available". Within the Agri-BALYSE programme we decided to focus on LUC between permanent grasslands and annual crops in mainland France. We considered that in this context we could ignore changes in above- and below-ground biomass and consider that Soil Organic Carbon (SOC) was the main source of GHG emissions. According to IPCC 2003 two reasons can lead to a C soil stock change: i) modification of management practices (ex: fertilisation, vegetation cover); ii) LUC (ex: conversion from permanent grassland to annual crop).

In LCA studies carried out in France soil C dynamics are rarely considered, except when permanent grassland is concerned (Doreau and Dollé, 2011, Doreau et al., 2011, Nguyen et al., 2012). These studies involving permanent grassland use a method proposed by Doreau and Dollé (2011). Based on experimental results published by Arrouays et al., (2002), this method proposes emission factors: -500 kg C/ha/yr for permanent grassland up to 30 years old; -200 kg C/ha/yr for permanent grassland more than 30 years old. For temporary grassland, the calculation considers a cycle of 5 years of grassland that emits -500 kg C/ha/yr, followed by two years of annual crops that emit 1000 kg C/ha/yr. In this method emissions for annual crops are 0 kg C/ha/yr. This method could not be implemented in Agri-BALYSE because it is in conflict with IPCC recommendations, as it considers that permanent grassland (i.e. not resulting from LUC) sequesters C under constant management.

Measured emission factors for permanent grassland and annual crops are also available in the literature (Schulze et al., 2009; Soussana et al., 2009). We chose not to use them for two reasons: i) differences in the methodologies used to determine them, which is contrary to the desired consistency for the Agri-BALYSE LCI database; ii) difficulties to accurately identify the cause of this CO_2 emission (LUC, management change, increased atmospheric CO_2 content,...).

In this paper, we focus on soil carbon dynamics resulting from direct LUC in mainland France. In Agri-BALYSE, soil carbon dynamics for areas without LUC, i.e. resulting from changes in management practices, could not be taken into account, as insufficient data on management of crops and grassland were available.

2. Methods

In collaboration with CITEPA, the French Interprofessional Technical Centre for Studies on Air Pollution which is in charge of reporting French national greenhouse gas emissions, and ARVALIS, a technical institute involved in Agri-BALYSE, two methods for estimating soil C dynamics from direct LUC were developed. Both are based on the mass balance method described in IPCC 2003. The basics of these methods are to compare soil C stocks at two dates (corresponding to two land use types) and allocate the carbon emissions / absorptions over the period between the two dates. IPCC gives a transition period of 20 years as a standard. C flows can be estimated from LUC area and estimations of soil C stocks for the initial and final land-use. Soil C dynamics are considered to be linear over the transition period. One important hypothesis of the method is that temporary grassland is considered as an annual crop.

2.1. Construction of matrices

The first step of the calculation is the construction of matrices for LUC between permanent grassland and annual crops.

In France, the Ministry of Agriculture implements a very large annual survey to monitor land use changes over time. This program, named TerUti and TerUti-Lucas since 2005, provides the land cover and the land use of more than 300,000 fields sampled across mainland France. The survey contains a detailed nomenclature to characterise the land use of each field (Table 1), this nomenclature is based on two parameters: the land cover which is strictly the physical occupation (grass, trees, etc.), and the land-use which describes human activities. These definitions can differ from common understanding of grassland and cropland.

Table 1. Nomenclature for permanent grasslands and annual crops for the survey TerUti-Lucas.

	Land cover codes	**Land-use codes**
Annual crops	2110 (wheat) to 2530 (annual hay)	All
	2730 (crop nursery) to 2742 (other crops)	All
	6030 (bare soil)	111 to 114 (agriculture)
	9999 (unknown)	111 to 112 (cultivation)
Permanent grasslands	4020 (lands with bush and trees<5%)	111 to 120, 364-365, 402 (agriculture, forestry, protected areas, etc.)
	5021 to 5025 (lands covered by grass)	111 to 120, 364-365, 402 (idem previous)
	9999 (unknown)	113 to 114, 364-365 (forestry, protected areas)

These data are used in the framework of the French National greenhouse gas inventory (CITEPA, 2012) and allow the construction of land use change matrices in particular between permanent grassland and annual crops. In this case the matrices correspond to a 20-year period. Yet, even if very accurate surveys exist, these long-term matrices remain difficult to implement for two reasons: i) changes between grassland and cropland are rather frequent and may occur several times during a 20-year period; ii) the network of monitored fields is not fixed over a long period and has changed twice since 1981. Thus different data treatments (CITEPA, 2012) have been necessary to gather time-series and provide reliable land use change areas. Basically, the first corrections are to link the different nomenclatures, and then one of the major assumptions is that the estimate of land use change depends on the duration of the survey, a correction is then applied to the areas of changes for each time series (Figure 1). Final estimations of land use changes areas, used for the methodology Agri-BALYSE are provided in Table 2.

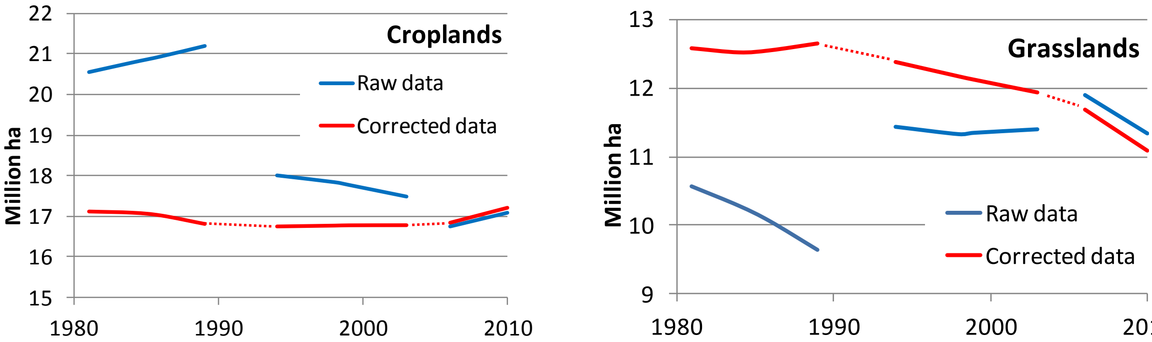


Figure 1. Cropland and grassland areas from 1981 – 2010, before and after correction.

Table 2. Twenty-year matrix of land use changes areas between permanent grasslands and annual crops for mainland France during the period 1990-2010, based on UNFCCC (CITEPA, 2012).

Land-use changes	Area (ha/year)
Annual crops becoming permanent grassland	118,158
Permanent grassland becoming annual crops	159,732

2.2. Simplified method to estimate soil C dynamics from LUC

We first calculate soil C stocks for permanent grasslands and annual crops. The data from the French Soil Survey Network (Martin et al., 2011), allowed the calculation of an average soil C stock for both types of land use. This was done at national scale (Table 3), these averages obviously integrate a significant variability.

Table 3. National average soil C stocks for permanent grassland and annual crops based on the French Soil Survey Network.

Land-Use	Average Soil C Stock (tC/ha)
Permanent grassland	72.7
Annual crops	53.3
Δ Grassland, Annual crop	19.4

Then, the land use data from the French UNFCCC NIR (CITEPA, 2012) allowed us to determine areas having undergone LUC less than 20 years ago for each year of the Agri-BALYSE reference period: 2005-2009.

The next step of the method is to determine, for each year of the reference period, annual soil C storage/emissions for areas having undergone LUC. The calculations have been done for each year of the reference period. This results in an emission factor for CO_2 from LUC expressed per ha of permanent grassland having undergone LUC (EF_{Grass_LUC} -3.16 t CO_2/ha) and a similar factor expressed per ha of annual crop (EF_{Crop_LUC} 3.40 t CO_2/ha), for each year of the 2005-2009 period.

An emission factor for permanent grasslands and annual crops, considering all areas of both land uses, including those that did not undergo LUC, is then calculated through Equations 1 and 2 from 2005 to 2009:

$$Annual\ soil\ C\ EF_{Grass\_Tot} = \frac{Grassland\ area\ having\ undergone\ a\ LUC}{Grassland\ total\ area} * EF_{Grass\_LUC} \qquad \text{Eq. 1}$$

$$Annual\ soil\ C\ EF_{Crop\_Tot} = \frac{Annual\ crop\ area\ having\ undergone\ a\ LUC}{Annual\ crop\ total\ area} * EF_{Crop\_LUC} \qquad \text{Eq. 2}$$

Grassland/annual crop area in ha
EF_{Grass_Tot}: Emission factor for CO_2 from LUC expressed per total area of permanent grassland (tC/ha)
EF_{Grass_LUC}: Emission factor for CO_2 from LUC expressed per area of permanent grassland having undergone LUC (tC/ha)
EF_{Crop_Tot}: Emission factor for CO_2 from LUC expressed per total area of annual crop (tC/ha)
EF_{Crop_LUC}: Emission factor for CO_2 from LUC expressed per area of annual crop having undergone LUC (tC/ha)

The average soil C emission factor over the Agri-BALYSE reference period is then calculated for grasslands and annual crops. It will be applied to each ha concerned.

2.3. Method to estimate crop-specific soil C dynamics from LUC

In order to take into account the regional specialisation of agriculture observed in France, we have tried to develop a more accurate method. This method should allow attributing LUC more to the crops which are most involved in LUC to and from permanent grassland.

The first steps are as for the simplified method. The data from Teruti-Lucas network and French Soil Survey Network are used to determine: i) permanent grassland and annual crop areas having undergone LUC less than 20 years ago for each year of the Agri-BALYSE reference period: 2005-2009.and ii) average soil C stock for both land uses.

Unlike the previous method, which is implemented at the scale of mainland France, this method is implemented at the scale of mainland France’s 22 administrative regions to calculate emissions of soil C at the regional scale. Negative emissions (i.e. C sequestration) are attributed to the areas in permanent grassland, while positive emissions are attributed to areas in annual crops, yielding regional emissions for permanent grassland and annual crops. The next step is to distribute emissions among crops within each region according to the relative area of each crop in the administrative region over the 2005-2009 period, using the statistic data base of the Office of Statistics and Studies (SSP) of the French Ministry for Food, Agriculture and Fisheries (AGRESTE, 2012). Finally a crop-specific emission factor for mainland France was calculated from regional crop emission factors as a weighted average.

3. Results and discussion

In a first approach, the simplified method was implemented and emission factors for permanent grassland and annual crops were calculated (Table 4).

Table 4. Total estimated areas of LUC from grassland to annual crops and from annual crops to grassland and CO_2 emissions per year per ha of annual crop and grassland in mainland France. Average total areas of annual crops and grassland and average CO_2 emissions from LUC expressed per ha of the total area of annual crop and grassland.

Land-Use Change	Area (ha)	Emission factor (t CO_2/ha)
Grassland to annual crops, 2005-2009	3 222 728	3.40
Annual crops to grassland, 2005-2009	2 764 221	- 3.16
Total annual crops, 2005-2009	16 686 248	0.66
Total grassland, 2005-2009	11 139 626	- 0.78

This was the first step to take in consideration carbon soil dynamics in Agri-BALYSE. However, even if: i) it is in accordance with one of the aims of Agri-BALYSE, i. e. the identification of a set of consistent and consensual methods for LCAs of agricultural products; ii) it is consistent with ILCD recommendations; iii) this methodology, based on IPCC 2003, has been used for several international studies/recommendations (BSI, 2011; Gerber et al., 2010; IDF, 2010; Leip et al., 2010; Commission Européenne, 2010); this method lacks sophistication, as it yields a single emission factor for all annual crops. Since the 70’s, a strong trend of regional specialisation of agricultural production systems has been observed in France (Dussol et al., 2003). Three main specialised regions can be distinguished: annual crops (Paris Basin and North); livestock production, involving grassland and annual crops (East, West, Massif Central and north of Alps); perennial crops and vegetables (South-West and South East). The proportion of permanent grassland in the agricultural area varies greatly between these regions. Therefore attributing the same emission factor to all annual crops is a major simplification of reality. For this reason we propose a more accurate method which takes in consideration spatial distribution of grasslands and annual crops across mainland France (Table 5). These emission factors take into account that some annual crops, mainly produced in regions where permanent grassland is rare, are less involved in LUC.

Table 5. Crop specific CO_2 emissions per year per ha of annual crops in mainland France, 2005 - 2009. Average CO_2 emissions, weighted by production area in each French region, from LUC expressed per ha of the total area of annual crop.

Annual crop	**Emission factor (t CO_2/ha)**
Sugar beet	0.30
Durum wheat	0.72
Wheat	0.58
Rapeseed	0.56
Faba bean	0.38
Silage maize	0.79
Maize	0.68
Barley	0.57
Pea	0.50
Potato	0.36
Temporary grassland	0.89
Sunflower	0.73
Triticale	0.92

In the methods proposed here, as soon as a variation of area for a type of land use was observed over the last 20 years, we considered that a LUC occurred, unlike for instance Milà i Canals et al., (2012), who propose a decision tree in two steps to determine if a LUC occurred over the last 20 years. The first step determines if the area of a crop increased in the country. If it increased, the second step determines if the total land type area for the considered crop also increased. If the answer to both is positive, a LUC is considered to have taken place. Furthermore, short-term fluctuations are smoothed out by the use of 5 years' average areas. It could be interesting to implement this last point in the method proposed here. Effectively, in the studied systems, temporary grasslands are frequent and can lead to mistakes in the counting of each land type area.

The methods proposed here aim to assess soil C dynamics associated with LUC in mainland France. However, the goal of Agri-BALYSE is to provide a methodology for harmonized LCIs of agricultural products, whether produced in mainland France or abroad. To achieve this goal at short term, and considering the current methodological developments, the method proposed by Milà i Canals et al., (2012) seems to be the most complete and operational, in particular when data availability and quality are insufficient to implement the methods proposed in this paper.

Within the Agri-BALYSE programme a considerable amount of effort has been invested in the development of the two methods described here, even though the development of methods is not part of the programme's mandate. After a major internal debate no consensus was found for the implementation of either method within the programme, mainly because these methods have not yet been validated by publication in a peer-reviewed journal. It was decided that CO_2 emissions resulting from land use change will not be considered in the Agri-BALYSE LCI database. However, a description of both methods proposed here and LCI results including CO_2 emissions according to both methods will be documented in the methodological report of the programme.

5. Conclusion

This work allowed a major advance in the consideration of soil carbon dynamic in LCAs of French agricultural goods. Two methods to estimate soil C dynamics from direct LUC for permanent grassland and annual crops in mainland France were developed. However, there is scope for a further refinement of these methods. We identified some specific points in the methodology which can be improved. These methods have not been implemented in the current version of the Agri-BALYSE database. They should be tested before implementing them in an operational database destined to support the sensitive issue of environmental labelling.

These improvements concern: i) estimation of areas that have undergone LUC and ii) calculation of crop-specific emission factors. The results presented in Table 5 are a first estimation of these crop-specific emission factors and need to be validated. Concerning estimations of areas having undergone LUC, while developing the two methods, we used data from Teruti-Lucas. These data need a treatment to be used in the calculations. This has been done by CITEPA on the one hand and by SSP on the other hand. The LUC areas obtained by these two methods do not match. Work on this land-use calculation must be undertaken to obtain harmonised and consensual estimation of land-use.

Consideration of soil C dynamics in French LCA can also be improved by taking into account the impact of modifications of management practices, for areas that have not undergone LUC. Within the Agri-BALYSE programme time was too short to collect the data required. However, these data exist, for annual crops and grasslands, and are currently being collected in the framework of an INRA –ADEME project and could be used to implement the IPCC 2003 method for a revised version of the Agri-BALYSE database.

6. References

Arrouays D., Balesdent J., Germon J.C., Jayet P.A. Soussana J.F. et Stengel P., 2002 – Contribution à la lutte contre l'effet de serre. Stocker du carbone dans les sols agricoles de France ? Expertise scientifique collective. Rapport. INRA, 332p.

AGRESTE, 2012. Statistiques agricoles annuelles – Occupation du territoire (1990-2010). http://agreste.agriculture.gouv.fr/page-d-accueil/article/donnees-en-ligne.

BSI, 2011. Guide to PAS 2050. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution, London, United Kingdom.

CITEPA, 2012. CCNUCC – Rapport national d'inventaire pour la France au titre de la convention cadre des Nations Unies sur les changements climatiques et du protocol de Kyoto. CITEPA, Paris, France.

Commission Européenne, 2010. Décision n°2010/335/UE du 10/06/10 relative aux lignes directrices pour le calcul des stocks de carbone dans les sols aux fins de l'annexe V de la directive 2009/28/CE.

Doreau, M., Dollé, J.B., 2011. Strategies for reducing greenhouse gas emissions in dairy production. A european perspective. Publication scientifique du Département Techniques d'Elevage et Qualité. http://www.inst-elevage.asso.fr/IMG/pdf_CR_1133009-strategies_reduc_emiss_GES.pdf.

Doreau, M., van der Werf, H.M.G., Micol, D., Dubroeucq, H., Agabriel, J., Rochette, Y., Martin, C., 2011. Enteric methane production and greenhouse gases balance of diets differing in concentrate in the fattening phase of a beef production system. Journal of animal science, 89: 2518-2528.

Dussol, A.M., Hilal, M., Kroll, J.C., 2003. 30 ans de PAC: plus de grandes cultures, moins de fourrages, autant de disparités géographiques, in Agreste: Cahiers n°3, pp: 28-33.

Gerber, P., Vellinga, T., Opio, C., Henderson, B., Steinfeld, H., 2010. Greenhouse gas emissions from the dairy sector. A life Cycle Assessment. FAO, Rome, Italy.

IDF, 2010. A common carbon footprint approach for dairy. The IDF guide to standard lifecycle assessment methodology for the dairy sector. Bulletin of the International Dairy Federation 445. International Dairy Federation, Brussels, Belgium.

IPCC, 2003. Good practices guidance for Land Use, Land-Use Change and Forestry, in: J., Penman, M., Gytarsky, T., Hiraishi, T., Krug, D., Kruger, R., Pipatti, L., Buendia, K., Miwa, T., Ngara, K., Tanabe, F., Wagner (Eds), IPCC National Greenhouse Gas Inventories Programme. IGES, Kamiyamaguchi, pp 632.

JRC, EIS, 2010. ILCD handbook: General guide for Life Cycle Assessment – Detailed guidance.

Leip, A., Weiss, F., Wassenaar, T., Perez, I., Fellmann, T., Loudjani, P., Tubiello, F., Grandgirard, D., Monni, S., 2010. Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS). Final report. European Commission, Joint Research Centre, Ispra, Italy.

Martin, M.P., Wattenbach M., Smith P., Meersmans J., Jolivet C., Boulonne L., et Arrouays D.. 2011. Spatial distribution of soil organic carbon stocks in France. Biogeosciences 8 (5): 1053-1065.

Milà i Canals, L., Rigarlsford, G., Sim, S., 2012. Land use impact assessment of margarine. Int. J. Life Cycle Assess.

Nguyen, T.T.H., van der Werf, H.M.G., Eugène, M., Veysset, P., Devun, J., Chesneau, G., Doreau, M., 2012 Effets ot type of ration and allocation methods on the environmental impacts of beef-production systems. Livestock science, 145: 239-251.

Schulze, E.D., Luyssaert, S., Ciais, P., Freibauer, A., Janssens, I.A., *et al.*, 2009. Importance of methane and nitrous oxide for Europe's terrestrial greenhouse-gas balance. Nature Geoscience 2: 842-850.

Soussana, J.F., Tallec, T., Blanfort, V., 2009. Mitigating greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. Animal, 4:3, pp 334-350.

Using packaging to reduce food environmental impacts for commercial food service: an example

Rita Schenck[1], Matthew Leighton

[1] Institute for Environmental Research and Education
* Corresponding author. E-mail: *rita@iere.org*

ABSTRACT

The very first life cycle assessment was one comparing packaging options (Damar and Nuss, 1969), and studies of food packaging continue to be a common in the LCA field. Part of the reason for that is that approximately one-half of all packaging is used for food, and used packaging is a conspicuous fraction of the solid waste stream in developed countries (Staley and Barlaz, 2009).

With few exceptions, the environmental impact of packaging tends to be much smaller than the environmental impact of the contents of the packaging. The primary function of packaging is to protect and conserve the contents of the packaging, with secondary functions to inform and to track and to market the product. We can then ask the question: can the packaging, by reducing the losses of the contents lead to overall reduction of environmental impacts?

We evaluated the opportunity for food service losses to be mitigated by packaging, comparing frozen food packaging to a shelf-stable retort packaging to no packaging (made from scratch) scenarios, with and without food waste. The recipe for the contents (a beef chilli) was kept constant. In no case was the packaging more than 15% of the life cycle environmental impact, when using the TRACI model. The three packaging scenarios had essentially equivalent impacts per serving, with the exception that photochemical smog formation was higher in the packaged scenarios, due to transportation impacts.

Primary data was obtained for approximately 60% of the mass of the retort packaging and its contents. Secondary data was provided for the remainder and for the other scenarios. Wherever choices were made, they were selected to provide the most conservative impact of the frozen and made from scratch scenarios (i.e., the choice causing the least potential environmental impact of the product system).

We evaluated the scenario when 100 servings were prepared, and different fractions of the servings were not consumed (i.e. were wasted). We assumed that chilli that was heated to serve was discarded, the likely outcome when the menu cycle is more than approximately three days. The retorted package can be heated and returned to storage on the shelf when not opened.

The results showed that the retorted pouch could potentially reduce wastes prior to food service, and this could reduce the overall environmental impact of the food service by as much as 50%. Since the majority of food waste in developed countries occurs in the consumer portion of the life cycle (FAO 2011), this could show substantial improvements in the environmental impact of food service, especially in the case where the number of servings required is uncertain.

This study illustrates how packaging, although causing a small proportion of the life cycle impacts can provide significant environmental benefits when its use leads to less food waste. It highlights the importance of food waste and the need to educate food service organisations of their choices to reduce food waste.

References

Darnar, A. and G. Nuss. 1969. Environmental Impacts of Coca-Cola Beverage Containers.

Gustavsson, J, C. Cederberg and U. Sonneson, 2001. Global Food Losses and Food Waste. Extent, Causes and Prevention. Food and Agriculture Organization, Interpack2011, Dusseldorf, Germany.http://www.fao.org/fileadmin/user_upload/ags/publications/GFL_web.pdf

Staley, B. and M. Barlaz. 2009. Composition of Municipal Solid Waste in the United States and Implications for Carbon Sequestration and Methane Yield. J. Env. Engr. October 2009: 901.

Recycling of aseptic milk packaging and reverse logistic chain accounting

Anna Lúcia Mourad[1,*], Fernando von Zuben[2], Eloísa E. Correia Garcia[1]

[1] CETEA – Packaging Technology Center / ITAL – Institute of Food Technology, Brazil
[2] Tetra Pak, Brazil
[*] Corresponding author. E-mail: *anna@ital.sp.gov.br*

ABSTRACT

The production of the liquid packaging board used for manufacture of aseptic milk packaging has been recently modernised. So, one of the objectives of the present work is to incorporate this data in a previous study considering also its reverse logistic chain. Although post-consumer packaging recycling is a goal pursued around the world, the methodological approach to calculate the environmental changes based on an LCA are not always clearly explained. In this project it was used an open-loop with closed-loop recycling procedure with expanded system boundaries to model the system. The following reductions were obtained for a 27% current recycling rate compared to no recycling: 23% in global warming, 8% in abiotic depletion, 13% in acidification, 15% in photochemical ozone creation, 11% in human toxicity and 8% in eutrophication measured by the functional unit of 1,000 litres of milk packaging. It can be concluded from this project that recycling can still bring about environmental advantages.

Keywords: recycling model, post-consumer packaging, open-loop recycling, system expansion

1. Introduction

Recycling is one of the best options to reduce the environmental impact of products and this objective has been pursued by Tetra Pak for the last two decades (Mourad et al., 2008a). As the production of the liquid packaging board (LPB) used for manufacture of aseptic structure for milk packaging has been recently modernised, one of the objectives of the present work is to measure the environmental benefits due to this significant component. Actually, the reverse chain of milk aseptic packaging is well established and the composite material is almost totally recovered. Cellulose fibre content is recovered by efficient modern hydropulpers which are able to separate wood fibre from the residue of polyethylene / aluminium. The high quality fibre is used to produce new paper products. The residue of PE/Al is partially converted into roof tiles and sheets and partially processed by a plasma technology which breaks down the long polymer structure in the short chain of hydrocarbons obtaining paraffin and also recovers the metal content in high purity aluminium ingots. Although the agricultural phase is usually responsible by most of the environmental impacts of the whole milk chain (Hospido et al., 2003, Berlin, 2002), beverage cartons has a significant environmental impacts such as land use, summer smog and terrestrial eutrophication (Falkenstein et al., 2010).

2. Methods

This study has been conducted in accordance with the recommendations and requirements for conducting life cycle assessment studies set forth in ISO Standards 14040 (ISO 2006) and 14049 (ISO 2000).

2.1 Functional unit

The packaging material system studied was assessed by using a functional unit of 1,000 litres of milk packaging consisting of Tetra Brik Aseptic (TBA) packages with a holding capacity of 1 litre each. The aseptic package is a composite structure of liquid packaging board (LPB), polyethylene and aluminium.

2.2 Energy and virgin packaging materials data

In Brazil, electric energy for public utility services is produced by an interconnected system of electric plants, mostly hydroelectric. Most of the energy used in oil refineries is generated from their own subproducts, i.e. from fossil fuels. The main aspects (consumption and emission) related to the extraction and production of fossil fuels (pre-combustion) such as diesel oil, fuel oil, coal, natural gas and liquefied petroleum gas, have been included within the boundaries of the study. The data concerning the generation and distribution of Brazilian electric energy were collected between 1997 and 1998 and updated in 2000. Site-specific data from 15 companies along with sector energy production data have been combined to build an energy production model (Coltro et al., 2003).

The emissions stemming from truck transportation due to the burning mainly of diesel fuel have been considered for the transport of all the main inputs for the manufacturing of packaging materials, from their extraction, manufacturing of intermediary products up to the obtainment of the final packaging material.
The data relative to the production of LPB were provided by Klabin, the main supplier of the paperboard used to manufacture the TBA packages. The material is produced from pinus and eucalypyus fibres through the kraft process at the Telêmaco Borba plant in the South of the country and its inventory collected for 2008 was employed in this study (Mourad et. al, 2012).
The data concerning the production of ethylene (catalytic cracking of naphtha) from crude oil refining were estimated based on public Brazilian sector data and the study published by Bousted (1992). The manufacturing of low density polyethylene (polymerisation) from ethylene was quantified from the data obtained from the two resin suppliers of Tetra Pak and weighted by the rate of supply in 2003.
The data concerning the manufacturing of aluminum foil refer to data collected between 1998 and 2000 and were provided by all three Brazilian manufacturers of this metal. These data are relative to average aluminium foil life cycle inventory information surveyed by the Project "Life Cycle Assessment of Aluminium Products" (Gatti et al., 2000). The energy required for the production of aluminium is almost entirely produced by hydroelectric plants owned by the aluminium manufacturers.

2.2 Aseptic laminating, filling stages and final disposal

The manufacture of the multilayer laminated plastic and aluminium coated paperboard consists of the printing of the paperboard, followed by its laminating with aluminium foil with extrusion of low density polyethylene (LDPE) and the application of internal LDPE layers, also by extrusion. The transportation steps considered for the manufacturing of the paperboard take into account the distance between the production sites of raw material suppliers (paperboard, polyethylene, aluminium and ink) and the Tetra Pak manufacturing plant located in the town of Monte Mor, SP. The consumption of electric power, vapour and water were based on the characteristics of the Tetra Brik Aseptic – TBA filling machines available on the Brazilian industrial equipment market. The average distribution radius for packaged milk was estimated at 200km, since there are long life milk producers in practically all regions of the country. The final disposal considers that post-consumer packages are recycled or landfilled. The modelling of the final degradation processes of the packages in landfills is the same as that described previously (Mourad et al., 2008b).

2.3 Reverse logistic chain and recycling modelling

The high quality of the fibres contained in the post-consumer packages turns this waste material into excellent raw material for the corrugated paperboard sector. The recycling of these requires hydropulpers specifically designed for this particular purpose. The machine separates cellulose fibres from a mixed material made up of aluminium and polyethylene and recovers around 90% of the total fibre content. The mixed material is partially used to manufacture plates and tiles after previous milling and subsequent hot-pressing (Zuben, 2006) and partially used to recover aluminium and produces paraffin using a plasma process (Szente, 1997).
Using the principles of ISO 14041 and ISO/TR 14049, data from the reverse logistic chain including the recycling processes were collected and an open-loop with closed-loop recycling procedure with expanded system boundaries was used to model the system.
Aseptic packages are collected post-consumption and undergo the first process of recovery of cellulose fibers made by Klabin Company in Piracicaba. This generates polyethylene and aluminum residue (PE/Al).
To model this stage, a hybrid inventory on fiber recovery was built from the data of power consumption provided by Klabin, from data on process yield provided by Tetra Pak relating to the production process of Mercoplas Company in Valinhos in 2011, and data on emissions published in the 2010 corporate sustainability report of Klabin (Klabin, 2011).
Since one of the purposes of the study was to predict the main environmental and benefits accruing from recycling post-consumer packages, the following considerations are deemed important: a) the recycled cellulose fibres from post-consumer milk cartons were reused in the manufacture of new sleeves; b) the aluminium obtained by the plasma process replaces primary aluminium from bauxite in the manufacturing process of aluminium foil and c) the environmental burdens of cracking naphtha used to manufacture ethylene, later on converted into polyethylene, were subtracted by than the paraffin obtained by the plasma process.
The credits of roof tiles co-production was done by the subtraction of the inventory of an equivalent mass of virgin aluminium roof tiles which are replaced by the recycled one. Aluminium recovery by plasma is con-

sidered as being at the boundary of this system as the recovered material has no changes in its inherent properties. The paraffin obtained is used to replace the petrochemical naphtha that is a precursor of polyethylene production. This reverse flow was called "recycled polyethylene" and was applied to discount the use of virgin polyethylene. Fig. 1 shows the boundaries of the system evaluated, along with the recycling model used. This method was used for modelling the system in 2000 with no recycling and the system in 2011 for recycling rates of 0%, 27%, 35%, 50% and 70%.

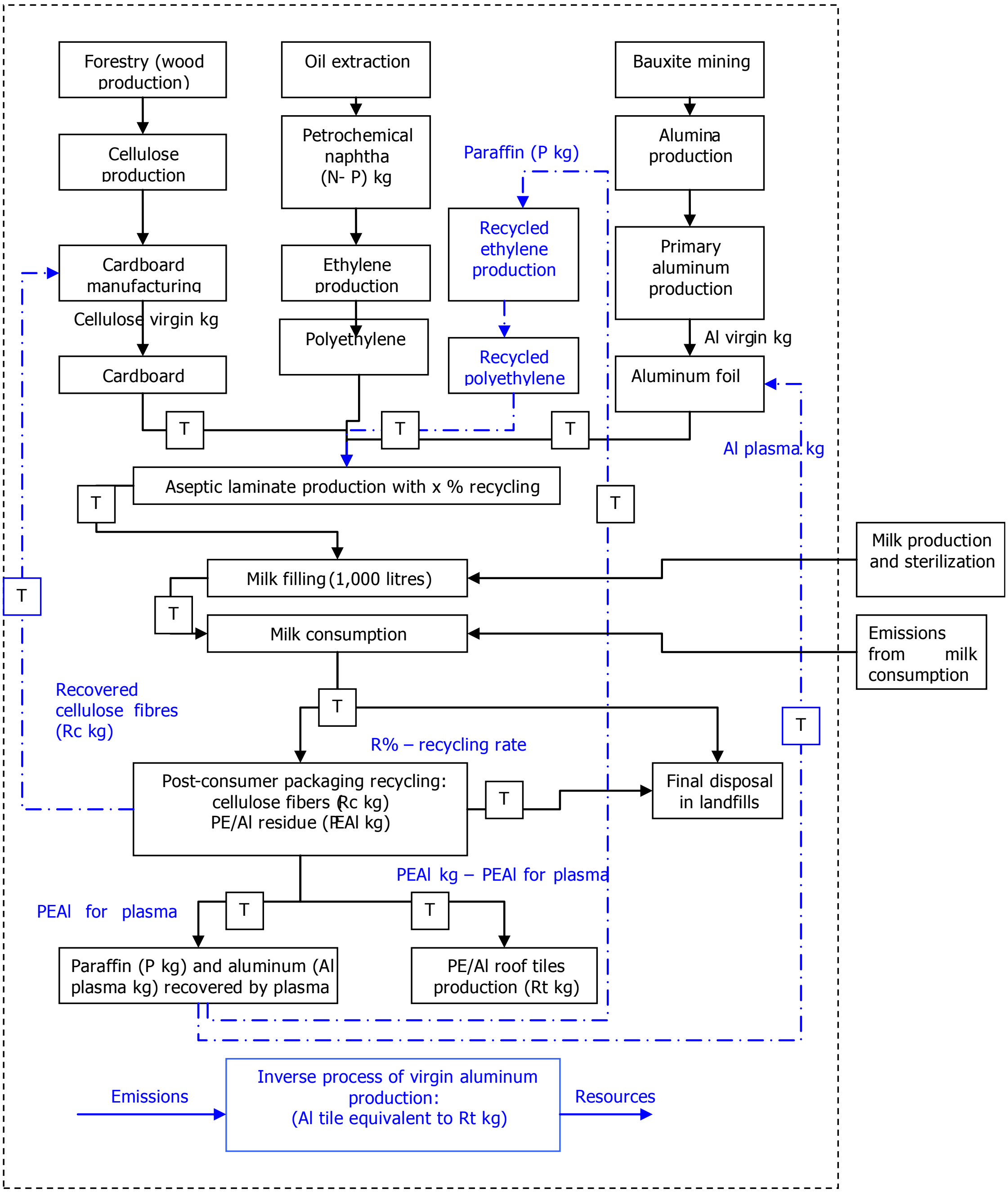


Figure 1. Model used to account the production and recovering of the aseptic milk packaging (open-loop with closed-loop recycling procedure and expanded system boundaries).

2.4 Environmental impact categories evaluated

The inventories obtained were evaluated according to the CML 2001 methodology developed by the Center of Environmental Science at Leiden University (Guinnée et al., 2001) and updated in December, 2007 for the following environmental impact categories and in accordance with the principles of ISO Standard 14044: abiotic depletion potential – ADP (kg Sb equiv), global warming potential – GWP (kg CO_2 equiv.), acidifica-

tion potential - AP (kg SO_2 equiv.), euthrophication potential – EP (kg PO_3^{4-} equiv.), human toxicity potential - HTP(kg DCB equiv.) and photochemical ozone creation potential - POCP(kg ethylene equiv.).
A sensitivity analysis was undertaken for the current recycling rate of 27% in order to measure the influence of the roof tile and paraffin co-production credits.

3. Results

3.1 Energy consumption

The analysis of Table 1 shows that the total energy for the system was reduced by 20% between 2000 and 2011 and that the reduction is mainly due to lower consumption of renewable energy. The present recycling rate of 27% saves 15% of energy compared to no recycling. If the recycling rate increases to a maximum estimate of 70%, the saving of energy can be raised up to 39%.

Table 1. Energy consumption profile of evaluated systems. Functional unit: 1,000 litres of milk packaging.

Year/Recycling rate	Energy (MJ/1,000 litres of packaging milk) Renewable	Non Renewable	Total	% Non Renewable energy	Reduçtion (%) relative to: 2000 0% recycling	2011 0% recycling
2000 0%	1444	728	2172	33.5	-	-
2011 0%	960	783	1743	44.9	20	-
2011 27%	755	728	1483	49.1	-	15
2011 35%	694	712	1406	50.6	-	19
2011 50%	581	681	1262	54.0	-	28
2011 70%	429	640	1069	59.9	-	39

3.2 Environmental impacts

The analysis of Table 2 shows that the system in 2011 had benefits for all the environmental impact categories analysed compared with the same production in 2000. Despite the fact that most transport distances in the country are long and there is a consumption of natural resources as well as air/water emissions in the reverse logistic chain, the increase of recycling brings environmental advantages.

Table 2. Main environmental impact indicators for aseptic packaging systems and different recycling rates. Functional unit: 1,000 litres of milk packaging.

Environmental impact indicator (*)	Reference year / recycling rate 2000 0%	2011 0%	27%	35%	50%	70%
Abiotic resource depletion – ADP (kg Sb equiv.)	0.45	0.38	0.35	0.34	0.32	0.30
Global warming potential – GWP (kg CO_2 equiv.)	189	178	138	125	103	72
Acidification potential – AP (kg SO_2 equiv.)	1.98	0.46	0.40	0.38	0.34	0.30
Euthrophication potential – EP (kg phosphate equiv.)	0.17	0.05	0.05	0.05	0.05	0.04
Human toxicity – HTP (kg de DCB equiv.)	1.56	0.55	0.49	0.47	0.44	0.39
Photochemical ozone creation potential – POCP (kg ethylene equiv.)	0.25	0.09	0.08	0.08	0.07	0.06

*=according to CML 2001 (updated in Dec. 2007)

The results of the sensitivity analysis undertaken for the current recycling rate of 27% are shown in Table 3.

Table 3. Sensitivity analysis related to the roof-tile and paraffin co-production credits in an aseptic packaging system evaluated in 2011. Functional unit: 1,000 litres of milk packaging.

System evaluated	Environmental impact indicators (*) ADP	GWP	AP	EP	HTP	POCP
System with no recycling	0.45	189	1.98	0.17	1.56	0.25
Current approach with 27% of recycling rate	0.35	138	0.40	0.05	0.49	0.08
Without roof-tile co-production credits (27% RR)	0.36	141	0.42	0.05	0.51	0.08
Without paraffin co-production credits (27% RR)	0.35	138	0.40	0.05	0.49	0.08
Without roof-tile and paraffin co-production credits (27% RR)	0.36 (3%)	141(2%)	0.42 (5%)	0.05	0.51(4%)	0.08

RR=recycling rate *= according to CML 2001 (updated in Dec. 2007)

The sensitivity analysis showed that the environmental impacts could increase from 2% (GWP) up to 5% (AP) if the credits were disregarded.

4. Discussion

The closed-loop recycling procedure together with the expansion of system boundaries were used in order to account for all the valuable products from this chain. The model used represents the real situation concerning the production of roof tiles, recovered cellulose fibres, and also aluminium and paraffin recovered by plasma technology. There are some uncertainties related to the credit of aluminium due to the production of roof tiles and the paraffin substitution of the petrochemical naphtha.

The aluminium credit is not totally correct because the main competitor of post-consumer roof tile is cement tile and not aluminium tile (the only inventory available). Paraffin is not a petrochemical naphta, but this approach is a way to account for the reduction in the oil required for ethylene production.

The sensitivity analysis showed that recycling has a favorable environmental profile even without considering the credit for co-products for a 27% recycling rate. In this recycling rate, the maximum increase in environmental impacts analysed was 5%.

However, while the approach used does not represent the real situation and presents minor scientific uncertainties, it is extremely important to find solutions for modelling recycling processes in order to account for all the co-products from the chain. The existence of these co-products makes this reverse logistic chain economically viable.

5. Conclusion

Based on the results of this study, it is possible to state: The increase in the process efficiency mainly due to the modernisation of liquid packaging board manufacturing brought a reduction of 20% of total energy consumption compared with the same functional units in 2000.

The increase in the recycling rate up to 70% could still bring about energy benefits even considering the consumption of new natural resources and air/water emissions related to the logistic reverse chain.

A comparison of the years 2000 and 2011 shows that for the current recycling rate of 27% compared with no recycling the following reductions could be observed: 23% in global warming, 8% in abiotic depletion, 13% in acidification, 15% in photochemical ozone creation, 11% in human toxicity and 8% in eutrophication measured by the functional unit of 1,000 litres of milk packaging.

The chain analysed in this study is not only a hypothetical model but also represents almost all the industrial plants involved in this chain. This is the result of Tetra Pak efforts during the last two decades finding partners who could implement these processes and to make the recovery of their post consumer packages an economic, environmental and socially viable solution.

6. Acknowledgments

The authors are grateful to Tetra Pak by providing data and valuable discussions for this work. Authors are also grateful for financial support of this company, to Fapesp who financed the first LCA study for CETEA in 1998 and for the financial support of travelling expenses.

7. References

Berlin, J., 2002. Environmental life cycle asseessmentg (LCA) of Swedish semi-hard cheese. International Dairy journal 12, 939-953.

Coltro, L., Castilho, G., Garcia, E.E.C., 2003. Life Cycle Inventory for Electric Energy System in Brazil. Int. J. LCA, 8(5), 290-296.

Falkenstein E.V., Wellenreuther F., Detzel, A.(2010) LCA studies comparing beverage cartons and alternative packaging: can overall conclusions be drawn? Int. J. LCA, 15, 938–945.

Gatti, J.B., Castilho, G., Garcia, E.E., 2000. Análise de Ciclo de Vida de produtos de alumínio (Available in Portuguese language, confidential document for internal use), CETEA/ITAL, Campinas. Relatório CETEA P004/00, 88p.

Guinée, J. B. et al., 2001. Life Cycle Assessment – An operational Guide to the ISO Standards. Leiden.

Hospido, A., Moreira, M.T., Feijoo, G., 2003. Simplified life cycle assessment of Galician milk production. International Dairy Journal 13, 783e796.

International Organization for Standardization – ISO, 2006. Environmental management – Life cycle assessment – Principles and framework – ISO 14040. Genève/Switzerland: ISO, 44p.

International Organization for Standardization - ISO 14049, 2000. Environmental management – Life cycle assessment – Examples of ISO 14041 to goal and scope definition and inventory analysis, Genève/Switzerland, 44p

Mourad A.L. et al., 2000. Análise de Ciclo de Vida de Embalagens para o Mercado Brasileiro, Relatório Setorial ABPO – Associação Brasileira de Papelão Ondulado (Available in Portuguese language, confidential document for internal use), CETEA/ITAL; Campinas, 1-120.

Mourad, A.L., Garcia, E.E.C., Vilela, G.B., Zuben, F., 2008a. Environmental effects from a recycling rate increase of cardboard of aseptic packaging system for milk using life cycle approach. Int. J. LCA 13, 140-146.

Mourad, A. L., Garcia, E.E.C., Vilela, G.B., Von Zuben, F., 2008b Influence of recycling rate increase of aseptic carton for long-life milk on GWP reduction. Resource, Conservation and Recycling, 52(4), 678-689.

Mourad, A.L., Silva, H.L.G., Nogueira, J.C.B., 2012. Carton for beverage: A decade of process efficiency improvements enhancing its environmental profile. Int. J. LCA, 17, 176-183.

Szente, R.N.; Schroeter, M.G.; Garcia, M.G.; Bender, O.W. 1997. Recovering aluminium via plasma processing, Journal of Metallurgy, 49/11:52-55.

Zuben, F.V., 2006. Recycling Polyethylene and Aluminium from Aseptic Packaging via Plasma Process. In: Annals of IV Global Conference on Sustainable Product Development and Life Cycle Engineering - Sustainable Manufacturing, 3-6 October 2006, São Carlos, São Paulo, Brazil, CD Rom, 5p (RR14).

A multicriteria analysis for food packaging end-of-life optimisation based on a configurable life cycle assessment approach

Claude Emma Komly[1], Catherine Azzaro-Pantel[2], Antoine Hubert[1], Alan Jean-Marie[1], Valérie Archambault[1]

[1] Altran Research, 2 rue Paul Vaillant Couturier, 92300 Levallois-Perret, France
[2] Université de Toulouse, Laboratoire de Génie Chimique, UMR 5503 CNRS/INPT/UPS, 4 all. E. Monso, 31432 Toulouse Cedex 4, France
[*] Corresponding author. E-mail: *Catherine.AzzaroPantel@ensiacet.fr*

ABSTRACT

A model based on life-cycle assessment results is developed to assess the environmental efficiency of the end-of-life management of a food packaging: polyethylene terephtalate (PET) bottles. The global environmental impacts associated with the treatment of PET bottles from their cradle to ultimate graves (incineration, landfilled, recycling by mechanical, chemical or thermal processes) are computed in function of the flow of bottles in the different valorisation paths. Genetic algorithms are used to solve the resulting multi-objectives. A decision support tool then determines the best compromise among the set of solutions. The model is applied to the case of France in 2010. The variables that minimize simultaneously abiotic depletion, acidification and global warming potential are determined, in particular the number of recycling loops. The approach can be easily adapted to any specific product like bio-based food packaging or organic wastes to find the optimal allocation between valorisation paths.

Keywords: life cycle assessment, food packaging, recycling, multi-criteria analysis, genetic algorithm

1. Introduction

This study proposes the development of a mathematical model based on LCA results to assess the environmental efficiency of the end-of-life management of a common food packaging, i.e. Polyethylene terephalate (PET) bottles. For this purpose, multi-objective optimisation involving Genetic Algorithms and decision support tools were used to define optimal targets for efficient waste management.

This issue is of first importance for food packaging companies which have to deal with their products sustainability all along their life cycle; their products being in competition more and more with tap water.

This model can be easily adapted to any other food packaging, or any specific product like bio-based plastics or organic wastes to find the optimal allocation between valorisation paths, or between supply paths (for instance supply of a bio-refinery), in order to minimize the associated environmental impacts.

2. Methods

The model developed in this work computes the global environmental impacts associated with the treatment of PET bottles (expressed in kg) from cradle to ultimate graves, *i.e.* either incineration or landfill, in function of the flow of bottles in the different valorisation paths.

After each use, PET bottles can be landfilled, incinerated, or recycled by mechanical, chemical or thermal processes. Each recycling process leads to a different product e.g. fibres, films, bottles, chemicals, fuels (Al-Salem et al., 2009)

Fig. 1 shows these different paths, d_i being the decision variables dealing with the waste flow repartition:

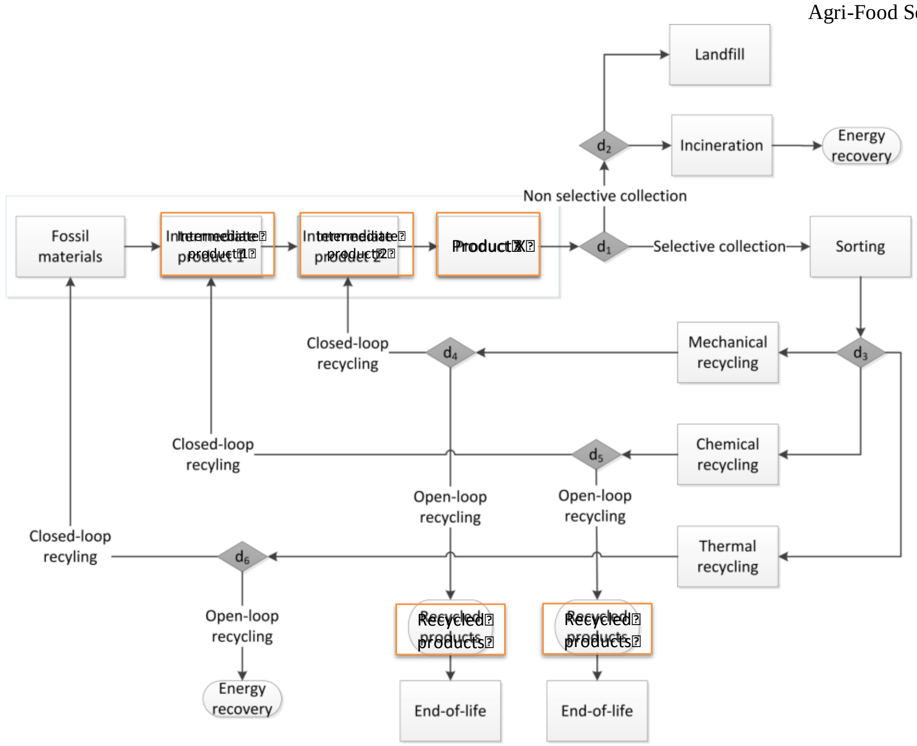


Figure1. The PET bottles waste management system, d_i decision variables (% in weight)

In this study, r-PET regenerated mechanically into a bottle can be further recycled, while the other recycled products (fibre, film, unsaturated polyester) can only be incinerated or landfilled. The fraction of the initial M kg of bottle that is recycled into bottle grade PET is called λ. It is then integrated into a flow of virgin PET and reprocessed into λM kg of bottle, which is submitted to the same recycling alternatives as in its first life, since it is not distinguishable from the virgin flow of bottle. The environmental impacts consequent to these successive end-of-lives are taken into account, until the ultimate graves. The life cycle of packaging waste is considered to be short enough to ignore any temporal evolution of the processes and parameters. The allocation of the flow of bottles is consequently the same for each end-of-life cycle. λ is therefore constant in time and given by the material balance of the system. After n recycling trips, the amount of PET recycled to bottles is given by λ^nM. The valorisation paths could be then simplified as presented in Fig. 2.

Transportation steps are also considered with an average distance between Material Recycling Facility and sorting plant (a value of 400 km is considered in the calculations, yet, the local situation may be very different).

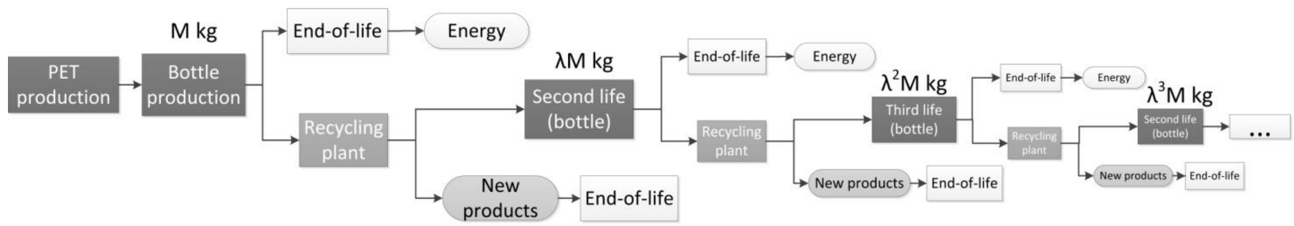


Figure2. An exhaustive LCA: the cradle-to-grave approach

The environmental impacts are based on the calculation of the impacts involved in each elementary process with Simapro LCA software tool, using the CML impact assessment method. The model also takes into account the fraction of PET regenerated into bottles that can be further recycled. A non-linear model for the bottle waste collection stage is also considered; reflecting that the more diffuse the flow of bottles is, the more difficult it is to collect and consequently, the more environmentally impacting.

Finally, the global impacts are the cumulative impacts corresponding to each "end-of-life", as expressed in Eq. 1:

$$I = \sum_{n=0}^{q} I_{l,n} = I_{l,0} + I_{l,1}\sum_{n=1}^{q}\lambda^{n} = I_{l,0} + I_{l,1}\frac{1-\lambda^{q}}{1-\lambda} \qquad \text{Eq. 1}$$

For each impact l, the total impact I_l is expressed as the sum of impacts caused by each end-of-life, $I_l = \sum_n I_{l,n}$, n being the life cycle number. The $I_{l,0}$ are the impacts resulting from the virgin life of the product, from the extraction of the raw materials to its discarding. $I_{l,1}$ are the impacts resulting from the first life of the used product, from its cradle (bottle waste collection) to its different graves.

Global Warming Potential (GWP 100), abiotic depletion and acidification have been selected among the indicators: ozone depletion and photochemical oxidation are excluded due to the low emissions associated to the processes considered. Toxicity and ecotoxicity have not been considered as well due to the important uncertainty in the calculations for several reasons: the synergies between pollutants are not considered, LCI are often incomplete and uncertain, and characterisation factors are lacking for many pollutants.

The resulting multi-objective problem is to find the allocation of bottles between valorisation paths that minimizes the environmental impacts of bottle end-of-lives. It is solved using a genetic algorithm (Ouattara et al., 2012), and the trade-off between environmental impacts is illustrated through Pareto curves (Fig. 3).

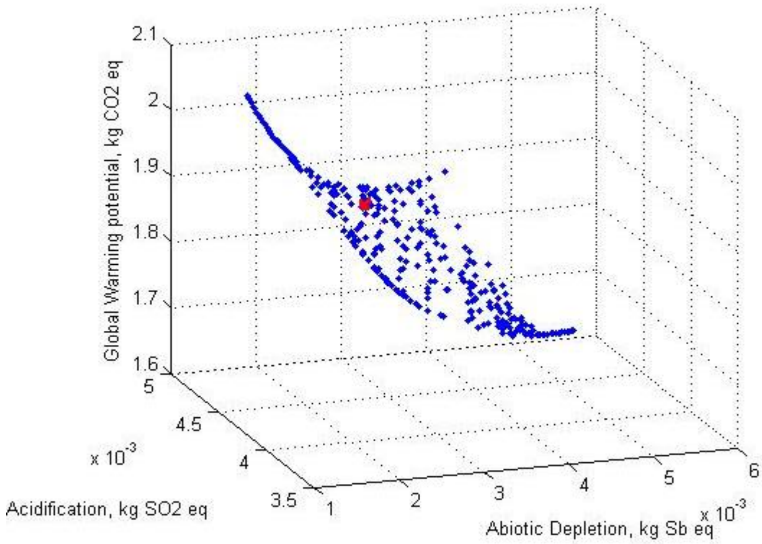


Figure 3. Triobjective optimisation results, in the case of multiple recycling loops (red point shows the best compromise found with TOPSIS method).

A decision support tool involving a variant of the so-called TOPSIS method (Technique for Order Preference by Similarity to Ideal Solution) then determines best compromises among the optimal solutions (Ren et al., 2007).

3. Results

The model has been applied first to the case of France in 2010, with the dedicated values of decision variables d_i (Fig. 1) given in the literature (ADEME, 2002; RDC-Environnement, 2010; Valorplast, 2011). When considering multiple recycling for PET bottles, the best solution is to collect 87% bottles and to regenerate all of them into bottles by mechanical recycling. In this case, abiotic depletion, acidification and GWP impacts respectively decrease by 141, 72 and 61%.

We then applied the model to the Ile-de-France region with its current waste treatment industrial sites. The PREDMA regulation is aiming at a 60% waste collection rate in 2014 and 75% in 2019. Results in Figure 4 show that infinite recycling loops reduce abiotic depletion, acidification and GWP respectively by 64.5, 65 and 65%; meanwhile a 75% collection rate leads to a decrease of abiotic depletion, acidification and GWP respectively by 28.5, 25 and 115%.

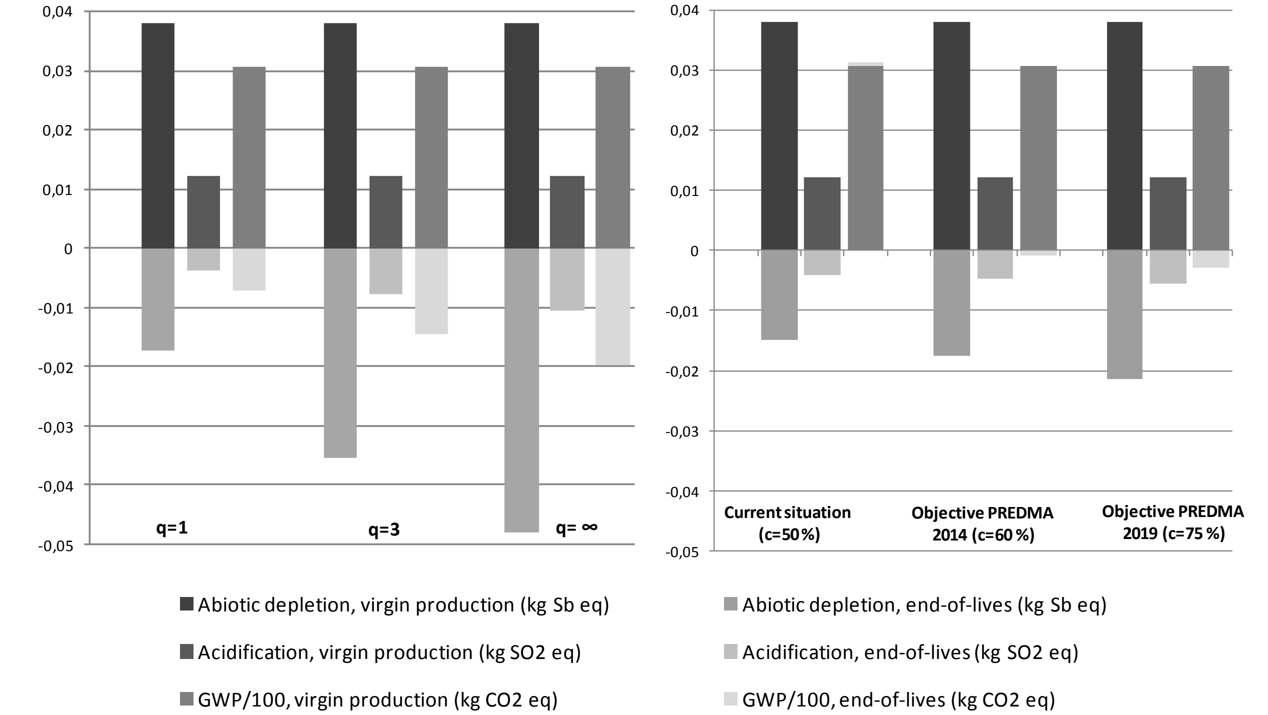


Figure 4. Effect of recycling loop number ($q \in [1;3]$) and collection rate (c = 50% ; 60% ; 75%) on the environmental impacts assessment of PET bottle waste treatment in Ile-de-France region. The unit of the y-axis is directly expressed in the unit of the considered impact.

The effect of multiple recycling trips for PET bottles on the final result is analysed (Fig. 5). With the France parameters, multiple recycling trips allow a reduction of abiotic depletion (resp. Acidification) of 11.1% (resp.10.9%) more than if there is only a single end-of-life. GWP decreases by only 3.4%. The effect of multiple recycling trips is significant from the third end-of-life (Fig. 5).

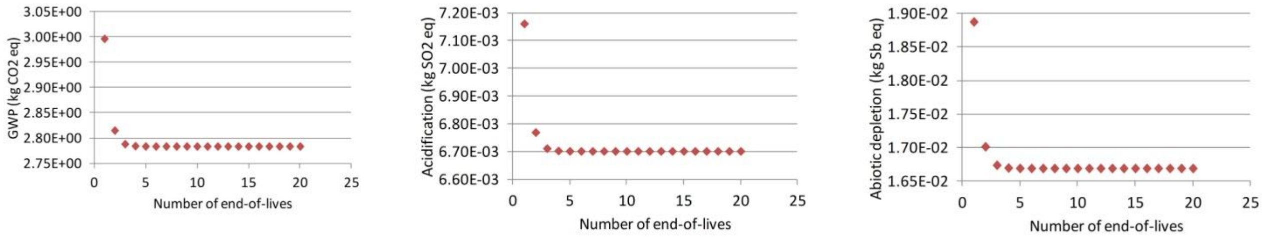


Figure 5: Effect of the number of recycling loops on the final impacts.

4. Discussion

4.1. Evaluation of the recycling routes.

First, it must be said that the three scenarios are in agreement with the literature. With the data used, either secondary or tertiary recycling is always preferable to thermal recycling. This is not surprising since thermal recycling techniques were not created to treat PET (ADEME, 2002), and are more adapted to composite waste flow that cannot be sorted. However, the data used for the LCI of pyrolysis and hydrocracking could be outdated as underlined in Perugini et al., 2005. Indeed, this range of recycling techniques suffers from a general lack of data concerning environmental impacts as highlighted in Al-Salem et al., 2009. The hypothesis on hydrogen production is even not relevant in ecoinvent: in this database used for our impact assessment, the hydrogen production is an average of different process, water electrolysis mainly, which is a non-sense or a continuous production. In reality, almost all industrial sites worldwide like refineries do use stream reforming of natural gas followed by water gas shift for hydrogen production (Chaumette, 1996; Raimbault, 1997; Yurum, 1995). Then, impacts assessment of hydrocracking is underestimated in our model, and those of pyrolyis are overestimated.

Second, for closed loop recycling, mechanical pathway is preferable to glycolysis followed by repolymerisation. This difference had already been demonstrated in (Shen et al., 2010) in the case of bottle to fibre recycling. Chemical recycling has long been the only possibility to achieve closed loop recycling, but since the beginning of the century mechanical processes have reached the necessary requirements, in terms of mechanical propriety of the polymer and decontamination of the matrix (Welle, 2011). Mechanical processes are also cheaper than chemical depolymerisation (Karayannidis et al., 2007; Awaja et al., 2005). This explains the renewal of interest for original applications of glycolysed PET with higher added-value, like unsaturated polyester or polyurethane. Unsaturated polyester synthesis from glycolysed PET seems to be an interesting environmental solution to avoid the used of several chemicals when multi-recycling is not possible. However, the data originating from a pilot plant, additional research is necessary to ensure the viability of the process and confirm the results of this study.

4.2. Multi-recycling loops and quality

It was also checked that the recycled bottle quality was not affected by successive bottle-to-bottle recycling, meaning that intrinsic viscosity and colour remain above the quality threshold standards. For this purpose, a property Q has been considered, e.g. intrinsic viscosity. After each recycling cycle, the property of the PET resin is degraded, and the relation between cycle n+1 and n is given by $Q_{n+1}=\alpha Q_n$ (Rieckmann et al., 2011), α being the quality retention rate. The propriety after n closed-loop recycling is therefore a geometric sequence and $Q_n = \alpha^n Q_0$. k is the fraction of r-PET introduced in a bottle. This fraction is a mix between PET recycled once, twice... q times, as represented in Fig. 6.

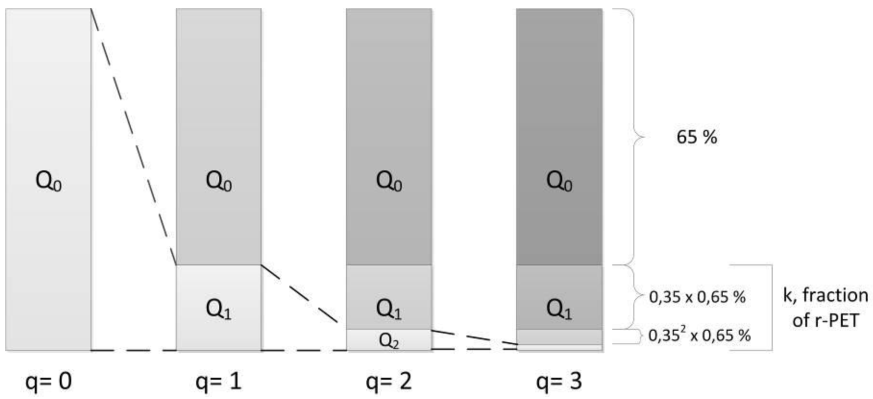


Figure 6. Mix of resins in a PET bottle. Q_n represents the quality, linked to the number of times this fraction has been recycled, Q_0 being virgin PET and q the number of recycling trips.

The quantity of resin recycled n times introduced in a bottle is equal to $f_n = k^n(1 - k)$ (see Fig. 5 for q = 1.4). Under the hypothesis that the blend of recycled and virgin PET obeys the law of mixes, the final quality Q of a bottle containing r-PET recycled up to q times is given by Eq. 2:

$$Q = \sum_{n=0}^{q} f_n \times Q_n = (1-k)Q_0 \sum_{n=0}^{q} k^n \alpha^n \qquad \text{Eq. 2}$$

When k = 0.35 and α= 0.95 (in the case of intrinsic viscosity, r is near to 0.98 according to the results in Rieckmann et al., 2011), for $q \rightarrow +\infty$, Q quickly converges towards $\approx 0.973.Q_0$. The hypothesis of infinity closed-loop recycling therefore does not imply serious degradation of the quality of the bottles considering current practices.

4.3. Economic viability

The environmental performance of the different processes is not the only argument that has to be considered to design PET waste management system. A low environmental impact solution that is not economically viable will not be selected. The difficulty of modelling the economical flows of the PET waste management system relies on the fact that involved stakeholders generally have conflicting interests, economical profits or costs being different for industrials, municipalities and citizens. A global approach similar to the one adopted for environmental impacts is therefore of limited interest. The model developed in this study is a useful tool to select most interesting processes in terms of environmental impacts. This process being selected, its economical viability must then be tested by classical methods: Net Present Value, annual cost (Ouattara et al., 2012).

5. Conclusion

The model of the PET packaging management system proposed here is particularly interesting to quantify the environmental impacts and determine optimal targets. The use of a genetic algorithm is an efficient method to define the optimal allocation of bottle waste between valorisation paths. The use of multi-recycling loops for PET bottles reduces significantly the environmental burdens. According to the number of recycling trips possible, the optimal collection rate varies between 80 and 90% which is in average ten percent higher than the target values fixed for 2019 in France.

This could lead to new strategies for food packaging companies like developing new partnerships with local communities and waste treatment companies.

Further work is now devoted to the application of this methodology both to LCA of other food plastic packaging (polylactic acid (PLA) bottles and bio-based PET bottles), other food packaging (paper, glass or metals) and to the optimal material supply for bio-based ethanol synthesis used in different biomaterials by comparing different crops.

6. References

ADEME, 2002. Recyclage chimique des matieres plastiques. Technical report, ADEME.

Al-Salem, S.M., Lettieri, P. and J. Baeyens, J., 2009. Recycling and recovery routes of plastic solid waste (PSW): A review. Waste Management, 29(10):2625-2643.

Awaja, F. and Pavel, D., 2005. Recycling of PET. European polymer journal, 41(7):1453-1477.

Chaumette, P., 1996. Conversion du gaz naturel in Revue de l'IFP N°5, 51(5), pp.711-727.

Karayannidis, G.P. and Achilias, D.S., 2007. Chemical recycling of poly-(ethylene terephthalate). Macromolecular Materials and Engineering, 292(2):128-146.

Ouattara, A., Pibouleau, L., Azzaro-Pantel, C., Domenech, S., Baudet, P., Yao, B., 2012. Economic and environmental strategies for process design, Computers & Chemical Engineering, 36(10):174-188.

Perugini, F., Mastellone, M.L. and Arena, U., 2005. A life cycle assessment of mechanical and feedstock recycling options for management of plastic packaging wastes. Environmental Progress, 24(2):137-154.

RDC-Environnement, 2010. Analyse du cycle de vie d'une bouteille PET. Technical report, Elipso, Valorplast and Eco-Emballages.

Raimbault, C., 1997. L'hydrogène industriel : synthèse, purification, L'actualité chimique.

Ren, L., Zhang, Y., Wang, Y., and Z. Sun, Z., 2007. Comparative analysis of a novel m-topsis method and topsis. Applied Mathematics Research Express, 10.

Rieckmann, T., Frei, F. and Volker, S. 2011. Modelling of PET quality parameters for a closed-loop recycling system for food contact, in Macro-molecular Symposia, volume 302, pp. 34-45. Wiley Online Library.

Shen, L., Worrell, E. and Patel, M.K. Open-loop recycling: A LCA case study of PET bottle-to-fibre recycling. Resources, Conservation and Recycling, 55:34-52, 2010.

Valorplast, 2011. Applications des bouteilles et flacons plastiques recycées. URL http://www.valorplast.com.

Welle, F., 2011. Twenty years of PET bottle to bottle recycling-an overview. Resources, Conservation and Recycling, 55(11):865-875.

Yurum, Y., 1995. Hydrogen Energy System, Kluwer Academic Publishers.

Environmental aspects used for food packaging design and product carbon footprint assessment

Carlos Naranjo[1,*], Juan Leal[1], Oscar Agudelo[2], Aura Ochoa[2], Octavio Morales[3], Diana Guevara[4]

[1] Gaia Environmental Services, Medellín, Colombia
[2] Industria de Alimentos Zenú S.A.S, Medellín, Colombia
[3] Microplast S.A., Medellín, Colombia
[4] Phoenix Group, Bogotá, Colombia
[*]Corresponding author. E-mail: *cnaranjo@gaiasa.com*

ABSTRACT

In developing new designs for long-shelf-life food packaging, this study took a product life cycle into account, with the product carbon footprint as an indicator. Technical and market variables were also included in the evaluation. The assessment examined four different packages for sausage products: aluminium can, rigid plastic, flexible 100% plastic pouch, and flexible aluminium pouch. The methodologies used for the study were ISO14040, ISO14044 and PAS2050:2008. The aluminium can had the largest carbon footprint, while the rigid plastic container and flexible plastic pouch had the lowest carbon footprints (80% and 60% lower than the aluminium can, respectively). The Aluminium can carbon footprint is four times higher than plastics ones.

Keywords: eco-design, carbon footprint, consumer behaviour, food packaging.

1. Introduction

"Packaging is necessary to protect food during storage and transportation in the food chain from producer to consumer. It also plays an important role in creating a product brand and in communicating with the consumer." (Williams, 2008)

The processed meat industry sells its products through retailers, such as supermarkets and grocery stores. Modern lifestyles and social changes have demanded that the processed meat industry develop new packaging that adds convenience and longer shelf life to the product. Particularly important is the development of packaging that provides individual portions and makes refrigeration unnecessary.

The main function of food packaging is to protect the product from spoilage and outside contamination. The packaging also provides other information, communicating to consumers the identity of the product and its particular characteristics.

The motivation can be government regulation, consumer pressure, retailer pressure, cost, or functionality of the package which improves the convenience of the final product. If a company wants to use sustainable packaging, the life cycle assessment (LCA) methodology represents the "perfect tool" for working on such procedures (Siracusa, 2011).

This study aims to assess the carbon footprint of four types of packaging for sausage products. The aluminium can, it is an existing package, and for this project the company development three prototypes in rigid plastic, flexible 100% plastic pouch and flexible aluminium pouch. In developing new designs for long-shelf-life food packaging, this study takes a product life cycle into account, with the product carbon footprint as an indicator. Technical and market variables are also included in the evaluation. The expected shelf life for the containers without refrigeration is two years.

2. Methods

Prototypes of the three containers were manufactured in a pilot plant and compared to an existing aluminium canned sausage product (Fig. 1). The containers were sent to a laboratory for testing and comparison of technical and sensory parameters:

- pH, texture profile (TPA), colour, sensory test and shelf life.

Samples were also sent for a marketing study with different target groups in three major markets:

- Use Test: a qualitative evaluation for the performance of each prototype in the location that consumers typically used the products. The target market included homemakers, working women and those responsible for household purchases.
- Channel Distribution Test: a qualitative evaluation through in-depth interviews with store owners and school store managers.
- Concept Test and Sensory Test: evaluation with men and women aged 15-50 of middle and high class.

LCA to estimate a carbon footprint of each product was accomplished following the methodology ISO14040, ISO14044 and PAS2050:2008. The manufacturer of each package provided design and testing prototype information. The consumption of tap water, steam and electricity, as well as packaging disposal was considered in the final analysis. The functional unit for the LCA was "packaging able to hold 180 g of sausage". The data for the life cycle of each container, including raw materials; packaging manufacture; transportation; and food production processes, distribution, use and disposal were modelled with Umberto software and the Ecoinvent 2.2 database. The raw materials are imported for Colombia, and the data for the carbon footprint were taken from the Ecoinvent database 2.2 The data to build the Inventory for the raw material processings were measured.

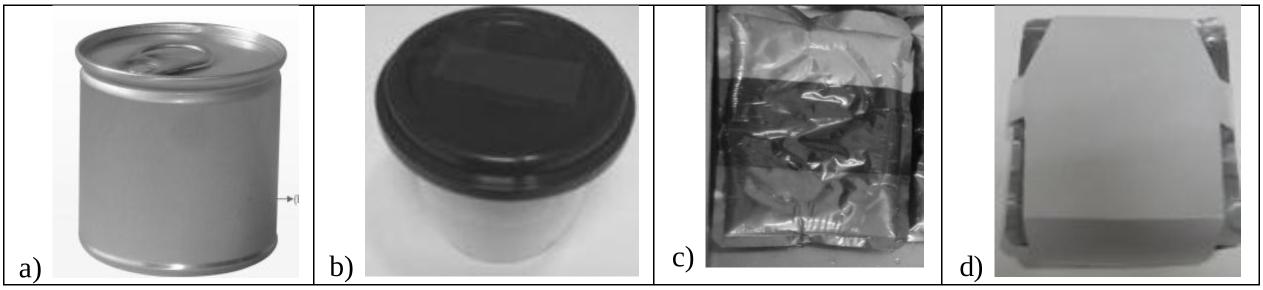


Figure 1. Package prototypes tested: a) aluminium can, b) rigid plastic, c) plastic pouch, d) aluminium pouch.

3. Results

3.1. Technical analysis

The alternate containers showed no significant differences with the existing packaging design for pH, TPA, colour or sensory parameters. However, shelf life in the flexible plastic pouch was 60% less than the existing container; shelf life in rigid plastic container was 58% less than the existing container, while the flexible aluminium pouch provided 50% less shelf life.

3.2. Marketing analysis

The flexible plastic pouch could not be studied because it had leaks. Marketing analysis revealed some positive and negative perceptions of the remaining three containers (Table 1).

Table 1. Main results of the marketing analysis of three container prototypes.

Packaging type	Positive perceptions	Negative perceptions
Aluminium can	Safest method of packaging. Protects against all outside contamination and spoilage.	The packaging has not changed over time. No consumer excitement.
Rigid plastic	Reusable packaging. Attractive new design.	Similar packaging as for refrigerated products. Less resistant to outside contamination as the container may break. Lower price expectancy not met.
Flexible aluminium pouch	New innovative packaging.	Difficulty opening. In some cases people need to use scissors. Consumers dislike putting their hands inside the bag to remove the product. Consumers believe the bag product should have a lower price. Consumers demand that the product be refrigerated. It is not easy to stack and consumes a lot of shelf space.

3.3. Environmental analysis: carbon footprint of the packaging

The raw materials used had a large influence for the carbon footprint of the packaging. For example, for the aluminium pouch, 95% of the carbon footprint came from raw materials, and aluminium was the main material.

To model the carbon footprint of the different packagings the Umberto for carbon footprint software was used (Fig 2).

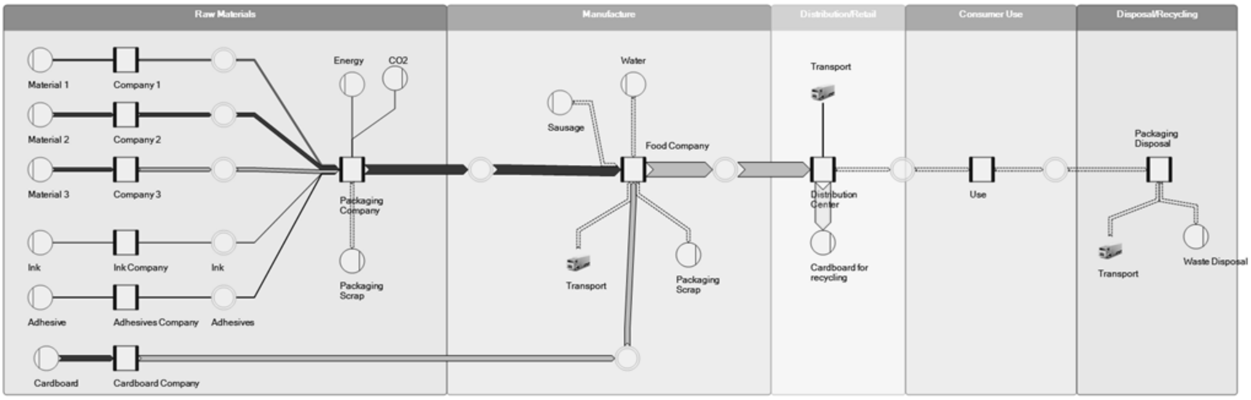


Figure 2. Life cycle carbon footprint flow for plastic pouch. Umberto for Carbon footprint v1.2 ®

For the rigid plastic container, 63% of the carbon footprint came from raw materials, mostly polypropylene. The rigid plastic container's three components, a polypropylene cup, an aluminium lid and a polypropylene cap and fork. For the carbon footprint of the rigid plastic container the polypropylene cup represents 24%, the aluminium lid 17%, and the polypropylene cap and fork 22%. For the flexible plastic pouch, 60% of the carbon footprint came from raw materials. For the flexible aluminium pouch, 95% of the carbon footprint came from raw materials, and the aluminium is 30% of the carbon footprint of the product. The corrugated box is 20% of the carbon footprint of the product. The corrugated cardboard box used to ship packaged products it is about 30% of the carbon footprint of all the different packages. The mass of the empty corrugated box is 255 g and holds only 23 flexible packages, 23 rigid containers, or 48 cans. The aluminium can have the largest carbon footprint, while the rigid plastic container and flexible plastic pouch had the lowest carbon footprints (80% and 60% lower than the aluminium can, respectively) (Fig. 3).

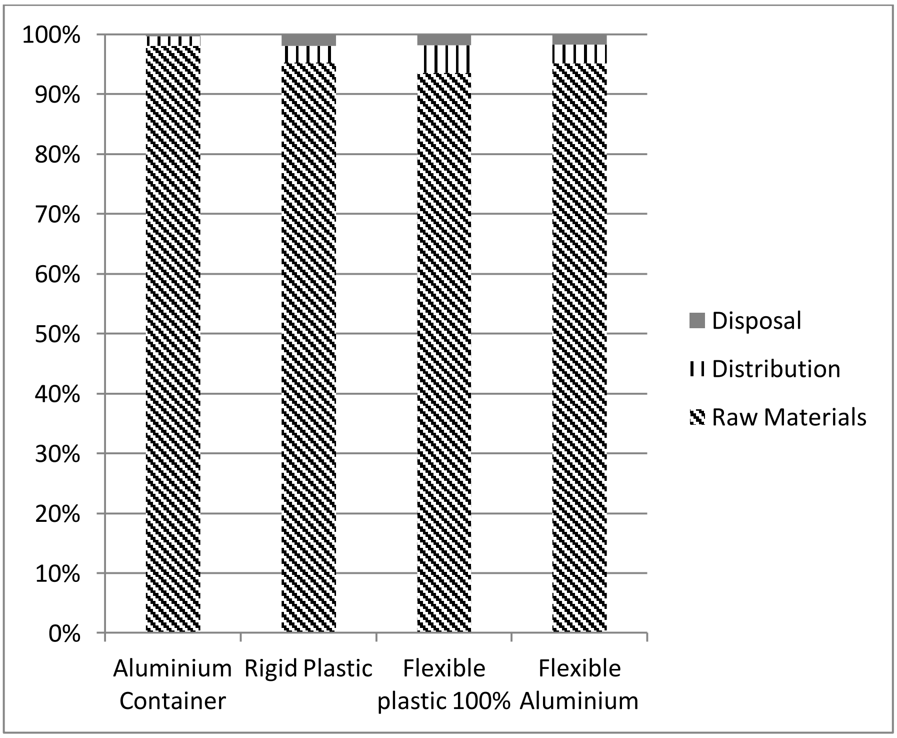


Figure 3. Relative carbon footprint of the "packaging able to hold 180 g of sausage".

4. Discussion

One important consideration for eco-design packaging is the marketing study. It will be a starting point for future eco-designs. The marketing study related to multiple aspects of public perception must be translated into design variables to achieve effective consumer satisfaction with the new packaging. The main costumer perceptions were concern with preserving and maintaining the quality of the product as well as the price. The aluminium can received positive comments for its ability to protect and preserve the product but the production cost is higher than other ones. However the consumers are willing to pay more for it.

5. Conclusion

- The carbon footprint of product is a tool that includes eco-design criteria in food packaging and can be used for marketing.
- Raw materials represent 50-80% of the carbon footprint of these packages.
- The Aluminium can carbon footprint is four times higher than plastics ones. Nevertheless, it provides the safest method of protecting and preserving the product as well as providing the longest shelf life.
- The marketing study reached an important conclusion, as consumers prefer the can container for the long life sausage; it will be a deciding factor for the packaging equipment design.
- All materials meet quality specifications, but the shelf life of the sausage is the most important factor to consider, as plastic packaging leaves the product with less shelf life.

6. References

Carbon Trust, DEFRA, BSI. PAS2050:2008. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. BSI, London. 2008

ISO. ISO 14040:2006. Environmental management -- Life cycle assessment -- Principles and framework. ISO, 2006.

ISO. ISO 14044:2006. Environmental management -- Life cycle assessment -- Requirements and guidelines. ISO, 2006

Siracusa V et al., Life Cycle Assessment of multilayer polymer film used in food packaging field. In: Procedia Food Science 1 (2011) 235-239.

Williams H et al., A life cycle perspective on environmental effects of customer focused packaging development. Journal of Cleaner Production. 16, 853-859.

The influence of packaging attributes on consumer behaviour in food-packaging LCA studies - a neglected topic

Fredrik Wikström[1], Helén Williams[1], Karli Verghese[2], Stephen Clune[2]

[1] Karlstad University, Sweden
[2] Centre for Design, RMIT University, Australia
[*] Corresponding author. E-mail: *fredrik.wikstrom@kau.se*

ABSTRACT

It is well known that consumers waste a significant amount of food product and that the functions of packaging can influence how much food is wasted. Examples include portion sizes, accessibility of the pack and ability to dispense the product. The role of packaging systems to reduce food waste is rarely modelled in LCA-studies. This means that a packaging system format with a lower environmental impact, but that causes high food waste, appears to be a better alternative than a packaging system format with a higher environmental impact, but reduces food waste. This can be contradictory to the purpose of using LCA to reduce the overall environmental impact of the packaged food product, because food has generally much higher environmental impact than the packaging. Ensuring that LCA studies of packaged products include associated food wastage across the supply chain can drastically change the outcome of an LCA study.

Keywords: functional unit, consumer behaviour, packaging,, food waste, life cycle assessment

1. Introduction

Attention to date has seen the focus of reducing the environmental impacts of the sourcing, production and waste management of packaging materials. This has included lighweighting of materials, changing packaging system formats, providing different portion sizes and enhancing the efficiency of material and energy consumption in the sourcing, production and conversion of these materials. Regulatory frameworks including the Directive on Packaging and Packaging Waste in Europe (European Council, 1994; European Commission, 2006) and the voluntary Australian Packaging Covenant (APC 2010) have provided mechanisms for the packaging supply chain to rethink the design of packaging materials and formats to reduce environmental impact. Packaging performs important functions in the provision of food including containment, protection, distribution, marketing of the brand and dispensing of the product. However, it is known that about 30% of the food purchased in the industrialised world is wasted by consumers (Engström and Carlsson-Kanyama, 2004; Ventour, 2008; Quested and Johnson, 2009). Understanding the reasons for this wastage and the role that packaging can play in reducing this waste is important and the remainder of this paper begins to investigate these reasons.

If LCA methodology is used as a tool to reduce the environmental impact of packaging, the impact on food waste and possibilities to reduce this should be included. However, the function of packaging to reduce food waste is rarely discussed in the Packaging Directive (European Council, 1994). Packaging plays an important role in reducing food waste, as Williams et. al's (2012) estimated, 20% of food waste in households could be attributed to packaging (not including food waste of fruit and vegetables due to too little packaging).

By excluding food waste when estimating the environmental impact of packaging systems means that packaging with a lower environmental impact that causes high food waste, may appear to be a better alternative than packaging with somewhat higher environmental impact, but reduces food waste. This is contradictory to the purpose of using LCA to reduce environmental impacts, because food generally has a much higher environmental impact than the packaging (Hanssen, 1998). For example, 1 kg of beef has approximately 1,730 times more global warming potential than a 5 g LDPE plastic bag (extrapolated from Eady et al., 2011, p.1; Ecoinvent 2011). If it was found that the packaging configuration of the plastic bag resulted in an increase in product wastage, a change in packaging material format to say a rigid plastic tray may be a better option. It would be important to understand the relationship between packaging format and food waste to ensure that optimisation of the product-packaging system as a whole is achieved. In this particular scenario, while a rigid tray would be heavier (e.g., 22 g), if it delivered a reduced product wastage rate then the overall environmental impact would be lower through savings in beef waste. Williams and Wikström (2011) found a similar scenario in the case of bread whereby the climate impact of bread packaging could be doubled, if it led to a reduction in bread waste by 5 percent .

The importance of consumer behaviour in a food LCA is sometimes stated (e.g. Verghese et. al., 2012a) but seldom included. There can be several reasons for this. The heterogeneous behaviour, including preferences and conditions among consumers is difficult to handle along with scarce knowledge of how the design of different food products, including packaging, affects consumer behaviour. However the broader design literature does acknowledge the 'scripting' role of designed goods (Jelsma 2006). Product attributes enable or restrict consumers to act in a particular way. Jelsma posits that we may design 'moralised products' that encourage consumers to act in the most desirable way. For example, packaging that reseals properly after opening may script a reduction in waste. If one acknowledges that packaging attributes contributes to or ameliorates food waste then more comprehensive packaging LCA's including food product and food waste are desirable. This requires an 'upscaling of the functional unit'(Verghese et al., 2012a) to become the delivery of consumed food.

'Upscaling the functional unit' to include food waste presents methodological challenges for LCA, in that it is hard to imagine how the user phase can be standardized to make LCA studies comparable. This also raises questions of why LCA is used. Is it to improve environmental performace or to compare products? The purpose of this paper is not to solve these problems, but to demonstrate how packaging attributes affect consumer behaviour and food waste, and thus should influence the outcome from a LCA study on packaging. Our intention is to establish this opportunity on the agenda for the future discussion on how this knowledge should be implemented into the LCA methodology for packaged food.

The structure of the paper first provides a background to issues of consumer food waste and introduces fourteen packaging attributes that can affect food waste.Streamlined LCA case studies are presented to illustrate the importance of considering food waste and packaging systems in conjunction with the food product. The paper closes with a discussion of possible methods that could assist in including food waste within LCA studies.

2. Background: packaging attributes that assist in reducing food waste

Recent studies in the United Kingdom suggest that up to a 1/3rd of food that is purchased is thrown away by the consumer, with 72% of the discarded food potentially avoidable, classified as either avoidable (57%) and possibly avoidable (16%) (WRAP 2009, p.5). This suggests that theoretically 24% of all money spent on food is wasted (1/3rd food wasted x 72% potentially avoidable). The highest wasted foods, classifed by WRAP, by category were salad items, bakery goods, fruit and vegetables. The reasons identified as to why food was wasted are outlined in Table 1.

Table 1. Reasons for food waste and packaging attributes that may ameliorate. Source: Ventour (2008, p. 139)

Reason for food waste	% weight	Description	Potential packaging interventions
Inedible	36.5	Bones, hard fruits, tea bags	Debone prior to packaging
Left on plate	15.7	Not eaten after the meal	Portion sizing
Out of date	15.1	Past 'used by' or 'best before' date	Portion sizing combined with labelling
Mouldy	9.3		Portion sizing, chemical protection & resealability
Looked bad	8.8		
Smelt/tasted bad	4.5		Chemical protection & resealability
Left from cooking	4	Not served up onto plate	Assist in portion sizing
In fridge/cupboard too long	1.5		Portion sizing,
Freezer burn	.5		Mechanical protection
Other	4.1		

With an understanding of the top ten reasons for food waste and potential packaging interventions, we turn our attention to looking in more detail at some of the numerous benefits that packaging materials and packaging systems play in containing, protecting and delivering food. One of the core functions of packaging is to protect the product. A wide range of packaging attributes exist that may ameliorate food waste. Williams et al., (2008) identified a range of packaging attributes that may affect food waste, see below.

Packaging should also provide for *mechanical protection*. The packaging should not leak and it should protect fragile products from mechanical impact. The packaging must resist pressure, strikes, rips and should facilitate ease of handling and stacking at the retailer, home transport and storage and handling at home. Packaging also offers *physical-chemical protection* of product, such as protecting the product against oxygen, water or other agents from the surrounding atmosphere. This can be achieved by introducing different kinds of barriers in the packaging material or by a modified atmosphere. These solutions can extend the time that the product is fresh.

The attribute *resealability* can affect physical-chemical protection by avoiding degradation of food in an opened packaging, for example a packaging placed where it can incorporate odour from other food products and result in a reduction of experienced consumer quality. A better resealability can also help to avoid spillage during consumer handling in home or "on-the go".

Spillage during handling could also be avoided by the attributes *easy to open*, *grip*, *dose* and *empty*. Packaging is normally handled by many different consumers, children, elderly, people with reduced strength in their hands, visually impaired, etc. About one-third of a group of elderly reported that spillage occurred frequently in connection with opening (Duizer *et.al.*, 2009). The design of the packaging's opening, the shape and the surface of the packaging can affect how much food is wasted. Packaging that is too large or too heavy can also increase the risk for spillage. A smooth surface can be safer to grip by using laminate on the surface, making creases in board packaging or by making the surface ribbed. The attribute easy to dose may be improved by e.g. introducing a spout mechanism. The easiness to empty can be influenced by surface treatment inside the package, possibility to reach all food in the package, and ability to mechanically squeeze the last food out of the packaging.

Contains the correct quantity is an important function of packaging and also environmental attribute. If the food quantity in a package is higher than the turnover of the food item in the household, the risk that the food item is wasted increases, either because of physical degradation of food, or because the product is out-of-date (see below). In a Swedish food waste diary study, the households documented "too large packaging" as one important factor for food losses (Williams *et.al.*, 2012). If the quantity of product in a package is slightly more than desired, there is a possibility that it increases the surplus that is wasted directly, or worse, too much food is prepared and wasted after the meal. The waste of prepared food can be significant in households (ibid.). As the amount of single households and elderly increases in many countries, it is especially important to offer suitable packaging sizes to avoid food waste. Williams *et. al.* (2012) noted that the amount of wasted food per person was higher in households with few persons.

Food safety/freshness information is also important. One of the most important reasons for food waste is consumer confusion about date coding. "Best before", "Sell by" and other dating that indicate the premium quality period are treated as dates when the food should be thrown away. These misconceptions cause substantial food waste, either at retailer (food items with "short" dates are rejected) and at home. Food waste could probably be reduced by more with better information on the packaging that explains the dating system, if and when the food item could be unhealthy, and how the consumer could judge the quality of the food item. Introduction of smart labels or 'intelligent indicators' that indicate when the food item is safe/of high quality is also a possibility (WRAP 2007, p. 37). Other information channels other than packaging could be used but this provides a disconnect with the actual packaged product that the consumer purchases.

This list is by no means complete, however, it demonstrates that there are many packaging attributes that influence consumer behaviour and food waste. In the "move towards sustainable food packaging, the relational complexity between the role of packaging and reduced food waste needs to be included beyond just extending shelf life to consider user behaviour" (Verghese et al., 2012, p 402). The following section introduces case studies of food waste scenarios quantified though streamlined LCA.

3. Case studies of food waste scenarios quantified though streamlined LCA

The environmental impact of packaging can be assessed in a streamlined fashion by adding the life cycle impacts of the wasted food to the lifecycle impact of the packaging as outlined in the following formula:

Impact (i) = packaging (p) + food wasted (fw)

The functional unit shifts to become 'the delivery of consumed food per kg', attributing the environmental impact of the unconsumed food to the total product-packaging system. To complete the formula requires data of three interrelated elements.:

(p) Input and outputs of process related to packaging type
(f) Input and outputs of processes related to food production from farm to fork
(w) The percentage of food wasted per packaging type (w =% wasted)

Life cycle data is readily available on packaging and food in a streamlined form. It is well known that different food types have differing environmental impacts and characteristics. For example, cattle require more feed per edible weight than poultry and emit methane from the foregut with a higher global warming potential (GWP) than mono gastric livestock (pigs and chickens) that in turn have a higher GWP than vegetables (Eady, et al., 2011, p.1). Peer reviewed LCA and Environmental Product Disclosure (EPD) data is widely available in the public domain on the majority of food types. For example the authors identified CO_2-eq data for 200+ foods from 62 studies, Audsley et al., (2009) study provide environmental impact data on 100+ foods in the United Kingdom, and the International EPDsystem (2012) lists 44 EPD's from a range of brands. With respect to packaging streamlined tools, the Packaging Impact Quick Evaluation Tool (PIQET) enables the environmental impacts of packaging types to be quickly evaluated and re-run with changed packaging system specifications (Verghese et al., 2010).

The part of the equation that lacks sound data is the percentage of food wasted per packaging type. Sensitivity analysis can be constructed modeling various ‘scenarios of use’ to measure the viability of increased packaging and possible reduced food waste. Two case studies are presented below based on recent lived experience of the authors. While acknowledging that this is not a valid sample, it is viewed to be indicative of the impact that food waste may have.

3.1. Case Study 1: rice packaging

Baker et al., estimated that Australians waste over $550 million in rice and pasta each year (2009, p. 8), with the Australian state of New South Wales (NSW) *Love Food Hate Waste* surveys indicating that 1/3 of recipients found it hard to estimate how much to cook per person (DECCW 2009, p.2). Rice is a dish that doubles in size when cooked, therefore making judgements on how much to cook difficult. Two packaging scenarios for rice are presented: i) uses a pre-cooked 250 g rice packaging in a plastic laminate flexible pouch and ii) a bulk purchase 1 kg bag (Table 2).

Table 2. Two types of rice packaging compared

Packaging (p)	= 8 g LDPE pouch[a]	= 10 g LDPE packet[a]
	= 8 x 2.92 g $CO_{2\text{-eq}}$/g	= 10 x 2.92 g $CO_{2\text{-eq}}$/g
	= 23.4 g $CO_{2\text{-eq}}$	= 29.2 g $CO_{2\text{-eq}}$
Food impact (f)	= 250 g rice[b]	= 1000 g rice[#]
	= 250 x 6.4 g $CO_{2\text{-eq}}$/g	= 1 000 x 6.4 g $CO_{2\text{-eq}}$/g
	= 1 600 g $CO_{2\text{-eq}}$	= 6 400 g $CO_{2\text{-eq}}$
Waste (w)[c]	= 2% Residual rice left in pack	= 20% rice (cooked to much)
Impact/pack	= p + fw	= p + fw
	= 23.4 + (1 600 x 2%)	= 29.2 + (6 400 x 20%)
	= 55.4 g $CO_{2\text{-eq}}$ /250 g pack	= 1 309.2 g $CO_{2\text{-eq}}$/1 kg pack
Impact/kg[d]	**=221.6 g $CO_{2\text{-eq}}$/kg**	**= 1 309.2 g $CO_{2\text{-eq}}$/kg**

a. impact factors from Ecoinvent database (2012) using the Australian impact method
b. Carlsson-Kanyama (1998)
c. emissions from the biodegradation of the wasted food were not considered in this model, if included the emission factor for food waste would be higher
d. functional unit of packaging is the ‘delivery of consumed food per kg’

The inclusion of food waste substantially changes the outcomes of the environmental impact. The pre-cooked rice appears preferable with an environmental impact 6 times less than the bulk packet once food waste is considered. If waste is not considered the pre-cooked packet has a higher environmental impact. As the pre-cooked packet contains the correct quantity it is viewed to be easy to dose. This should not rubber stamp smaller portions as always being preferred, instead the results indicate challenges for packaging designers to redesign rice packaging to make it easy to dose and by eliminating the potential for over portioning. This could be achieved by a number of design innovations in either visual communication and or packaging design so that rice is dosed in appropriate amounts including clearly illustrating the relationship between uncooked rice and cooked rice.

3.2. Case Study 2: yogurt packaging

Youghurt once opened has a limited shelf life within the refrigerator. The two packaging types: i) 6 pack 175 g connected tubs of yogurts are purchased in comparison to ii) one large 900 g polypropelene tub (Table

2). The authors' experience is that once opened, the large yogurt needs be consumed in a timeframe that is not always met, whereas individual packets are consumed in one serving.

Table 2. Two types of yogurt packaging compared

Packaging (p)	= 6 x 7 g PS [a] tubs	= 35 g PP [a] tub & lid + 2 g aluminium[a] foil
	= 42 x 3.97 g $CO_{2\text{-eq}}$/g	= 35 x 2.90 g $CO_{2\text{-eq}}$/g + 2 x 12.57 g $CO_{2\text{-eq}}$/g
	= 166.7 g $CO_{2\text{-eq}}$	= 126.7 g $CO_{2\text{-eq}}$
Food impact (f)	= 6 x 175 g yogurt[b]	= 900 g yogurt [b]
	= 1 050 x 1.22 g $CO_{2\text{-eq}}$/g	= 900 x 1.22 g $CO_{2\text{-eq}}$/g
	= 1 281 g $CO_{2\text{-eq}}$	= 1 098 g $CO_{2\text{-eq}}$
Waste (w) [c]	= 5% residual yogurt in pack	= 30% yogurt (mouldy in fridge)
Impact/pack	= p + fw	= p + fw
	= 166.7 + (1 281 x 5%)	= 126.7 + (1 098 x 30%)
	= 230.8 g $CO_{2\text{-eq}}$ /6 pack	= 456.1 g $CO_{2\text{-eq}}$/900 g tub
Impact/kg[d]	**= 219.8 g $CO_{2\text{-eq}}$/kg**	**= 506.7 g $CO_{2\text{-eq}}$/kg**

a. impact factors from Ecoinvent database (2012) using the Australian impact method
b. Lindenthal et al., (2010)
c. emissions from the biodegradation of the wasted food were not considered in this model, if included the emission factor for food waste would be higher
d. functional unit of packaging is the 'delivery of consumed food per kg'

The results indicate that once food waste is included the environmental impact differ substantially. In the yogurt example, the bulk 900g packet has a 24% lower GWP than the 6 pack when viewed in isolation of food waste, and a 230% higher GWP when included in the above scenario. Appropriate portion sizing that reduce food waste has a dominant impact, even with a food type such as yogurt that has a relatively low GWP in comparison to meat based products.

From a design perspective it is also possible to foresee that packaging design to reduce food waste should not always result in an increase in the environmental impact of packaging in isolation. Traditional packaging design solutions such as lightwieghting and material selection could apply to make the 6 pack less material intense than the 900 g tub.

4. Discussion: Possible means to integrate food waste into LCA

The results of the rice and yogurt case studies indicate scenarios where increases in packaging may reduce the overall environmental impact by avoiding food waste. The most critical data gap to successfully completing replicable LCA's is in estimating the amount of food wasted. A very small variation in the percentage of food wasted that has a high GWP potential like red meat substantially differs the environmental impact of packaging once food waste is included. It is also acknowledge that packaging is only one of many ways that food waste could be reduced. In a recent stakeholder engagement forum in Australia brand owners identified that they rarely complete user trials on how packaging is actually used in the home (Verghese et al., 2012b). To test the success of alternate packaging scenarios requires 'additional fieldwork and empirical research outside the traditional boundaries of LCA' (Verghese, et al., 2012a, p. 403). Understanding food waste in the home is a fundamental first step to improving the resolution of waste estimates. Multiple sensitivities can quickly be run by altering the percentage wasted and packaging type to see the impact of reduced wastage and they can be used to develop hypothesis that require empirical testing.

The packaging attributes that influence food waste elaborated in section 2 provide a useful guide for packaging designers to assist them in designing packaging to faciliate reduced food waste. By taking a service perspective the focus can move from the product itself, to the process it is used for (Vargo and Lush, 2004; Edvardsson *et al*., 2005). The product can be described by attributes, each attribute providing prerequisites for the service. Each packaging attribute assists to script individual behaviour and experiences, and potentially the environmental outcome e.g. , the amount of food waste generated. The consumer interaction with the product depends on the design of the product, the consumer preferences and experiences, and the context of the consumer (Löfgren, 2006).

For example, consider the attribute contain the desired quantity. If the offered quantity of fresh bread does not agree with the desired, the service of eating fresh bread may not be used for the entire piece of bread, some may be frozen, some may be eaten “old”, and some may be wasted, depending on consumer preferences and behaviour. Therefore, the size of the bread can affect the consumer behaviour and the amount of food that is discharged, and thus the environmental outcome. In Australia Bakers Delight introduced small block loaves of bread provide a simple alternative to the full loaf of bread – potentially reducing associated food waste for single person or small households (Verghese et al., 2011). By understanding the consumer preferences and behaviour the services that are provided from a specific packaging attribute can be designed to better meet the consumer needs and facilitate consumers to waste less food.

5. Conclusion

There are many packaging attributes that influence consumer behaviour and the portion of food that is wasted. A small change in the percentage of food wasted significantly affects the outcome of an LCA of food products. Due to the high environmental impact of the food sector, it is important to include these aspects in the LCA of food products, especially when the results are used for decision-making about packaging. There is certainly a long journey ahead until this can be done in a standardized way. It will be necessary for the food packaging supply chain including food producers, manufacturers, brandowners and retailers to include both qualitative and quantitative information about how different packaging attributes influence consumer behaviour and food waste. It will also be important to consider and include consumer behaviour in LCA-studies of food products to take into account the environmetnal impacts of the 30% of bought food that is ultimately wasted. If a LCA study is used for packaging development, or for packaging regulations, the packaging function to avoid food waste certainly should be included to inform better decisions. In such studies, the proper functional unit should be set to “consumed food” rather than “delivered food” or “bought food”. The assumptions of how different packaging attributes affect food waste should also be clearly stated and discussed.

6. References

APCC. 2010. Australian Packaging Covenant – A commitment by governments and industry to the sustainable design, use and recovery of packaging. Australian Packaging Covenant Council, Amended 10 October 2011.

Audsley, E., Brander, M., Chatterton, J., Murphy-Bokern, D., Webster, C., and Williams, A. 2009. “How low can we go? An assessment of greenhouse gas emissions from the UK food system and the scope to reduce them by 2050”. FCRN-WWF-UK.

Baker, D., J. Fear and R. Denniss. 2009. What a waste: An analysis of household expenditure on food. Policy Brief No. 6. Canberra, The Australia Institute.

Carlsson-Kanyama, A. 1998. "Climate change and dietary choices—how can emissions of green-house gases from food consumption be reduced?" Food Policy 23(3/4): 277-193.)

DECCW. 2009. "Food Waste Avoidance Benchmark Study 2009." Retrieved 12/10/2010, 2010, from http://www.lovefoodhatewaste.nsw.gov.au/portals/0/kit/print/10242FWAFactsheetCMYK_print.pdf.

Duizer, L.M., Robertson, T., Han, J., 2009. Requirements for packaging from an ageing consumer's perspective. Packaging Technology and Science. Vol 22, 4, 187-197

Eady, S. J., P. Sanguansri, R. Bektash, B. Ridoutt, L. Simons and P. Swiergon. 2011. Carbon foot-print for Australian agricultural products and downstream food products in the supermarket. ALCAS Conference. Melbourne, ALCAS.

Edvardsson, B., Gustafsson, A., Roos, I., 2005. Service Portraits in Service Research – A Critical Review, International Journal of Service Industry Management. Vol. 16, No. 1, pp. 107-121.

Engström, R., Carlsson-Kanyama, A., 2004. Food losses in food service institutions: Examples from Sweden. Food Policy. Vol. 29, No 3, pp. 203-294.

European Commission. COM(2006) 767 final. 2006. Report from the Commission to the Council and the European Parliament on the implementation of Directive 94/62/EC on packaging and packaging waste and its impact on the environment, as well as the functioning of the internal market.

Ecoinvent .2012. Ecoinvent Database v2.2. Swiss Centre for Life Cycle Inventories. Hamburg.

European Council. 1994. Directive 94/62/EC on packaging and packaging waste. Official Journal of the European Union. 31/12/1994; L365:0010 - 0023.

Hammond, G. and C. Jones. 2011. Inventory of Carbon & Energy (ICE). University of Bath. Department of Mechanical Engineering. Bath. ICE V2.0.

Hanssen, O.J., 1998. Environmental impacts of product systems in a life cycle perspective: a survey of five product types based on life cycle assessments studies. Journal of Cleaner Production. Vol. 6, pp. 299-311.

Jelsma, J. 2006. Designing ‘moralized’ products. User Behavior and Technology Development. P.-P. Verbeek and A. Slob. Netherlands, Springer 221-231.

Lindenthal, T., T. Markut, S. Hörtenhuber and G. Rudolph. 2010. Greenhouse Gas Emissions of Organic and Conventional Foodstuffs in Austria.

Löfgren, M., 2006. The leader of the pack. Faculty of Economic Sciences, Communication and IT. Karlstad, Karlstad University. ISBN: 91-7063-089-5.

Löfgren, M., Witell, L., 2005. Kano's Theory of Attractive Quality and Packaging. Quality Management Journal. Vol. 12, pp. 7-20.

Quested, T., Johnson, H., 2009. Household food and drink waste in the UK. wrap. Banbury UK. 2009. ISBN: 1-84405-430-6.

Roy, P., Nei, D., Orikasa, T., Xu Q. and Okadome, H. 2009. A review of life cycle assessment (LCA) on some food products. Journal of Food Engineering 90, 1-10.

Vargo, S.L, Lusch, R.F. 2004. Evolving to a New Dominant Logic for Marketing, Journal of Marketing, Vol. 68, No. 1, pp. 1-17.

Ventour, L. 2008. The food we waste. WRAP. Banbury UK. ISBN: 1-84405-383-0.

Verghese, K., R. Horne and A. Carre. 2010. "PIQET: the design and development of an online 'streamlined' LCA tool for sustainable packaging design decision support." International Journal of Life Cycle Assessment 15(6): 608-620.

Verghese, K., H. Lewis, S. Clune, S. Lockrey and P. Crittenden 2011. AFGC future of packaging background paper. Melbourne, RMIT University, Helen Lewis Research, and Sustainable Business.

Verghese, K., S. Lockrey, S. Clune and D. Sivaraman. 2012a. Life cycle assessment of food and beverage packaging. Innovative and sustainable food and beverage packaging. K. L. Yam. Cambridge, Woodhead.

Verghese, K., Clune, S., Lockrey, S., Lewis, H and Crittenden, P. 2012b. Future of Packaging White Paper; Executive Summary (April), Australian Food and Grocery Council (AFGC), Canberra, Australia

Williams, H., Wikström, F., 2011. Environmental impact of packaging and food losses in a life cycle perspective: a comparative analysis of five food items. Journal of Cleaner Production. 19, pp. 43-48.

Williams, H., Wikström, F., Löfgren, M., 2008. A life-cycle perspective on environmental effects of customer focused packaging development. Journal of Cleaner Production 16, pp. 853-859.

Williams, H., Wikström, F., Otterbring, T., Löfgren, M., Gustafsson, A., 2012. Reasons for household food waste with special attention to packaging. Journal of Cleaner Production 24 pp.141-148

WRAP. 2007. "Minimisation of Household Food Waste." Retrieved 15/12/2011, 2011, from http://www.wrap.org.uk/downloads/WRAP_Food_Waste_Minimisation_adp_-_27_10_071.7aaf9619.4672.pdf.

WRAP. 2009. Household Food and Drink Waste in the UK. Final Report. H. J. Tom Quested. Oxon, Waste & Resources Action Programme.

Comparative LCA of fruit and vegetable packaging trays

Jean-François Ménard[1,*], Claude Belley[1], Réjean Samson[1], Dominique Maxime[1], Mathieu Roberge[2]

[1] CIRAIG, École Polytechnique de Montréal (QC, Canada)
[2] Cascades Speciality Products Group, Consumer Product Packaging – CSP, CPP (QC, Canada)
[*] Corresponding author. E-mail: *jean-francois.menard@polymtl.ca*

ABSTRACT

Seven types of fruits and vegetables retail packaging trays currently used in Quebec are compared cradle-to-grave: polystyrene foam (XPS, 100% virgin), oriented polystyrene (OPS, 90% virgin), polyethylene terephthalate (PET, 90% virgin, and RPET, 100% recycled), polylactide polymer (PLA, 90% virgin), polypropylene (PP, 90% virgin), and molded pulp (MP, 100% recycled). Main contributors to potential impacts throughout the entire life cycles of the trays are linked to production processes (raw materials and manufacturing energy production). PLA, PET and PP post the worst performances while XPS and MP have the best overall performances. The main asset of the XPS tray lies in its significantly lower weight despite the virgin material and the impact from tray production. Seven sensitivity analyses are performed on key parameters, confirming the robustness of initial findings. Plastic systems are very sensitive to the electricity grid mix used at the production step.

Keywords: packaging, fruits and vegetables, plastics, PLA, moulded pulp, life cycle impact assessment

1. Introduction

The Consumer Product Packaging sector is a member of the Cascades Specialty Products Group that designs, manufactures and markets packaging products made of moulded pulp and plastic, some of which are 100% recyclable. It covers the food, hardware and consumer product industries. Its diversified portfolio also includes site furnishings and construction materials made of 100% recycled plastic. In an effort to reduce its potential environmental impacts, CSPG, CPP is currently carrying out internal evaluations of the environmental performances of its products. The company seeks to compare its own products and take a proactive stance given the increased number of green procurement policies of business and institutional buyers. Single-use packaging trays are widely used in Quebec (Canada) for the retail sale of fruits and vegetables.Earlier studies comparing bio-based and hydrocarbon polymer resins for food packaging generally showed no clear environmental preference but rather trade-offs when all the environmental indicators are considered (Athena Institute, 2006 ; Detzel and Kruger, 2006). In addition, procurement logistic and the transport of the packaging containers' are key parameters within the North american context (Madival et al., 2009). This study aims to carry out a comparative life cycle assessment of the various materials currently used to manufacture the trays, including moulded pulp trays. The results will enable CSPG, CPP to enhance its understanding of the potential environmental impacts associated with the life cycle of their food trays sold in Quebec and provide answers that will help the company reduce its impacts and position itself to meet market-driven sustainability requirements.

2. Methods

2.1. Goal and scope, and functional unit

The goal of this study is to establish and compare the environmental profiles of the complete life cycle of various single-use fruit and vegetable packaging trays sold in Quebec (Canada) by CSP, CPP. The compared options are actual trays that are, or were in the recent past, manufactured by CSPG, CPP and available on the market; the extruded polystyrene foam currently being the most used one. The 10% recycled content for some of the trays represents post-industrial recycling (CSPG, CPP estimate).

The functional unit is to "*Contain and permit the stacking and retailing of an amount of fruits or vegetables that can be contained in a tray volume of 52 cubic inches to consumers in Québec in 2010*." The volume refers to a tray that measures 8.38 inches in length, 5.88 inches in width and 1.06 inches in height.

The reference flows are the amounts of product necessary to deliver the functional unit – one tray of each type. The characteristics of each tray are presented in Table 1.

Table 1. Studied food tray options

System	Tray	Weight (g)
XPS	100% virgin extruded polystyrene foam	10.45
OPS	90% virgin-10% recycled oriented polystyrene	20.85
PET	90% virgin-10% recycled polyethylene terephthalate	27.15
RPET	100% recycled PET	27.15
PLA	90% virgin-10% recycled polylactide polymer	25.20
PP	90% virgin-10% recycled polypropylene	19.80
MP	100% recycled moulded pulp	20.00

2.2. System boundaries and assumptions

The study includes all of the flows and processes involved in the production, distribution and end-of-life stages of the trays, including the production and transport of the resources consumed and the management of the waste generated at each stage (Figure 1).

Preproduction generally includes plastic (granulates) and old newspapers procurement. Depending on the type of raw material, the following processes are included:

- Virgin plastic: Oil extraction, refinement and transport, polymerisation and granulation.
- Recycled plastic: Transport of the used plastics and production of the recycled plastic granules (electricity consumption only).
- PLA: Corn production, starch extraction, dextrose and lactic acid production, polymerisation and granulation.
- Recycled newspapers: Transport of the old newspapers and production of the recycled pulp (electricity consumption at CSP, CPP plant only).

The production sub-system consists of the various stages that occur at the tray manufacturing site (CSPG, CPP plant in Québec). It therefore includes the operation of the manufacturing infrastructure for the extrusion and thermoforming (only modeled as an electricity consumption) of the plastic trays or the moulding and drying of the MP trays (only modeled as a consumption of electricity and heat from natural gas), and the finished product packaging (LDPE bags for the XPS tray and the same corrugated board boxes for the other plastic trays and a slightly heavier corrugated board box for the MP tray).

The distribution sub-system includes the transport of the trays from the production sites to the retailers.

Finally, the use sub-system includes the packaging of the fruits and vegetables in the tray (no impacts associated with this activity) and their transport to the consumer's house. The end-of-life sub-system includes the end-of-life management of the trays and the various packaging items used in the life cycle and the transports between the consumer and waste management sites.

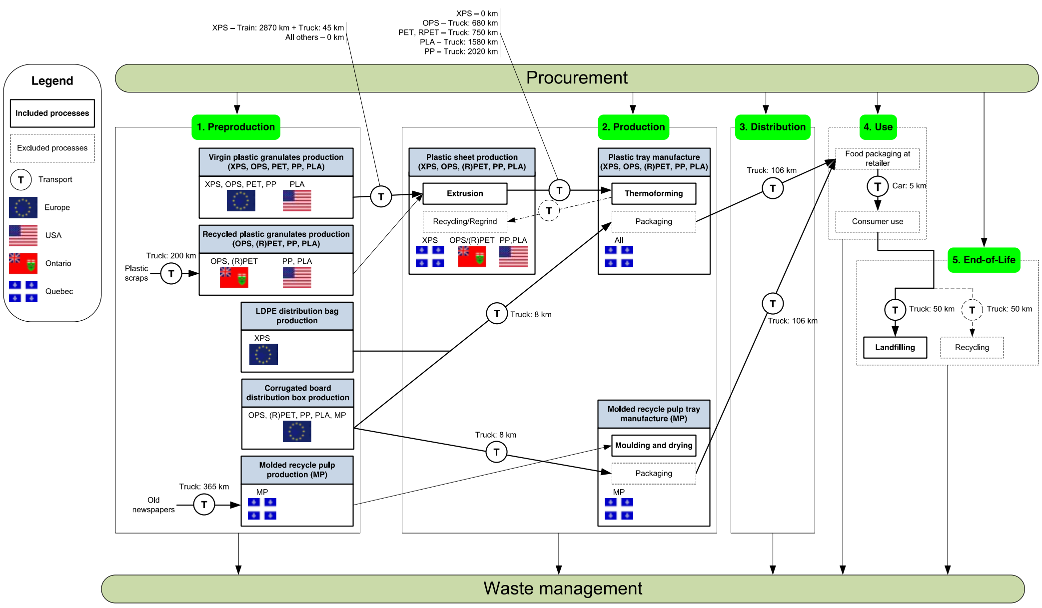


Figure 1. Boundaries of the product system. (Note: the European flag only means that the processes were modeled with European data from the ecoinvent database, whereas they are most probably produced in North America)

The following elements were excluded from the system boundaries: the transport of the plastic granulates from the production sites to the plastic sheet extrusion sites for the plastic trays other than the XPS one; the warehousing and handling of the trays by retailers (and wholesalers or other intermediaries, if applicable); the production and transport of the plastic film used to package the fruits and vegetables; human activities (e.g. employees' daily travel); the life cycles of the capital goods (e.g., roads, production equipment), with the exception of those already included in the ecoinvent data modules used in the LCA model. In addition, the materials recovered by the studied systems (plastic scraps and old newspapers) are considered to be resources consumed by these systems (and not as waste generated by other systems). The recovery (i.e. collection and recycling) processes are therefore entirely attributed to the use of these materials and the studied systems. However, in the same fashion, the recovery of the trays at the end of their service lives is therefore not attributed to the studied systems.

General assumptions pertaining to the studied systems are:

- The OPS and XPS granulates production processes were considered identical;
- The tray manufacturing (only the electricity consumption for the sheet extrusion and thermoforming was considered) was assumed to be the identical for all plastic trays, per kilogram of tray, based on the primary data obtained for the XPS tray from CSPG, CPP;
- The truck transport during the distribution of the trays to the retailers was modeled as volume limited (or volume based) since the trailer of the truck is full before its weight limit is reached because of the bulk of the packaged trays;
- The average transport distance of the trays to the retailers was assumed to be 106 km;
- No tray loss was assumed during the distribution transport between the manufacturing site and the retailers;
- The average transport distance of the trays to the consumer's house was assumed to be 5 km;
- The average transport distance of the trays at the end of their service lives to the waste management (i.e. landfill) sites was assumed to be 50 km;
- The plastics average recycling rates were assumed to be 0% for the XPS, 15% for OPS, 38% for the PET and RPET, 17% for the PP and 0% for the PLA, based on the latest residential solid waste characterisation published by Recyc-Quebec (Recyc-Quebec, 2010);

All studied trays are considered equivalent in regard to the above-mentioned function even if their individual technical characteristics (e.g. rigidity, water-resistance) are not the same. No actual performance measurements were made but the fact that each tray is actually used in stores for the considered function supports this equivalency assumption.

2.3. Life cycle inventory (LCI) data, and Life cycle impact assessment (LCIA)

This study mainly relied on available primary data collected from CSPG, CPP and it suppliers by way of questionnaires and direct communications, they represent annual averages. Any missing, incomplete or inaccessible data was completed with hypotheses and secondary data (i.e., generic or theoretical data available in the literature or LCI databases, essentially from the European ecoinvent database (www.ecoinvent.ch), version 2.0). For all of the activities taking place in Québec, Ontario or the United States, the generic modules were adapted by replacing the European electricity grid mixes with the average Québec, Ontarian or North American grid mixes for the foreground processes, and with the North American grid mix for the background processes.

LCIA was carried out using the IMPACT2002+ method (Jolliet et al., 2003) (version 2.05). The inventory calculation and the assessment of the potential impacts of the inventoried emissions were carried out using the SimaPro software developed by PRé Consultants (www.pre.nl).

3. Results

Table 2 illustrates the relative contributions of the various foreground processes as they pertain to the total obtained for each system, for the four damage indicator results and the two impact indicator results not aggregated into one of the damage category (aquatic eutrophication and acidification). Globally, it shows how the virgin material production (plastic granulates or PLA), the production of recycled pulp, and the forming (extrusion, thermoforming, or moulding) dominate damages and impacts. The distribution box production for the MP tray also contributes significantly, and to a lesser degree the tray distribution packaging.

Table 2. Relative contributions (in%) of the foreground steps to the total for each tray system (shaded cells in red are contributions that are equal to or greater than 15%) (Note: recycled pulp production and tray moulding are aggregated together in the results for MP).

	Human health							Ecosystems quality							Climate change						
	XPS	OPS	PET	RPET	PLA	PP	MP	XPS	OPS	PET	RPET	PLA	PP	MP	XPS	OPS	PET	RPET	PLA	PP	MP
Virgin Material	68	44	74	0	42	25	0	32	11	38	0	65	2	0	81	47	44	0	14	22	0
Recycled Material	0	0	0	8	0	1	0	0	1	0	6	0	1	0	0	0	0	8	1	1	0
Transport Material	13	3	1	5	2	4	5	16	7	5	8	3	9	5	6	1	2	3	3	3	1
Forming	13	44	21	76	52	66	44	45	49	37	57	27	72	22	9	44	48	80	77	69	78
Packaging	1	4	2	6	1	2	34	1	23	13	20	3	12	63	1	5	4	6	3	4	18
Distribution	1	2	1	2	1	1	6	4	7	4	7	1	4	7	1	1	1	1	1	1	1
Transport Consumer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
End-of-life	3	3	1	4	1	1	11	3	3	1	2	1	1	3	3	2	1	2	1	1	2

	Resources							Aquatic acidification							Aquatic eutrophication						
	XPS	OPS	PET	RPET	PLA	PP	MP	XPS	OPS	PET	RPET	PLA	PP	MP	XPS	OPS	PET	RPET	PLA	PP	MP
Virgin Material	88	53	50	0	29	42	0	77	50	47	0	1	13	0	70	19	67	0	56	57	0
Recycled Material	0	0	0	8	1	0	0	0	0	0	8	1	1	0	0	0	0	5	0	0	0
Transport Material	3	1	1	2	2	3	1	12	3	3	5	3	3	5	16	8	4	12	5	7	3
Forming	6	42	45	84	64	52	81	6	39	43	76	91	80	45	5	15	8	22	17	19	7
Packaging	1	3	2	4	3	3	15	1	4	3	6	2	2	32	0	30	11	33	7	9	48
Distribution	0	1	0	1	1	1	1	1	1	1	2	1	1	5	2	6	2	6	1	2	4
Transport Consumer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
End-of-life	1	1	1	1	1	1	2	3	3	2	3	2	1	12	7	23	7	21	14	6	38

Comparing the environmental profiles of the systems studied provides answers as to the potential environmental gains associated with the use of different materials (virgin or recycled) to manufacture the trays, without, however, confirming whether one or another posts better performances in all damage/impact categories (Figure 2). The Figure 2 allowed deriving a system hierarchy, where the green grouping represents the systems with the least potential impacts, the red one those systems with the most potential impacts and the grey one represents the intermediate systems.

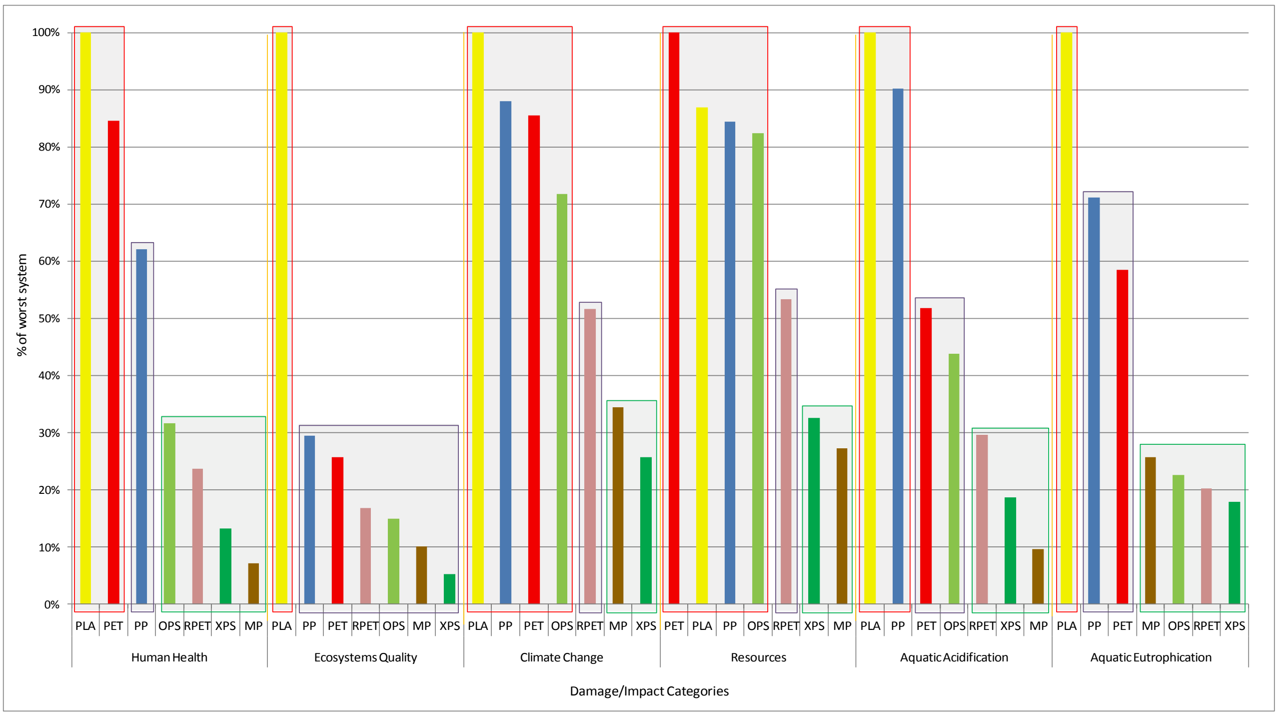


Figure 2. Comparative cradle-to-grave damage/impact of the packaging trays. Green boxed grouping: systems with the least potential impacts, red grouping: systems with the most potential impacts, grey groupings: intermediate systems.

4. Discussion

Given this hierarchy, the PLA tray holds the last place since it is, for all categories, among the systems with the most potential impacts. The high contribution to ecosystem quality of PLA material production is explained by the use of large areas of arable land for the corn cultivation generating a very high land occupation score. The PET, OPS and PP trays follow since they are for several categories among the systems with the most potential impacts (however, the OPS tray seems to be better since it is among the systems with the least potential impacts for two categories). Then comes the RPET tray since it shows the least potential impacts for two categories and shows intermediate results for all the other. Finally, the XPS and MP trays come in first and second place since they show the least potential impacts for four out of the six categories. The main advantage of the XPS tray lies in its mass, which is far less than that of other trays (up to 62% less). On the contrary, the PLA and PET trays are the heaviest of the trays and the virgin material production processes are important contributors. The main advantage of the MP tray lies in its recycled nature since no virgin material production is necessary and its manufacturing is relatively low energy intensive (58% higher than the XPS tray but 27% lower on average than the other plastic trays). The system is however penalised by a heavier distribution packaging.

Seven sensitivity analyses were carried out to assess how robust the results are with regards to the uncertainty behind main assumptions, generic data modules and various methodological choices.

1. *Tray weight*: it is an important parameter since it affects the amount of material used and the amount of energy used during the tray manufacturing and its distribution (the marginal amount of energy per unit of mass is still considered in the volume-based transport). A 20% higher and lower weight was considered in the analysis. The results show a high degree of overlapping for the plastic trays other than the XPS (i.e. lower in the comparative hierarchy). Yet, the XPS and MP trays show very little overlapping except with each other and thus maintain their better ranking overall.
2. *PET recycling*: In the baseline scenario, a generic process was used to model the recycled PET. The use of a recycled PET-specific process makes it possible to assess the degree to which the relative contribution of the RPET trays can change as compared to the other trays. As compared to the first approximation (simple electricity consumption), the specific process yields practically identical results and the conclusions therefore remain the same.
3. *Allocation approach*: In the baseline scenario, the allocation approach attributed 100% of the impacts of recycling to the user of the recycled material (the system that generates the waste to be recycled is attributed 0% of the impacts). A sensitivity analysis was carried out to include the end-of-life recycling process within the system boundaries through system expansion so as to account for the affected processes, e.g. the use of recycled plastic in the manufacturing of the tray avoids the landfilling of plastic wastes, thus the landfilling process is credited (i.e. has a negative contribution to the total score) to the system and is included in the Recycled Material life cycle step; and the recycling of the tray avoids the production of virgin plastic in other products systems, thus the virgin production process is credited to the system and is included in the End-of-Life life cycle step. The conclusions remain the same. In fact, the MP system shows an overall negative result for the ecosystem quality indicator associated with the non-deinked old newspaper used to manufacture the tray (the deinking process is avoided since the old newspaper would have been deinked had it been recycled into new newspaper).
4. *The North American electricity grid mix*: This analysis used the North American electricity grid for the recycled granulates/pulp production and tray manufacturing (extrusion and thermoforming). The XPS tray remains the favourite plastic option and the MP tray becomes the clear favourite for all indicators (it is practically equal to the XPS tray for the ecosystem quality and aquatic eutrophication categories) in light of its low electricity consumption (MP consumes natural gas, which, per MJ, generates less potential impacts than the North American energy grid).
5. *The Québec electricity grid mix*: This analysis used the Québec electricity grid for the recycled granulates/pulp production and tray manufacturing. Because the RPET system uses no virgin material and only consumes electricity during granulates production and tray manufacturing, it comes out as the favoured option (even if it shows a little higher results than the XPS tray for the ecosystems quality category because of the corrugated board distribution box and granulates transport which have not changed). The XPS tray retains its advantage over the other plastic trays because of its lower weight. The systems that profit the most from this grid mix change are those that relied on the U.S. grid (PLA and PP trays for the recycled granulates production and the sheet extrusion), which is far more impactful as compared to the Québec grid mix than is the Ontarian mix (used by the OPS, PET and RPET trays); the PLA tray obtains an even lower result for the aquatic acidification cate-

gory than the XPS tray. The MP tray shows good results compared to the plastic trays, but is still penalised for the climate change and resources depletion categories by the natural gas used to dry it once it is moulded.

6. *The LCIA method*: The European method ReCiPe (version 1.03, Hierarchist vision) was used to verify whether the variability of the receiving environments and characterisation models changed the conclusions. It therefore aimed to test the robustness of the IMPACT2002+ results. This second method confirmed the advantage of the XPS and MP for the human health and ecosystem quality indicators (the climate change category contributes to these two indicators). In the case of resource consumption, the clear preference for the RPET is based on the uranium characterisation factor, which is much lower for ReCiPe (nuclear energy being the main contributor for the RPET in the IMPACT2002+, its contribution is significantly curbed).
7. *The tray distribution to retailers (distance)*: This distance was increased from 106 km for all of the scenarios to 1500 km for the sensitivity test. Clearly, for the ecosystem quality indicator, the relative contribution of the transport increases without changing the comparative analysis of the trays. The low weight of the XPS trays puts the option ahead of MP.

The sensitivity analyses do not change the preliminary conclusions, and XPS and MP remain the systems with the least potential impacts.

5. Conclusion

The results obtained indicate that the main contributors to damages and impacts categories throughout the entire life cycles of the trays are linked to production processes (raw materials and manufacturing energy production). From a comparative perspective, the results provide answers regarding the potential environmental gains associated with the use of different materials (virgin or recycled) for tray production, without affirming the advantage of one or another in every damage category. Specifically, PLA, PET and PP post the worst performances while XPS and MP have the best overall performances. The main advantage of the XPS tray lies in its mass, which is far less than that of other trays (up to 62% less). To enhance the performance of the other trays, it would be important to reduce their respective weights and optimize the subsequent transformation processes (extrusion and thermoforming or moulding).

The sensitivity analyses show that the plastic systems are very sensitive to the electricity grid mix used to form the tray, highlighting the environmental advantage of manufacturing the trays in Quebec Province where hydroelectricity is 95% of the grid mix. Besides reversing the popular belief over the negative attributes of polystyrene packaging (in the Quebec context where almost no recycling channel for PS exist), this critically reviewed LCA study has supported strategic decision processes within the company. Since this analysis was performed, Cascades has improved its manufacturing process by integrating upstream with an RPET extrusion and thermoforming line in its plants in Quebec province and is working continuously to enhance the recycled content of RPET material (currently 60%). Preliminary results indicate that the environmental impacts of RPET with a high degree of recycled content could be compared to those of XPS foam and moulded pulp.

6. References

Athena Institute, 2006. Life cycle inventory of five products produced from polylactide (PLA) and petroleum-based resins. Available from: www.athenasmi.ca/projects/docs/Plastic_Products_LCA_Summary_Rpt.pdf

Detzel A., Kruger M., 2006. Life cycle assessment of polylactide. A comparison of food packaging made from NatureWorks PLA and alternative materials. IFEU Institute (Institute for Energy and Environmental Research) Report No 168.

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G and Rosenbaum R., 2003. IMPACT 2002+: A New Life Cycle Impact Assessment Methodology. Int. Journal of LCA, 8(6), 324-330.

Madival S, Auras R, Singh SP, Narayan R, 2009. Assessment of the environmental profile of PLA, PET and PS clamshell containers using LCA methodology. J. Cleaner Production, 17(13), 1183-1194.

Recyc-Quebec, 2010. Caractérisation des matières résiduelles du secteur résidentiel au Québec 2006-2009, Document complémentaire, http://www.recyc-quebec.gouv.qc.ca/Upload/ Publications/doc_comp.pdf

A systems-LCA model of the stratified UK sheep industry

Adrian Williams*, Eric Audsley, Daniel Sandars

Centre for Environmental Risks and Futures, School of Applied Sciences, Cranfield University, Bedford, MK43 0AL, UK.
* Corresponding author. E-mail: *adrian.williams@cranfield.ac.uk*

ABSTRACT

A systems model of the stratified UK sheep industry was developed to provide the activity data inputs for life cycle assessment (LCA) in the Cranfield systems-LCA model. It includes the biophysical performance of the lowland, upland and hill sheep flocks, which are economically interlinked and provide meat and wool as outputs. The sheep metabolisable energy requirements are calculated from standard formulae and the grazing needs are determined from the feed supplied by grass coupled with a bespoke grass production model. This calculates the N requirements and productivity over a wide range of climatic and soil zones.

Baseline results were compared with alternative scenarios. The potential for improvement under reasonable changes in management and technology could reduce main burdens by about 14% and about 24% with more demanding changes. Changes in the industry structure have major effects on land occupation, but smaller effects on energy use and greenhouse gas emissions.

Keywords: lamb, systems model, greenhouse gases, enteric methane, wool

1. Introduction

The UK sheep industry can be divided into hill, upland and lowland flocks. The overwhelming purpose is to produce meat (of which lamb is economically dominant), with wool as a co-product. The milking flock is negligible and not addressed in this paper. There are many breeds ranging from the smaller breeds most suited to the severe conditions of hill farming, which struggle to produce a lamb each year, to the larger breeds associated with lowland farming. These include very local traditional breeds (e.g. Herdwick) to more recently-imported breeds, such as the Texel. The system that has developed combines the qualities of these different breeds to develop hybrid vigour and make best use of the available land at different altitudes (Figure 1).

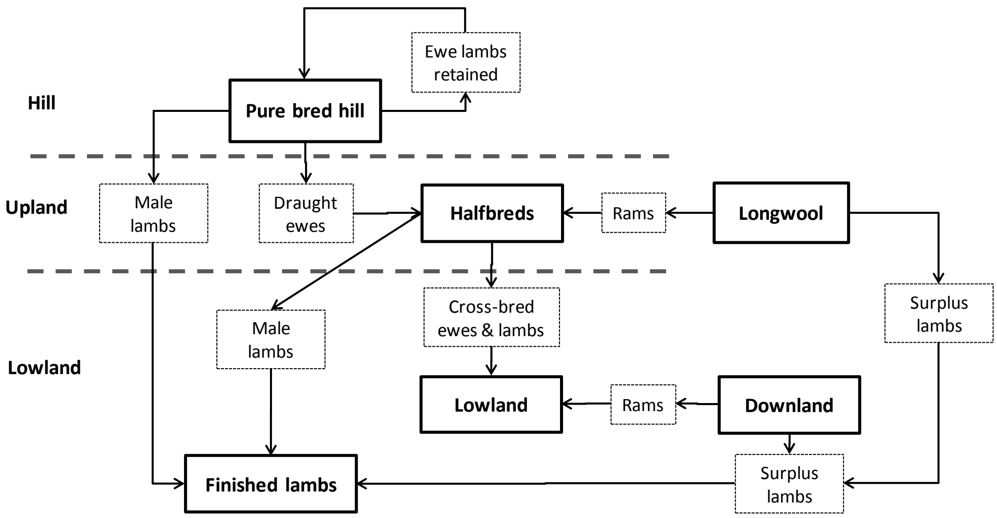


Figure 1. Simplified structural model of sheep production in the UK.

Thus, hill farms produce ewes that are sold to upland farms where they are crossed with high genetic merit rams to produce ewes that are, in turn, sold for breeding to lowland farms. In addition, there are purebred flocks with intermediate characteristics. Whilst most male lambs on lowland farms can be finished (i.e. reach slaughter weight and condition) before the autumn, a decreasing proportion on upland and hill farms can achieve this and many must be sold to lower land farms, where they are grown and finished over winter or kept as "stores", typically kept on maintenance rations until being finished the next spring. A small proportion of lowland flocks lamb very early and lambs are reared with a high concentrate diet to meet the Easter market (high prices). Lambing then proceeds with the rest of the lowlands, upland and hills in sequence.

Sheep are major emitters of enteric methane and so contribute about 8% the UK national agricultural methane emissions and about 7% of direct agricultural N_2O emissions (Sneddon et al., 2010). The sheep sector has historically operated at the margins of livestock production in the UK, but is important in supporting rural communities as well as particular grassland ecosystems and utilising these relatively poor grasslands.

The LCA analysis of the production of lamb meat thus has to take into account the different sizes of the breeds and consequent feed requirements, different types of land and consequent yields of grass (and management requirements), and different rates of lamb growth and ewe productivity.

The Cranfield LCA model addresses this using systems modelling to link the various sub-systems. The equations linking the systems ensure that the numbers are coherent and enable various options to be analysed. Options that can be explored may be economic, long term breeding goals, such as greater fecundity or policy oriented, such as concentrating sheep production in the hills or lowlands, as land demand changes.

2. Methods

2.1 Functional unit

The functional unit (FU) is 1000 kg expected edible lamb carcass at the national level. This is derived as liveweight for slaughter multiplied by the killing out percentage (47%), so that the system boundary is the farm gate, but the FU represents the useful carcass weight that can be expected. The relatively small burdens of transport and slaughtering are not included. Adult sheep meat and wool are co-products.

2.2 Biophysical performance of flocks

The basis of the model is the definition of the biophysical performance of each type of sheep flock. These include terms for ewe and ram longevity, fecundity (lambs per ewe per year), ram/ewe ratios, growth rate, mature body weights and mortality rates. This gives the data to calculate flock replacements required and hence how many surplus lambs are produced for exports to other flocks or for slaughter (Table 1). Data came from standard agricultural texts (e.g. Nix, 2005; Agro Business Consultants, 2005), contact with the industry and further research (Warkup et al., 2008). It is important to note that a distribution of lamb weights comes from each system, not just an average. Hill sheep are clearly smaller and shorter lived than lowland ones.

Table 1. Main characteristics of the sheep flocks

	Hill pure bred flocks	**Upland purebred flocks**	**Upland crossbred flocks**	**Lowland purebred flocks**	**Lowland crossbred flocks**
Ewe life, years	3.5	4.3	4.3	4.3	4.3
Ram life, years	3	3	3	3	3
Ewes/ram	47	47	47	20	20
Annual ewe culling rate	5%	4%	4%	4%	4%
Annual ewe mortality rate	3%	3%	3%	3%	3%
Lambs per year	1.0	1.4	1.4	1.5	1.6
Ewe weight, kg	52	62	76	74	78
Lamb daily liveweight gain, kg/d	0.10	0.14	0.17	0.18	0.19
Lamb mean final liveweight, kg	28	34	38	38	39
Distribution of lamb weights					
25-32, kg	92%	29%	2%	2%	1%
32-36, kg	8%	49%	23%	21%	17%
36-39, kg	0%	18%	38%	37%	35%
39-45.5, kg	0%	4%	36%	40%	46%
Wool, kg	2.1	2.9	2.9	3.1	3.1

2.2 Structural model of the national flock

The individual flocks are linked by a set of linear equations to allow transfers between them (e.g. a store lamb moves from the hills to an upland or lowland flock to be finished). If the balance of production systems is changed, the model recalculates the systems in order to supply the functional unit. The balance may be altered by changing the productivity of one part, e.g. more lambs per ewe in upland flocks, or by changing the proportions of hill, upland and lowland flocks. The linking equations have the following structure. The solution is the amount, X, of each activity, i, that produces the desired mass of the functional unit, Z,

$$Z = \sum_{i=1}^{n} z_i X_i \quad \text{Eq. 1}$$

where z_i is the output (types of sheep) of activity i, and also satisfies the set of flows between activities:

$$\sum_{i=1}^{n} c_{ij} X_i = 0,\ j = 1 \ldots p$$

Eq. 2

where c_{ij} is the supply or demand of j by activity i, which describes the relationship between enterprises. Demands are negative and supplies are positive and total supply must equal total demand. For example, purebred lowland flocks produce rams, which are, in turn, demanded as terminal sires by lowland finishing flocks. The total amount of material k flowing into the system is:

$$M_k = \sum_{i=1}^{n} m_{ik} X_i, k = 1 \ldots q$$

Eq. 3

where m_{ik} is the flow of material k into activity i. The life cycle inventory (LCI) for the system is the total of each burden l

$$B_l = \sum_{k=1}^{p} M_k b_{kl}, l = 1 \ldots r$$

Eq. 4

where b_{kl} is the amount of burden l produced by the use or disposal of material k and M_k is the total amount of material. The LCI identifies the contribution of each material

$$B_{kl} = M_k b_{kl}$$

Eq. 5

or activity

$$B_{il} = X_i \sum_{k=1}^{q} m_{ik} b_{kl}$$

Eq. 6

2.3 Feed demands

The main demand for feed energy is met by grazing, with relatively small contributions from conserved forage or concentrates. The energy demand of a ewe was calculated from the AFRC feeding system (Alderman and Cottrill, 1993), much of which is derived from ARC (1980). These include terms for maintenance (a function of body weight, wool production and activity), pregnancy (a function of lamb weight) and lactation until weaning, a function of lamb size and daily liveweight gain (DLWG). The metabolisable energy demand is calculated per ewe each year and includes the energy needs for lambs until they leave the system.

Energy demand is converted to a demand for grazing dry matter intake (DMI) by allowing for the management choice of the quantitative supply of concentrates and conserved forage to lambs and ewes and thus obtaining the grazing energy demand by difference. This is converted to DMI from the energy density of grazed pasture. The combination of grazing, conserved forage and concentrates normally meets sheep protein requirements when energy demand is met. So, these are not explicitly calculated separately.

2.4 Grazing land requirements

A grass sub-model was developed to calculate the area of grass required together with the managerial inputs needed. Grass yield was modelled using the grass site class system (Brockman and Gwynn, 1988). Site classes range from 1 to 7, with 7 being least productive. The main determinants of site class are summer rainfall (low rainfall restricts production) and soil texture (lighter soils drain more quickly and have lower yield potentials). An allowance is also made for reduced productivity at higher altitudes. Effects of site class and N application rate on grass yield are shown in Figure 1.

In a grazing system, N may be supplied applied fertiliser, fixation by clover, atmospheric deposition and by the animals' excreta. This causes the organic matter to build up and cycle round the system to become available to both the pasture and loss processes to air and water. The resulting system can be described by a system of equations which can be solved for a steady state.

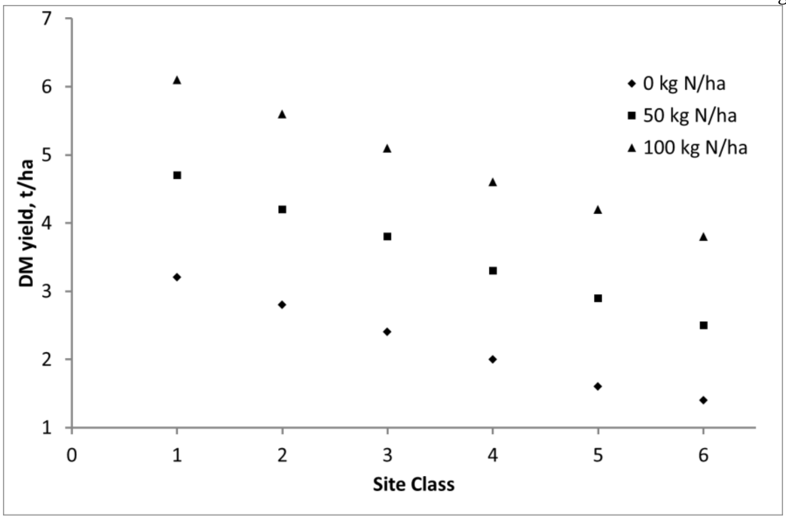


Figure 1. Effects of site class and N application rate on grass DM yield

The total nitrogen available annually to the grass crop is:

$$N_a = N_{atm} + N_c + N_f + N_s + N_e, \qquad \text{Eq. 7}$$

where N_{atm} is the atmospheric nitrogen deposition, which ranges from 15-35 kgN/ha, N_c is the nitrogen fixed by the clover proportion, N_f is the fertiliser N applied, N_s is the nitrogen surplus not leached or denitrified over the winter, N_e is the nitrogen in grazed animal excreta, which is not lost.
The N content of the grass (g_N) is a function of the N available.

$$g_N = 20.14 + 0.0136(N_a - N_{atm}) \qquad \text{Eq. 8}$$

The total N taken up by the crop is thus $g_N Y_{GDM}$. The balance is thus at risk of loss to air and water. The proportions emitted to air and water are given by Sandars et al., (2003). When grazing, 20% of the dry matter yield is assumed to be spoilt by trampling and defoliation and thus unavailable for consumption by the animal and returned as organic matter to the soil. When grazing, sheep utilise about 6% of forage N and the balance is excreted. Of the excreted proportion, 70% is urine and 10% of the dung is soluble. 15% of this soluble N is volatised as ammonia and the balance of excreted N becomes incrementally available to crops, N_e. This system of equations is solved (iteratively) for N_a and the resulting N_f were found to be comparable with typical values of fertiliser application as found in the British Survey of Fertiliser Practice (BSFP, 2005).
The proportions of grass productivities to sheep types was described by Williams et al., (2006). This also describes other managerial inputs to grassland, e.g. reseeding rates and silage harvesting.

2.5 Other LCI data

LCI data, such as the burdens of concentrate production and direct management energy requirements came from Williams et al., (2006). Emissions of ammonia were calculated from N excretion (N_{EX}) at grazing and in housing using values from the UK ammonia inventory (Misselbrook et al., 2010). Nitrous oxide emissions from N_{EX} during grazing followed the IPPC (2006). Nitrous oxide and methane emissions from manure management were derived from the UK 1997 inventories for GHG emissions (Sneath, 1997, Chadwick, 1997). Enteric methane was calculated from the IPCC (2006) Tier 2 formula in which an average gross energy density of 18.4 MJ/kg DM was used (McDonald et al., 2011).

2.6 Implementation

The model was implemented in Microsoft Excel, using code in VBA for Applications to solve the equations. The results were expressed as LCIs using the IPCC (2007) coefficients for GWP and the CML coefficients for other impacts.

2.7 Scenarios

Scenarios were investigated to consider changes in sheep management that represented the ways farmers may respond to economic drivers, changes in genetic potential and management quality (including animal health) and the emission factor for enteric methane. Results were compared with the current, baseline state of sheep production and considered energy use, GHGE land occupation. All changes in scenario variables caused effects that were linear or close to being linear within the range explored.

3. Results

With the current national flock, it takes about 22 GJ/t to produce 1 t expected edible carcass weight with the emission of about 22 t CO_2e/t, of which about 67% was from enteric methane (Table 2). "Good quality" land included grassland from site classes 1-3 and arable land for crop production, while poor quality land included grassland of site classes 4 to 7.

Table 2 Baseline burdens of producing 1 t expected edible carcass weight of lamb at the national level.

CED, GJ	GWP, t CO_2e	Eutrophication potential, kg PO_4 eqv.	Acidification potential, kg SO_2 eqv.	Abiotic resource use, kg Sb	Good quality land, ha	Poor quality land, ha	All land, ha
22.2	22.3	100	87	13	0.5	10.0	10.5

Halving the proportion of the flock that is lowland could result from economic drivers. It increases land occupation (mainly on poorer land) and causes a relatively small increase in GHGE (from more slow growing sheep) and decrease in CED from lower N fertilisation and concentrate use (Table 3).

Table 3 Results of changing scenarios on main burdens of producing lamb. Values are the ratio of scenario results over baseline results.

			Effect of results				
					Land occupation		
Scenario description	**Baseline value**	**Effect of scenario**	**CED**	**GWP**	**Good quality**	**Poor quality**	**Total**
Proportion of ewes on lowland	37%	Halve	93%	102%	107%	125%	125%
		Double	112	97%	88%	58%	60%
Proportion lowland lambs finished in early spring	10%	Halve	99%	100%	99%	100%	100%
		Double	103%	101%	103%	100%	100%
Increase sheep weight	66 kg#	+5%	113%	113%	113%	109%	109%
		+10%	124%	126%	123%	119%	119%
Increase in fecundity	1.2 lambs per ewe#	-10%	111%	108%	111%	107%	107%
		+10%	96%	95%	96%	95%	95%
Increase in killing out percentage	47%	+2.35%	99%	97%	99%	96%	96%
		+4.7%	95%	93%	95%	92%	92%
Increase in ewe longevity, years	3.7#	+0.5 year	100%	98%	100%	98%	98%
		+1 year	98%	96%	98%	96%	96%
Combined improvements Low			86%	88%	86%	89%	89%
Combined improvements High			76%	78%	77%	80%	80%

Note: # Weighted average across sheep types

Doubling the proportion of early spring lamb production increase GHGE and CED through demanding more concentrates, but has no effect on land occupation. Increasing sheep weight (ewes and lambs) increases all burdens, mainly because of the extra maintenance overheads of larger breeding sheep. Increasing the fecundity of ewes by 10% reduces all burdens by 5-6%, while decreasing fecundity by 5% (e.g. through ill health, such as Schmallenberg virus) increases burdens by 6-8%. Increasing the killing out percentage from 47% to 51.7% (i.e. relative increase of 10%) reduces all burdens by 8%. Increasing ewe longevity by 0.5 years (about 13% increase) reduces burdens by about 5%.

The scenario variables that are under most managerial and technical influence were combined at low and high levels to assess the potential for improvement (this excluded a change in the proportions of the national flock). The low level improvements combined to reduce burdens by about 14% and the high level improvements by about 24% (Table 3).

4. Discussion

The results show similarity with other studies in the UK, e.g. Taylor et al., (2010) found the carbon footprint of Welsh lamb to be 24 kg CO_2e/kg. The results are very sensitive to the enteric methane emission factor and method of calculation. This is under active research in work to improve the UK GHG inventory for agriculture (http://www.ghgplatform.org.uk/). Applying a Tier 2 approach will lead to different outcomes from a Tier 1 approach, which assumes the same emission rate for all sizes and diets, which is far from reality is sheep production. Any change in the structure of the industry that maintains output, but at different altitudes is clearly constrained by land availability. The potential for improvement is substantial, but depends on a synergistic combination of improvements in animal health, management and breeding. These are often only enabled by a positive economic climate, to which government policies often play a major role.

5. Conclusion

The systems approach to LCA is a powerful tool for assessing the complexity that is the UK sheep industry. It allows exploration of alternative approaches. Potential for improvement through technical and managerial changes exists and could reduce burdens significantly. Changes in the structure of the industry could have profound impacts on land occupation, particularly the balance between good and poor quality land.

6. Acknowledgments

The authors thank the UK Department for the Environment, Food and Rural Affairs for funding and Dr Stewart Ledgard of AgResearch, NZ for helpful comments on the model.

7. References

Agro Business Consultants, 2005. The Agricultural Budgeting and Costing Book, (59th edition). Agro Business Consultants, Melton Mobray.

Alderman, G. and Cottrill, B., 1993. Energy and protein requirements of ruminants. An advisory manual prepared by the AFRC Technical Committee on Responses to Nutrients. CAB International, Wallingford.

ARC, 1980. The Nutrient requirements of ruminant livestock: technical review. Published on behalf of the Agricultural Research Council by the Commonwealth Agricultural Bureaux, Wallingford, UK.

Brockman, J.S. and Gwynn, P.E.J., 1988. Journal of the British Grassland Society, 19, 169-155.

BSFP, 2005. British Survey of Fertiliser Practice. Defra, London.

McDonald, P., Greenhalgh, J.F.D., Morgan, C.A., Edwards, R., Sinclair, L., Wilkinson, R., 2011. Animal Nutrition. 7th Edition, Prentice Hall, Harlow, UK.

Measuring holistic carbon footprints for lamb and beef farms in the Cambrian mountains initiative.

Misselbrook, T.H., Chadwick, D.R., Gilhespy, S.L., Chambers, B.J., Smith, K.A., Williams, J, Dragosits, U, 2010. Inventory of Ammonia Emissions from UK Agriculture 2009. DEFRA Contract AC0112 Inventory Submission Report October 2010, http://preview.tinyurl.com/UK-NH3-Inv

Nix, J.S., 2005. Farm Management Pocketbook, 36th (2006) Edition. The Andersons Centre, Imperial College London, Wye Campus.

Sandars, D.L., Audsley, E., Canete, C., Cumby, T.R., Scotford, I.M. and Williams, A.G., 2003. Environmental Benefits of Livestock Manure Management Practices and Technology by Life Cycle Assessment, Biosystems Engineering, 84,, 3., 267-281.

Sneddon, S., Brophy, N., Li, Y., MacCarthy, J., Martinez, C., Murrells, T., Passant, N., Thomas, J., Thistlethwaite, G., Tsagatakis, I., Walker, H., Thomson, A., Cardenas, L., 2010. Greenhouse Gas Inventories for England, Scotland, Wales and Northern Ireland: 1990 – 2008. AEAT, Harwell. http://preview.tinyurl.com/UK-GHGI-2008

Taylor, R., Jones, A. Edwards-Jones, G., 2010. Countryside Commission for Wales Policy Research Report No. 10/8. http://preview.tinyurl.com/CCW-Sheep-CF

Warkup, CC., Jones, H.E., Audsley, E, Williams, A.G., 2008. A study of the scope for the application of research in animal genomics and breeding to reduce nitrogen and methane emissions from livestock based food chains. Final report to Defra on project AC0204.

Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Final report to Defra on project IS0205. http://preview.tinyurl.com/Cranfield-IS0205

Carbon footprint of sheep farms in France and New Zealand and methodology analysis

Armelle Gac[1], Stewart Ledgard[2,*], Elise Lorinquer[1], Mark Boyes[2], André Le Gall[1]

[1] Institut de l'Elevage, Building and Environment Department, BP 85225, F-35652 Le Rheu, France
[2] AgResearch, Ruakura Research Station, Hamilton, New Zealand
[*] Corresponding author. E-mail: *elise.lorinquer@idele.fr*

ABSTRACT

For eco-labelling of products, the environmental performance is an important factor and this can be significant for meat products. This study deals with the carbon footprint of French and New Zealand lamb production at the cradle-to-farm gate. In France, the analysis was performed on two contrasting systems: “in-shed lamb” *vs* “grass lamb”, while in New Zealand, the main “grass” system was studied. The average carbon footprint of “grass” (19 farms) and “in shed” (85farms) French lamb was 12.7 and 12.9 kg CO_2 eq/kg live weight (LW) sold, respectively, and of the main New Zealand system (151 farms) was 8.52 kg CO_2 eq/kg LW The results were highly dependent on methodological choices, especially regarding enteric fermentation assessment and allocation. These results provide insights into requirements for building an internationally common framework to allow comparison of results.

Keywords: life cycle assessment, lamb, farming systems, methodology

1. Introduction

The environment is becoming increasingly important to consumers and there are economic implications in the comparison between products based on their environmental performance. For eco-labelling of products, this issue is particularly important for meat products because of their significant impact on climate change (Hamershlag, 2011). In this context, environmental assessments of farm systems and agricultural products are becoming essential at an international scale. This should include identification of methodology issues to ensure reliable and equitable comparisons. These environmental assessments will also increase understanding of the different emissions sources in order to identify options to reduce them.

This study deals with the carbon footprint of French and New Zealand lamb production. These two countries are both major lamb producers and are reacting positively to environmental concerns about their production. New Zealand is the third largest sheep meat producer in the world (0.6 Mt) and is one of the largest exporters to Europe (FAO). It has initiated environmental research to respond to the eco-labelling initiatives in Europe (e.g. Ledgard et al., 2010). France has the fourth largest ovine flock in Europe and the planned move to environmental labelling of products by the end of 2012 has lead to several recent or ongoing studies on agricultural products (e.g. Dollé et al., 2011, Van der Werf et al., 2010). This study has been led by two applied research teams who have already worked on the topic of environmental impacts of lamb production. The objectives were to analyse the differences between systems and countries and to share knowledge about methodological issues, through collaboration (Lorinquer, 2011).

2. Methods

2.1. The lamb production systems

The analysis was performed in the two countries for the year 2008, on a sample of farms. In France, sheep meat comes from a large diversity of farming systems, all with a significant part of the year where sheep are in-doors. A sample of farms representing two contrasting specialised systems was selected: “in-shed lamb” (85 survey farms with significant use of housing systems) *vs* “grass lamb” (19 survey farms with significant use of perennial grasslands) (Table 1). The data comes from the French Breeding Network database led by Institut de l’Elevage. In New Zealand, the main “grass” system (mixed sheep and beef on North Island hill country; perennial grasslands) was studied (Table 1). The data comes from 151 farms surveyed by Beef + Lamb New Zealand.

Most of the technical data were provided by those survey sources (size of the flock, weight, area of the farm, practices, etc.). Nitrogen excretion and the dry matter intake by the animals was assessed using information on the forage stocks and feed purchased by farms in France. In New Zealand it was calculated using a validated energy-requirement model (Clark et al., 2007; from the National Inventory methods) based on animal productivity data.

Table 1. Technical description of the lamb production systems studied.

Technical parameters	France "In-shed"	France "Grass"	New Zealand "Grass"
Number of survey farms	85	19	151
Main characteristics	Significant use of housing	Significant use of perennial grasslands	Significant use of perennial grasslands
Mixed (M) or specialised (S)	S	S	M
Average effective area (ha)	102	114	430
% of grassland in the area (%)	63%	88%	100%
Number of ewes / farm (head)	555	670	2000
Ewe productivity (lambs weaned / ewe/year)	1.35	1.19	1.21
Kg concentrate fed / ewe / year	210	145	0
Time of grazing (days / year)	205	280	365

2.2. Environmental assessment

The system boundary of the study covered the production of lamb from cradle-to-farm-gate. The functional unit is one kg of total sheep live weight exiting the farm gate. Capital, medicines and cleaning products were not included.

Each system was analysed using a methodology developed to fit its own country. Those methodologies are described respectively in the GES'TIM guidebook (Gac et al., 2010) for France and by Ledgard et al., (2010) for New Zealand. Table 2 lists the models used for each source of emissions, occurring on-farm and off-farm, as well as the atmospheric carbon stored by soil under pastures and hedges for France (as an assumption of the compensation of emissions). The impact on climate change was assessed by using the 100-year global warming potentials proposed by IPCC (2007).

To ensure comparability of the results, a common mass allocation was firstly used to allocate impacts between meat and wool, the two co-products of the sheep production systems.

Table 2. Sources of emissions and models used in France and New Zealand

Sources of emissions	Gas	France	New-Zealand
Enteric fermentation	CH_4	Vermorel et al., 2008	Clark et al., 2007 (Tier 2)
Manure in buildings	CH_4, N_2O	GES'TIM	-
Manure storage	CH_4, N_2O	GES'TIM	-
Grazing	CH_4, N_2O	GES'TIM	Overseer®
Nitrogen inputs - direct (fertilisers, crop residues) and indirect (volatilisation, leaching)	N_2O	IPCC 2006	Overseer®
Energy use on farm (fuel combustion)	CO_2	GES'TIM	
Inputs (e.g., feed, fertilisers, pesticides, energy purchased)	CO_2eq	GES'TIM	Ecoinvent, Ledgard et al., 2010, 2011
Carbon storage (pasture, hedges)	CO_2	Arrouays et al., 2002	-

Overseer® (Wheeler et al., 2007)

A sensitivity analysis was performed by comparing the same methodologies across both countries, i.e., the New Zealand data set was analysed with the French methodology and vice versa. Effects of allocation method were also tested, i.e., economic *vs* mass allocation.

3. Results

The average carbon footprint of French lamb was 12.9 kg CO_2 eq/kg live weight (LW) sold (Table 3). There was no significant difference between the average values for the two French systems (Grass 12.74 – 19 farms surveyed; In-shed 12.94 – 85 farms surveyed), but there was a high variability between farms within each system (standard deviations of 2.9 and 2.8, respectively). The carbon footprint of the main New Zealand system was 8.52 kg CO_2 eq/kg LW.

Table 3. Carbon footprint of French and New Zealand lamb at the farm gate and the relative contribution from different sources of emissions (using mass allocation).

	France Average	France "In-shed"	France "Grass"	New Zealand "Grass"
Carbon footprint (kg CO_2 eq/ kg LW)	12.9	12.9	12.7	8.5
Enteric fermentation (%)	53.3	53.1	54.1	72.5
Manure in buildings (%)	5.7	5.9	4.5	-
Manure storage (%)	3.7	3.8	3.2	-
Fertiliser use (%)	6.3	6.5	5.5	3.5
Grazing (%)	10.6	9.9	13.3	20.1
Energy use on farm (%)	3.3	3.2	3.9	1.4
Inputs[1] (%)	17.0	17.5	15.0	2.5
Carbon compensation (%)	25.9	23.6	36.7	-

[1]Inputs: feed, fertilisers, energy purchased, ...

Carbon sequestration in pastoral soils can potentially have a significant effect on reducing the carbon footprint at farm level. In the French Grass system, application of the carbon sequestration method of Arrouays et al., (2002) was calculated to reduce GHG emissions by 36.7% (pasture constituting 88% of the farm area), while in the In-shed system pastures it reduced emissions by 23.6% (pasture constituting 63% of the farm area).

The sensitivity analysis showed that results were also highly dependent on methodological choices. The cross-test performed resulted in variation of results between -2.5% to +47%. In particular, two points can be mentioned. Methane from enteric fermentation, which is the main source of emissions in France (53%) and in New Zealand (73%) was calculated using tier 1 and 2 methods respectively.

The choice of allocation between meat and wool is also crucial; there was a small difference between countries when mass allocation was used and a much larger difference using economic allocation (Table 4). In New Zealand, wool has an economic value because of its use in carpet making, whereas in France it has little economic value (less than 1% of total economic returns in 2008). This led to variation of results from -5.9% to 11.7%.

Table 4. Percentage of the carbon footprint allocated to meat depending on the allocation methodology.

Country	Economic allocation	Mass allocation
France	99.7%	89.6%
New Zealand	78.0%	85.4%

4. Discussion

Previous lamb carbon footprint studies covering the cradle-to-farm-gate showed a wide variation in results (Table 5). As in our study, this could be explained by technical characteristics of the systems and also by methodological choices. Most of those studies represented European production systems. Our results are in the range of those presented, especially for France.

Table 5. Carbon footprints of lamb from countries in some publications

Publication	Country	Carbon footprints in kg CO_2eq/ kg LW average and (range of values)
Ledgard et al., 2010	New Zealand	8.6
Leip et al., 2010	Europe	9.5[1]
Dollé et al., 2001	France	9.7 (8.3-11.7)
Benoit et al., 2010	France	9.7 [1]
Williams et al., 2006	UK	6.6 (4.7-8.2)[1]
Eblex, 2009	UK	6.9[1]

[1] The results were published with the FU 1 kg of carcass weight meat. Conversion into LW using a standard killing-out percentage of 47%.

The higher carbon footprint of French lamb was due to the use of external feed inputs and the fact that sheep are housed in-shed for part of the year in all French systems, with emissions from manure management (especially for the In-shed system in both cases). Conversely, in New Zealand, where productivity is often higher due to warmer climatic conditions, the animals stay outside all year round eating perennial pastures and therefore there are no gaseous emissions linked to external food production and manure management.

The methodological choices influence the results. Allocation procedure is a crucial factor: ideally, it has to be common between countries but the choice should be meaningful in every context. That was not the case for economic allocation, usually used in New Zealand, because in France there was no real market for wool. Another factor is the method of modelling of the systems and the environmental fluxes. In France, a lot has been done concerning data collection at farm level to assess, among others the animal intake and excretion, while the models concerning the emissions were quite simple (emission factors). In contrast, in New Zealand, the pasture intake was modelled due to inaccuracy in estimating it in hill country conditions. It is recommended that sufficiently detailed models are used that have been tested and validated for pasture types and farm systems and adapted for the country of interest. In that way the models will reflect the main determining factors and be used to identify options for increasing feed conversion efficiency and reducing the carbon footprint.

To make the results comparable between countries, there is a need to build a common methodology about the carbon footprint of lamb. This requires a common framework with respect to some methodological rules but with the possibility for countries to take into account their specificities and to use their own validated models or parameters.

5. Conclusions

Our study provides data on the carbon footprint of lamb for two countries with specific production systems. Methodological issues were very important because of their influence on the results. Currently, because of the potential economical stakes linked to this topic and the need to build an international common framework to allow comparison of results, a range of international groups are working on a standard for carbon footprinting of lamb, through the initiative of the Beef + Lamb New Zealand organisation. This should be available in 2013. This group has also already identified requirements for future work, in the areas of soil carbon sequestration and development of a biophysical allocation approach for meat *vs* wool.

In the future, it will be important to obtain results over several years to analyse the annual variability of the results. Indeed, it is known that the practices could vary a lot from year to year, due to climatic or economic conditions, such as drought, high costs of some inputs, diseases and market opportunities.

Learnings from studies such as this one, involving research teams from several countries, enables identification of key factors for carbon footprint and life cycle assessment methodology, to ensure objective comparability of the ecological performance of systems.

6. Acknowledgements

The authors want to thank the farmers of the French Breeding Network and Beef+LambNZ for farm data, Interbev (the French Meat sector board), Institut de l'Elevage and AgResearch for its funding.

7. References

Arrouays, D., Balesdent, J., Germon, J.C., Jayet, P.A., Soussana, J.F., Stengel, P., 2002. Contribution à la lutte contre l'effet de serre. Stocker du carbone dans les sols agricoles de France? Expertise scientifique collective. INRA report. 332 pp.

Benoit, M., Laigner G., Roulenc, M., 2010. Emissions de gaz à effet de serre et consommations d'énergie en élevage ovin viande. Renc. Rech. Ruminants 17, 351-354.

Clark, H., Kelliher, F.M., Waghorn, G.C., Johnstone, P., Rys, G., 2007. An updated New Zealand enteric methane inventory calculated using algorithms developed from an analysis of New Zealand experiments conducted between 1997 and 2005. Greenhouse gas and Animal Agriculture Conference Book. Christchurch, New Zealand. p. 163.

Dollé J.B., Manneville V., Gac A., Charpiot A., 2011. Emissions de gaz à effet de serre et consommations d'énergie des viandes bovines et ovines françaises: revue bibliographique et évaluations sur l'amont agricole. Institut de l'Elevage, France.

Eblex, 2009. Change in the air. The English beef and sheep production roadmap-Phase 1. AHDB. 50 p.

FAO. Faostat.fao.org

Gac, A., Cariolle, M., Deltour, L., Dollé, J.B., Espagnol, S., Flénet, F., Guingand, N., Lagadec, S., Le Gall, A., Lellahi, A., Malaval, C., Ponchant, P., Tailleur, A., 2010. GES'TIM — Guide méthodologique pour l'estimation des impacts des activités agricoles sur l'effet de serre. Réalisé dans le cadre du projet «Gaz à Effet de Serre et Stockage de Carbone en exploitations agricoles» (CASDAR 6147).

Hamershlag, K., 2011. Meat eater's guide to climate change and health. Environmental Working Group, USA.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (Eds). Published: IGES, Japan.

IPCC, 2007. IPCC Fourth Assessment Report (AR4) — Climate Change 2007. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Working Group I Report "The Physical Science Basis". Cambridge University Press, Cambridge, UK, New York, NY, USA.

Ledgard, S.F., Lieffering, M., McDevitt, J., Boyes, M. and Kemp, R., 2010. A greenhouse gas footprint study for exported New Zealand lamb. Report for Meat Industry Association, Ballance Agri-nutrients, Landcorp and MAF. AgResearch, Hamilton. 26p.

Ledgard, S.F., Boyes, M. and Brentrup, F., 2011. Life cycle assessment of local and imported fertilisers used on New Zealand farms. In: Currie, L.D., Christensen, C.L. (Eds), "Adding to the knowledge base for the nutrient manager". http://flrc.massey.ac.nz/publications.html. Occasional Report No. 24, Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand. 13p.

Leip, A., Weiss, F., Wassenaar, T., Perez, I., Fellmann, T., Loudjani, P., Tubiello, F., Grandgirard, D., Monni, S., Biala, K., 2010. Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS), final report. Europ. Comm. Joint Res. Centre, 32p.

Lorinquer, E., 2011. L'analyse environnementale de la viande ovine française et néo-zélandaise: Enjeux-Résultats. Doit-on aller vers une harmonisation des méthodes ? Mémoire de fin d'études. ENSAIA. 39 p.

Van der Werf, H.M.G., Gaillard, G., Biard, Y., Koch, P., Basset-Mens, C., Gac, A., Lellahi, A., Deltour, L., 2010. Creation of a public LCA database of French agricultural raw products: agriBALYSE. Proceedings of the 7th International Conference on Life Cycle Assessment in the Agri-Food Sector. Bari, Italy, September 22-24, 2010.

Vermorel, M., Jouany, J.P., Eugène, M., Sauvant, D., Noblet, J., Dourmad, J.Y., 2008. Evaluation quantitative des émissions de méthane entérique par les animaux d'élevage en 2007 en France. INRA Prod. Anim. 21, 403–418.

Wheeler, D.M., Ledgard, S.F., de Klein, C.A.M., 2007. Using the OVERSEER nutrient budget model to estimate on-farm greenhouse gas emissions. Australian Journal of Experimental Agriculture 48, 99-103.

Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report Defra Research Project ISO205, Bedford: Cranfield University and Defra.

Regionalised land use impact modelling of milk production in the U.S.

Rosie Saad[1,*], Dominique Maxime[2], Manuele Margni[2]

[1] Quantis, 395 Laurier Ouest, Montreal, QC, H2V 2K3, Canada ;
[2] CIRAIG, Mathematical and Industrial Engineering Departement, P.O. Box 6079, École Polytechnique de Montréal (Qc), Canada H3C 3A7
[*] Corresponding author. E-mail: *rosie.saad@quantis-intl.com*

ABSTRACT

As a natural resource essential in the food and agriculture sector, the sustainable use of land requires informed decisions based on a meaningful impact assessment. However, most studies addressing milk and dairy-based products account for land use impacts as an inventory data related, by simply reporting the amount in square meters of land occupied or transformed within a time span. Part of an US fluid milk beyond carbon Life Cycle Assessment study, this project aims to provide insights on a regionalized assessment of land use impacts related to milk production. The main focus is to assess the role of dairy livestock, for which different supply mix models and feed produced at different states locations were assumed in the ration. A spatial differentiated approach was considered, using regionalized impact factors at the state level to address potential land use impacts.

Keywords: land use, impact, spatial, milk, LCA

1. Introduction

Land is a natural resource that is often taken for granted, yet essential in the food and agriculture sector. Thus, a sustainable use of land requires informed decisions based on a meaningful impact assessment in Life Cycle Assessment (LCA), a decision-support tool used to address potential environmental impacts of a product over its entire life cycle. So far, most studies addressing milk and dairy-based products account for land use impacts based on an inventory data related, simply by reporting the amount in hectare or square meters of land occupied or transformed within a time span (Thomassen et al., 2008; Basset-Mens et al., 2009; Roy et al., 2009).

In light of its extremely slow renewal rate (Pimentel et al., 1987), soil is considered to be a non-renewable resource. Though unintentional, mechanisms of natural environment quality deterioration induce substantial changes in land cover and land use, constituting a primary source of soil degradation (Lambin et al., 2001). While such impacts are highly relevant from an environmental perspective, the development of land use impact assessment methods are still in their infancy. Most methods found in the literature account for impacts on biodiversity and are limited to one geographical scope (e.g. Europe) resulting in damages to ecosystem quality. Recent developments have advanced the necessity to address impacts on soil ecological functions and their contribution to ecosystem services (Mila i Canals et al., 2007; LULCIA, 2008-2012).

1.1. Goal and scope

This project is part of US fluid milk comprehensive LCA study (Jolliet et al., 2012). The system boundaries include the upstream processes up to the farm-gate level, namely feed production (on farm and purchased), fertilisers and pesticides application, animal and manure management as well as the downstream processes, such as milk processing, transportation/distribution, retail and use at the consumer level. The functional unit is defined as one kg of fat protein corrected milk (FPCM).

The main goal of this paper is to provide insights on a regionalised assessment of land use impacts related to milk production in the US with a particular focus on the feed production life cycle stage and soil ecological functions impact indicators and biodiversity.

2. Methods

2.1. Life Cycle Inventory

From a Life Cycle inventory perspective, land occupation processes needed at each life cycle stage are accounted for. The corresponding inventory flows are measured in units of area used during a period of time (m^2.year). For feed production, land occupation flows refer to agricultural activities, which were assumed to occur in existing agricultural land in the U.S. Thus, no land transformation was assumed. In addition, it is considered that the land is occupied during the year long for agricultural use, independently of the length of the growing season.

Given data availability limitations on agricultural land uses and growing seasons in the same year, no allocation among different types of agricultural production (crop rotations) was established. Allocation between milk and meat was done based on the quantity of feed required to make milk and meat, as feeds have different available energies for growth and lactation. Other co-products allocations include soybean and soybean meal, corn grain and dried distillers grains with solubles (DDGS).

To perform land use impact assessment related to feed production, feed mix in the rations and crop production data were collected. Average rations (in kg dry matter / kg FPCM) that were identified in the Milk carbon footprint study (Thoma et al., 2010) and calculated for each of all five major dairy productions regions were considered. In addition, annual data on crop production and area harvested were collected at the state level for each feed component from the United States Department of Agriculture (USDA) and National Agricultural Statistics Service (NASS) (USDA NASS, 2010). These parameters were also used to calculate the corresponding annual yields of production.

Eleven main feed components were considered in the ration, including:

- commodity crops that are traded and shipped on national scale markets are assumed to be produced in any state
- domestic crops such as light forage and haylage are assumed to be produced within a region (out of the five defined regions) where the farm is located
- pasture and local crops that are heavy or not transportable feed are assumed to be grown locally on-farm

Table 1 shows the different feed supply mix models that were assumed for a spatial impact assessment related to feed production.

Table 1. Supply mix models for each feed components

Supply mix models	**Feed components**
National, based on inter stated transport data	DDGS dry
	DDGS wet
	Corn grain
	Soybean
	Soybean meal
State	Alfalfa silage
	Corn silage
	Grass pasture
	Grass silage
Regional	Alfalfa hay
	Grass hay

2.2. Life Cycle Impact Assessment

From a life cycle impact assessment (LCIA) perspective, the method used is built upon recommendations of previously published works as well as the framework and the principles established by the UNEP/SETAC Life Cycle Initiative Task Force on Land Use LCIA phase 1 (Milà i Canals et al., 2007) and phase 2 (LULCIA, 2008-2012). Unlike traditional categories that assess impacts of emissions, land use impact characterisation is not structured according to the conventional steps of fate, exposure and effect, nor is it based on mass and energy balance. Rather, the assessment stems from the change in environmental quality (EQ) induced by land use over time (Fig. 1).

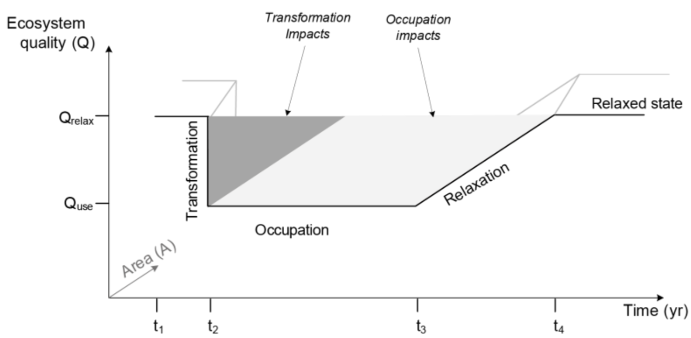


Figure 1. Environmental quality curve (adapted from Milà i Canals et al., (2007), Saad et al., (2011))

The general equation used to quantify land use impact is shown in Eq.1.

$$\mathrm{LOI} = \mathit{Inventory}\mathrm{flow} \times \mathrm{CF}_{occ} = \mathrm{A.t}_{occ} \int \Delta(E\mathrm{Q(t)}).\mathrm{dt} \qquad \text{Eq.1}$$

where LOI corresponds to land occupation impacts, A is the used land surface area (m^2), t_{occ} the occupation time (the product of both term being the inventory flow) [m^2.yr] and CF_{occ} is the characterisation factor for the occupation [m^2·yr·impact_indicator].

2.3. Characterisation factor data

Location is required to enable the link between the inventory flow and a spatial impact assessment using regionalised impact factors, also known as characterisation factors (CFs).

Different impact pathways are considered, impacts on biodiversity and the impacts on a series of ecosystem services. CFs from Pfister et al., (2010) are used for impacts on biodiversity. Such factors are based on the ones provided in EI 99 (Goedkoop and Spriensma, 2001) and include ecosystem scarcity and vulnerability. CFs for impacts on soil ecological functions, namely groundwater recharge potential, erosion resistance potential, physico-chemical filtration and mechanical filtration potential are provided by Saad et al., (submitted).

An example of regionalised CFs for land use agricultural impacts on biodiversity adapted to the states geographical context is illustrated in Figure 2. The CF for the remaining impact categories are given in the main report (Jolliet et al., 2012).

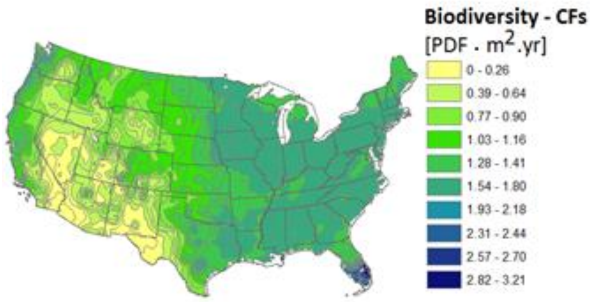


Figure 2. Land use impact factor for biodiversity for the U.S. geographical context

3. Results

3.1 Inventory flow for feed production

Yield production measured in kg crop dry matter produced per unit surface (kg DM/m^2) is a main parameter that enables converting a crop quantity needed to be grown into the necessary harvested area. Thus, the smaller the yield, the larger the harvested area is required. As illustrated in Figure 3, yields for corn grain vary considerably across the states from 0.4 to 1.1 kg DM/m^2. In addition to Western U.S. states, where Washington has the highest yield, states located in the Midwest, not only have high yields but also high productivity (annual total bushels produced). These high-yield regions require a smaller harvested area than states located in the South-eastern U.S., which have smaller yields. In general, between 1.5 and 2.5 m^2 of land area are needed to be harvested to produce 1 kg DM of corn grain.

Combining state corn grain yields (on the y-axis) with national production fractions (on the x-axis), the variable width graph presented in Figure 4 indicates the total land use area requirement needed in each state to produce one kg of corn grain at the national level. Results per state, shown as the areas of the rectangles, highlight states in the Corn Belt (Iowa, Illinois, Nebraska, Minnesota and Indiana) having high crop production fractions and moderate yields. Conversely, states like South Dakota have slightly smaller yields (higher land area) but smaller production, thus do not have sizeable contributions to corn grain land use area requirements at the national level. Total land use impacts, reported as land use area requirements, for national corn grain production sum up to 1.26 m^2.yr/kg DM.

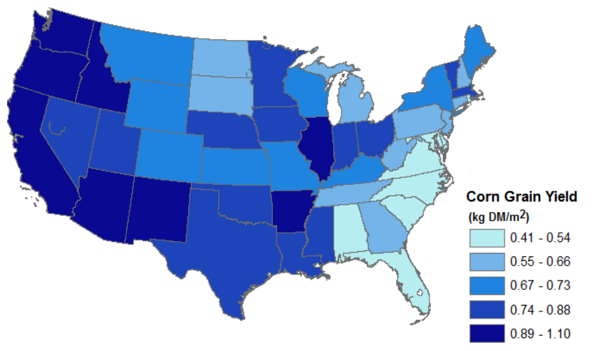


Figure 3. Yield production of corn grain per state (kg DM corn grain/m^2)

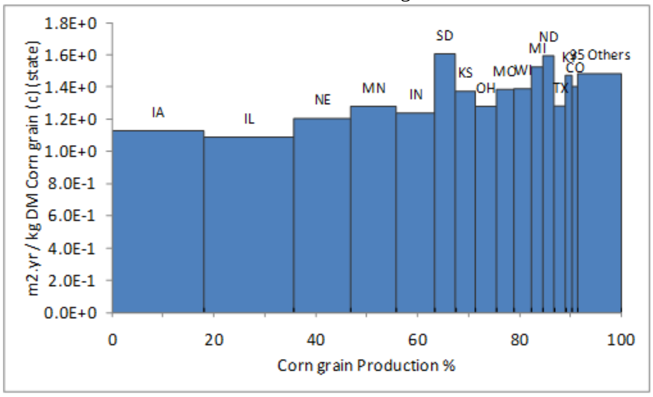


Figure 4. Land use area requirements and contribution to corn grain national production per state (m^2·yr/kg DM corn grain)

3.2 Land use impacts related to feed production

Part of the national impact assessment, Figure 5 indicates land use impact factors on biodiversity for the production of corn grain in each state. Although results show that highest potential impacts occur in Florida and Alabama, such states do not necessarily produce significant amounts of corn grain. Thus, potential land use impacts on biodiversity were quantified based on two key-parameters. These are indicated in the variable width graph presented in Figure 6; the land use impact on biodiversity due to state production of corn grain (on the y axis) and the national crop production fraction (on the x axis).

Typical land use impacts on biodiversity are expressed in units of potentially disappeared fraction (PDF) of species on a unit surface. Results are shown as the area of each state rectangle, quantifying land use impacts per kg of crop (PDF.m^2.yr /kg DM) at the national level. A state with a high crop production fraction and a high impact factor on biodiversity contributes to a larger share of the national impacts of the crop produced. States with the highest impacts are Indiana (18% of total share), Illinois (16%) and Iowa (16%). Land use impacts on biodiversity for a national corn grain production sum up to a total of 1.24 m^2.yr.PDF/kg DM.

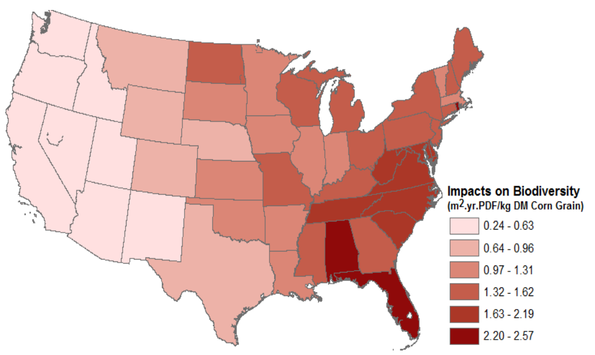


Figure 5. Impacts on biodiversity per state per kg DM corn grain (m^2.yr.PDF/kg DM corn grain)

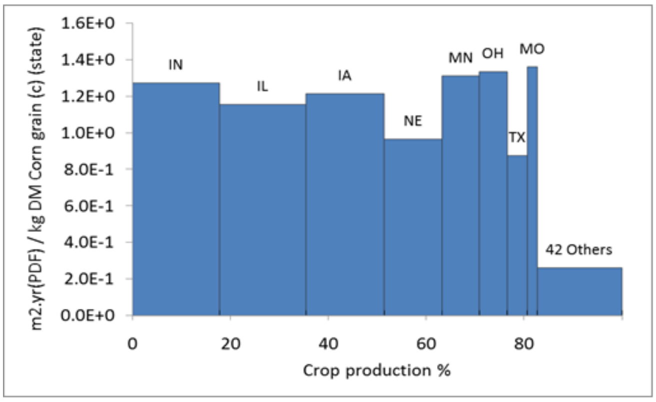


Figure 6. Land use impacts on biodiversity and contribution to the national corn grain production per state (m^2.yr.PDF/kg DM corn grain)

3.3 Land use impacts related to milk production

When accounting for a national production of 1 kg of milk, overall results indicate a large variation of land use area requirements (in m^2.yr/kg milk) across states. Considering all types of crop in the feed, California, Wisconsin and New York share the largest total land use area requirement to produce 1 kg milk at the national level. This result is driven by these states' large contribution to national milk production of 22%, 13% and 6.5% respectively (shown in Figure 7).

Results in terms of land use impacts on biodiversity are shown in the variable width graph presented in Figure 8. California requires a larger land area for producing 1 kg of milk at the national level, and shows a greater share of the national-level impact, which is mainly due to a large national milk production share.

Total land use impacts on biodiversity across states, when accounting for all crops, are equal to 1.37 m^2.yr.PDF/kg milk at the national level. California and Wisconsin contribute to 17% and 15% respectively.

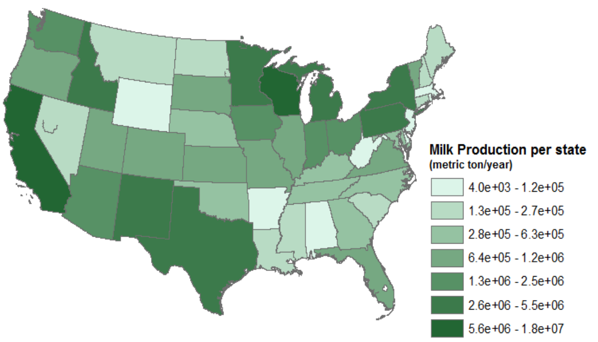


Figure 7. Milk production per state (metric ton/year)

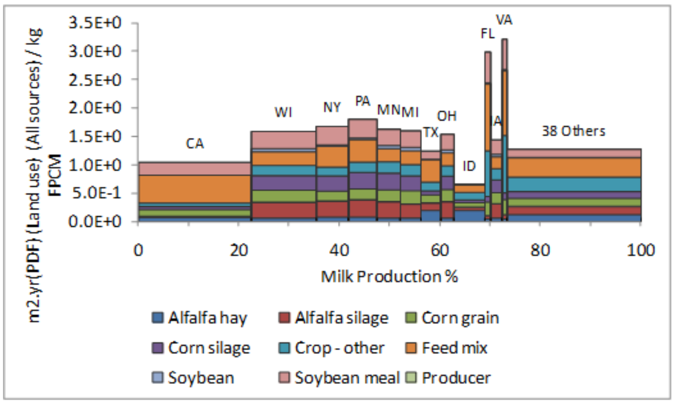


Figure 8. Land use impacts on biodiversity and contribution to the milk at the national level production per state (m^2.yr.PDF/kg FPCM)

4. Discussion

Land use area requirements, which are mainly driven by the yield, for each crop to be produced in the feed ration varied up to 6 times from one crop to another. Another key-parameter is the crop production contribution per state. Results of land use impacts per kg of milk produced from state to state varied up to 15 times for nationally produced crops. A comparison of the results indicate different share of contribution to the total impacts and for which the crop provenance could be traced back to its production location.

Given the distribution of feed components' production and their share to the national level production, states such as California and Wisconsin, which are major contributors to the national milk production require significant amount of crop to feed cows. However, not being a sizeable in-state corn grain production, they require a large supply of feed from other states. The latter are located in the Corn Belt region and constitute the main suppliers' to many other dairy producers states (i.e. receiver states). Consequently, land use impacts on biodiversity per kg of milk produced are greater than the ones induced by Minnesota's milk production, as it only contributes to 4% of the national milk production. Thus, impacts from an inducer perspective are larger than those from a receiver perspective. This reasoning applies to feed components, which their supply mix is assumed to be a national production (eg. commodity crops such as corn grain). For local and not transportable crops, assumed to be produced within the state where the farm is located, their contribution to land use impacts of milk production at the national level is mainly local. Thus, induced and received impacts are equivalent.

As part of the spatial assessment, results for land use impacts on a series of ecosystem services (groundwater recharge potential, erosion resistance potential, physico-chemical filtration and mechanical filtration potential) wer also accounted for. Impact results were mainly driven by the milk production fraction at national scale and thus aligned with the one observed for the biodiversity indicator. California, Wisconsin, Pennsylvania and New York were consistently among the top contributors to total impact (results not shown). Detailed results can be foun in the original report (Jolliet et al., 2012).

5. Conclusion

Land use impacts related to milk production across its entire life cycle are mainly driven by land use needed for feed production, which is required for dairy consumption. The remaining processes beyond the farm gate contribute to 5% of the total impact to biodiversity (results not shown). These are dominated by forest land use necessary for pulp production used in manufacturing paperboard cartons and other packaging.

We found that impacts of feed vary significantly depending on the production location. Both inventory and impact assessment contributes to this variation. Inventory is directly influenced by the yield that determines the area requirements per unit of crops production. Impact assessment is directly influenced by several bio-geographic factors, such as climate conditions, vegetation patterns and soil type properties. We found that spatial differentiation at the state level (the chosen regionalisation scale for this study) is as a key element when addressing land use impacts. However, future work and improvement could address a finer and more relevant scale assessment. The latter can be addressed on U.S. ecoregions scale level instead of being based on states' level. This could bring additional discrimination to the results considering more specific and spatially-differentiated CFs.

When inventory data are available, it is shown that spatial differentiation increases the discriminating power of LCA.

Finally, when considering the global environmental profile related to the US dairy production industry, possible trade-offs between impact categories (land use, water use, climate change, etc.) may arise. In this respect this work provides valuable insights to guide decision makers to identify priorities for action to reduce overall potential impacts related to milk production.

6. References

Basset-Mens, C., Ledgard, S., Boyes, M., 2009. Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecological Economics 68(6): 1615-1625

Goedkoop M, Spriensma R. (2001): Eco-indicator 99, a damage oriented LCA impact assessment method. Methodology report Nr. 1999/36a. 3rd ed. The Netherlands, Amersfoort.

Lambin, E. F., Turner, B. L., Geist, H. J., Agbola, S. B., Angelsen, A., Bruce, J. W., Coomes, O. T., Dirzo, R., Fischer, G., Folke, C., George, P. S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E. F., Mortimore, M., Ramakrishnan, P. S., Richards, J. F., Skånes, H., Steffen, W., Stone, G. D., Svedin, U., Veldkamp, T. A., Vogel C., Xu, J., 2001. The causes of land-use and land-cover change: moving beyond the myths. Global Environmental Change 11(4): 261-269

LULCIA (2008–2012). Operational Characterization Factors for Land use Impacts on Biodiversity and Ecosystem Services in the Life-Cycle Impact Assessment—Compatible with the Framework of the UNEP- SETAC Life Cycle Initiative (LULCIA). http://www.pes.uni-bayreuth.de/en/research/projects/LULCIA/

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007. Key Elements in a Framework for land Use Impact Assessment within LCA Int J LCA, 12(1), 5-15.

Pimentel, D., Allen, Beers, J., Guinand, L., Linder, R., McLaughlin, P., Meer, B., Musonda, D., Perdue, D., Poisson, S., Siebert, S., Stoner, K., Salazar, R., Hawkins, A., 1987. World Agriculture and Soil Erosion. BioScience 37(4): 277-283

Pfister, S., Curran, M., Koehler, A., Hellweg, S., 2010. Trade-offs between land and water use: regionalized impacts of energy crops. In: Proceedings of the7th International Conference on LCA in the Agri-Food Sector, Bari, Italy

Roy, P., D. Nei, T. Orikasa, Q. Xu, H. Okadome, N. Nakamura and T. Shiina (2009): A review of life cycle assessment (LCA) on some food products. Journal of Food Engineering 90 (1): 1-10

Saad R., Koellner, T., Margni, M., (submitted). Land use impacts on freshwater regulation, erosion regulation and water purification: a spatial approach for a global scale level. Int J LCA

Saad R, Margni M., Koellner T., Wittstock B., Deschênes L., 2011. Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. The International Journal of Life Cycle Assessment 16(3): 198-211

Thoma, G., Popp, J., Shonnard, D., Nutter, D., Ulrich, R., Matlock, M.,Kim, D, Neiderman, Z., East, C., Adom, F., Kemper, N., Mayes, A., 2010. Greenhouse Gas Emissions from Production in Fluid Milk in the US. Dairy Management, Inc., Rosemount, IL, USA

Thomassen, M., R. Dalgaard, R. Heijungs and I. de Boer (2008): Attributional and consequential LCA of milk production. The International Journal of Life Cycle Assessment 13 (4): 339-349

USDA NASS, 2010. USDA/NASS - Data and Statistics - QuickStats 2.0. http://quickstats.nass.usda.gov/

Jolliet, O., Asselin, A., Burek, J., Heller, M., Henderson, A., Humbert, S., Kim, D. S., Lessard, L., Matlock, M., Margni, M., Saad, R., Thoma, G., Vionnet, S. (in alphabetical order), (2012): US fluid milk comprehensive LCA, Report in progress, www.css.snre.umich.edu/project/us-fluid-milk-beyond-carbon-lca-study

Greenhouse gas emissions from production of imported and local cattle feed

Lisbeth Mogensen*, Troels Kristensen, Thu Lan T. Nguyen, Marie T. Knudsen

Dept. of Agroecology, Aarhus University, Blichers Allé 20, Postbox 50, DK-8830 Tjele, Denmark
* Corresponding author. E-mail: Lisbeth.Mogensen@*agrsci.dk*

ABSTRACT

Emissions related to feed production are hotspots in milk production. In this paper, carbon footprint (CF) of different feedstuffs is estimated, taking into account contribution from growing, processing, transport and land use change (LUC). Subsequently, the effect of a 'local' versus an 'import' feeding strategy on the GHG emissions from feed was investigated. There were large variations in CF of different feedstuffs and between concentrated feed and home grown roughage. When calculating CF of a complete feed ration for cows by attributional LCA, the main reason for differences was related to contribution from transport, and especially from LUC. However, if calculated by consequential LCA there was no differences between a 'local' and an 'import' strategy regarding GHG emissions from feed production.

Keywords: animal feed, green house gas, transport, land use change, attributional and consequential LCA

1. Introduction

Studies of carbon footprint (CF) of milk have shown that 80-90% of the total greenhouse gas (GHG) emissions are related to processes before the milk leaves the farm (Hermansen and Kristensen, 2011). On the farm, methane emission from enteric fermentation gives the highest single contribution to CF of milk followed by the emissions related to feed production (Kristensen et al., 2011). The hypothesis of the present study was that by choice of feed, farmers can reduce CF of milk.

2. Methods

The aim of this paper is to 1) estimate carbon footprint (CF) of different feedstuffs for dairy cattle and 2) compare two feeding strategies for dairy cows regarding GHG emissions from the feed production.

The functional unit (FU) is '1 kg dry matter (DM) of feed ready to eat' (1) and for the comparison of the two feeding strategies, the FU is 'amount of feed for production of 1 kg milk (ECM)' (2). The main system studied was production of conventional grown fodder crops at dairy farms in Denmark. Besides that, a system producing soybean from which soybean meal is extracted – assumed to be located in Brazil - was also included.

An attributional life cycle assessment (ALCA) approach has been used, handling co-products by economic allocation and taking into account LUC which was quantified based on PAS2050 (BSI,2008). For comparison, a consequential approach (CLCA) was applied, where co-products were handled by system expansion and contribution from indirect LUC was estimated according to Audsley et al., (2009). The same inventory data on the production of fertiliser, manure, diesel, and electricity were used for both ALCA and CLCA.

The rate of resource use, energy use and output in relation to growing 1 ha of different crops is presented in Table 1. Regarding soybean, data related to growing the crop in Brazil were taken from Ecoinvent (2010). For other crops, data represent Danish average production level.

Crop yield and input of fertiliser and manure

Data for crop yield and input of fertiliser for different crops are the national norms (Plantedirektoratet, 2010) and correspond to the typical level found at Danish dairy farms (Kristensen et al., 2011). For the roughage produced at the farm, manure produced at the same farm is used at a rate up to 170 kg total N/ha. The gap between the input of 170 kg N from manure and N norm was assumed to be filled by supplying fertiliser N. According to the national rules (Plantedirektoratet, 2010), 70% of N in cattle slurry could be utilised by the plants. In relation to the issue of allocation of emissions burden created by the two sources, manure spreading and manure in housing and storage, emissions from the former were fully allocated to crop production, whereas those from the latter to the milk production. Barley and rape seed were assumed to be imported from other Danish farms where they are grown without input of manure.

Table 1. Resource use, energy use and output per ha of crops (farm gate)

Feed		**Grass silage**	**Maize silage**	**Barley**	**Rape seed**	**Soybean**
Place of production		Denmark Dairy farm	Denmark Dairy farrn	Denmark Crop farm	Denmark Crop farm	Brazil
Input						
Mineral fertiliser[1)], kg N/ha		118	39	119	178	0,3
Mineral fertiliser, kg P/ha		32	45	23	30	13
Mineral fertiliser, kg K/ha		192	139	49	82	25
Manure, kg total N		170	170	0	0	0
Diesel, /ha		102	207	118	118	26
Seed, kg		13	5	150		106
Electricity, kWh/ha[2)]		378	34	17	23	0
Output						
Crop yield,	kg DM/ha	8269	11325	4121	3321	2300
	(kg/ha)	(-)	(-)	(4848)	(3590)	(2544)

1) Calcium ammonium nitrate
2) Electricity used for watering was assumed to be based on natural gas

Estimation of emissions

Factors used for estimating emissions are shown in Table 2. For the Danish grown crops, factors for estimating nitrous oxide emissions were from IPCC (2006) whereas those for ammonia emission were national emission factors. For soybean, data from Ecoinvent (2010) was updated using IPCC guidelines (IPCC, 2006).

Table 2. Factors for estimation of emissions from crop production and inventory of GHG emissions embodied in different inputs

	Emission source	Amount		Emission Factor (EF)	Source - EF
N_2O direct, kg	Spreading of				
	- slurry	kg N in manure ab storage		0.01	1)
	- fertiliser	N in fertiliser		0.01	
	Crop residues,	Crop:	kg N/ha/year	0.01	1) + 2)
	kg N pr ha per year	Grass	60		
		Maize, whole crop	25		
		Other arable crops	28		
NH_3-N, kg	Spreading of				2)
	- slurry	kg N in manure ab storage		0.12	
	- fertiliser	N in fertiliser		0.022	2)
	Crop residues	Grass		0.5 kg / ha	3)
		Other arable crops		2.0 kg / ha	
N_2O, kg indirect	From NH_3	NH_3-N		0.01	1)
	From leaching	NO_3-N=0.33*(manure-N + fertiliser-N)		0.0075	4) + 1)
Input	Fertiliser-N	4.4 kg CO_2/kg N			5)
	Fertiliser-P	2.7 kg CO_2/kg P			6)
	Fertiliser-K	0.8 kg CO_2/kg K			6)
	Seed	0.4 kg CO_2/kg			7)
	Diesel	3.3 kg CO_2/L			6)
	Electricity (natural gas based)	0.66 kg CO_2/kWh			6)

1) IPCC (2006)
2) Mikkelsen et al., (2006)
3) Gyldenkærne and Albrektsen (2008)
4) Nielsen et al., (2009)
5) Elsgaard (2010). GHG emissions from average N-fertiliser (Calcium ammonium nitrate) used in Denmark: 60% of this N-fertiliser is imported from Yara, where it is produced in accordance with Best Available Techniques for N_2O emission reduction , and 40% of the N-fertiliser is imported from Baltic
6) Nielsen et al., (2003)
7) GHG emissions rate from growing 1 kg barley was used for approximation of GHG emissions per kg seed

Place of origin
Data on Danish import and export of different foodstuffs (www.statistikbanken.dk) used to quantify the proportion of feed import. The amount of feed imported from different countries was quantified by using FAO Stat data (www.faostat.fao.org). For calculating the contribution from feed transport the actual place of origin was used. Whereas, when calculating the contribution from growing the crop, only one place of origin was included for each crop. For example, growing of soybeans is only based on production conditions in Brazil.

Feed transport
The inventory data for GHG emission per tkm feed import were taken from LCAFood database (Nielsen et al., 2003) which are based on ETH data. Data on distances and means of transportation were obtained from the feed industry and the literature.

Co-product handling – ALCA and allocation
In the calculation based on ALCA, economic allocation was used to distribute the environmental burdens among co-products. One hectare planted with rapeseed yields 3590 kg, of which 36.4% is extracted as rapeseed oil and 61.6% as rapeseed cake (Dalgaard et al., 2008). The CF of rapeseed (708 g CO_2e/kg) was allocated by a ratio of 24% to the rapeseed cake and 76% to the oil taking into account the difference in the prices of the two products (information from FAO). The energy consumption for processing one tonne of rapeseed was 50 kWh electricity and 340 MJ heat (Dalgaard et al., 2008). Soybeans yield in Brazil is 2544 kg/ha (Ecoinvent, 2010) of which 15.8% is extracted as soy oil and 82.6% as soybean meal (Dalgaard et al., 2008). The CF of soybean (399 g CO_2e /kg) was divided between soybean meal and oil by a ratio of 66.3:33.7. We assumed that the soybean was milled where it was produced. The use of energy for processing 1 kg soybean meal creates 34 g CO_2e (Ecoinvent, 2010).

Co-product handling - CLCA and system expansion
When calculating CF of feeds in a consequential approach, it is sufficient to have LCA data on those crops, whose production is affected by an increased demand for feed (e.g., in the present case the concentrated feed) in the market. In the CLCA calculation of CF of soybean meal we used system expansion. How an increased demand for soybean meal affects agricultural production follows the soybean loop suggested by Dalgaard et al., (2008), where soybean oil is sold on the market and assumed to substitute palm oil, which is a mix of palm oil and palm kernel oil. As with the ALCA calculation of the CF of soybean meal, the CLCA calculation was based on data taken from Ecoinvent (2010) on soybean cultivation with the update of emissions estimate taking into account the 2006 IPCC Guidelines for National Greenhouse Gas Inventories.
The CLCA calculation of CF of rapeseed meal was based on the assumption by Dalgaard et al., (2008) that an increased demand for rapeseed meal will not affect the production of rapeseed as the rapeseed oil is the main product, whereas the production of soybean meal and barley will be affected.

Land use chance (LUC)
In the ALCA calculation, GHG emissions from direct LUC (dLUC) were calculated according to PAS2050 (BSI, 2008) on the feeds grown where deforestation takes place. In the present study, only the use of soybean meal from South America was considered a source of GHG emissions from dLUC. Direct LUC resulting from the production of soybean meal in Brazil and Argentina would add 7.7 kg and 0.93 kg CO_2 per kg soybean meal, respectively. Import of soybean meal to Denmark is made up of 73% from Argentina and 17% from Brazil. A weighted average dLUC-GHG emissions rate of 2.0 kg CO_2/kg soybean meal was thus calculated.
In the CLCA calculation, GHG emissions from LUC were referred to as indirect use change (iLUC) emissions calculated according to Audsley et al., (2009) on all feeds. This is essentially based on the assumption that use of land for crop production would increase pressure on land use and thereby causes LUC somewhere in the world. From a total LUC-related-GHG emissions of 8.5 GT CO_2 per year, the fraction of 58% that agriculture is responsible for would result in a contribution of 1.43 t CO_2/ha/year, taking into account the total agriculture area of 3,475 Mha (Audsley et al., 2009). In the present study, iLUC was calculated by multiplication of land use rate (m^2a/kg feed) by an iLUC-GHG emissions factor of 143 g CO_2/m^2a.

Land use (LU) and soil carbon change
Enhancing carbon (C) sequestration in soil is a way to reduce GHG emissions. However, the size of the sequestration potential is debatable. In this paper, the effect of changes in soil C was included in a very simple

way, where the changes were assumed to depend only on type of land where crops are grown. According to Vleeshouwers and Verhagen (2002), grassland is a carbon sink (i.e. more carbon is stored than released) of 0.52 Mg C/ha/year (191 g CO_2/m^2/year), whereas cropland (arable land) is a carbon source (i.e. more carbon is released than stored) of 0.84 Mg C/ha/year (308 g CO_2/m^2/year).

Feeding strategies

Two different strategies for cow feeding were given, a ‘local’ and an ‘import’ strategy. In both strategies, the cows have the same level of feed intake (energy) and milk production. In the ‘local’ strategy all feeds are grown in Denmark and the concentrated protein feed is based on rapeseed cake and cereals (27% and 16% of energy requirement, respectively). In the ‘import’ strategy, 70% of the feeds are grown in Denmark and the concentrated protein feed is based on imported soybean meal and cereals (19% and 24% of energy requirement, respectively). In both strategies, the main roughage is maize silage (43%) which is supplemented with grass silage (14%).

3. Results

CF values of animal feeds, based on ALCA, are shown in Table 3a. Roughage is grown on the farm, so there is no contribution to its CF from transport. Roughage has relative high dry matter (DM) yields per ha, especially maize silage. Therefore CF of roughage (g CO_2e./kg DM) is lower than that of concentrated feeds. Rapeseed cake and soybean meal are both protein feeds and they are co-products from oilseed production. GHG emissions from growing and processing rapeseed cake and soybean meal are quite similar, but due to transport of soybean meal from South America, CF of soybean meal is nearly double that of rapeseed cake, and much higher if contribution from dLUC is also included.

Table 3a. Carbon footprint (CF) of conventionally grown animal feed – contribution from growing, processing, transport and LUC, per kg dry matter, based on ALCA

Feedstuff	**Origin**	**Contribution to CF, g CO_2e per kg DM**				**Land use**	
		Growing	Processing	Transport	Total	m^2/kg DM	dLUC g CO_2/kg DM
Grass silage	The farm	318	71 [1)]	0	389	1.15	0
Maize silage	The farm	147	63[1)]	0	210	0.89	0
Barley	Denmark	487	11	17	515	2.43	0
Rape seed cake	Denmark/ Germany	310	28	118	456	1.24	0
Soybean meal	Argentina/ Brazil	367	39	422	828	1.83	2288

[1] Including diesel used for traction and transport at the farm

Table 3b. Carbon footprint (CF) of conventionally grown animal feed – contribution from growing, processing, transport and LUC, per kg dry matter, based on CLCA

Feedstuff	**Origin**	**Contribution to CF, g CO_2e per kg DM**				**Land use**	
		Growing	Processing	Transport	Total	m^2/kg DM	iLUC g CO_2/kg DM
Barley	Denmark	528	11	18	557	2.42	346
Rapseed cake	Denmark/ Germany	368	0	118	486	2.24	320
Soy bean meal	Argentina	150	59	463	672	2.41	345

As can be seen in Table 4, in the attributional calculation (ALCA), there is little difference in GHG emissions related to growing and processing the mixture of feedstuffs between the two strategies. However, the inclusion of GHG contribution from transport increases the difference making the ‘local’ strategy a moderately better choice than ‘import’ strategy. The inclusion of GHG emissions from dLUC increases significantly the CF of feed in the ‘import’ strategy to a range that is double that in the ‘local’ strategy. As also seen in Table 4, if calculated based on CLCA there were no differences in the CF of the feed between the ‘local’ and ‘import’ strategy as GHG contribution from iLUC was similar for the two strategies.

Table 4. Effect of feeding strategy and LCA method (ACLA and CLCA) on GHG emission from feed production

	ALCA		CLCA	
Strategy	**Local**	**Import**	**Local**	**Import**
GHG from feed[1)], g CO_2e/kg ECM				
From growing the feed	0.20	0.21	0.23	0.19
From processing the feed	0.01	0.01	0.00	0.01
From transport	0.02	0.05	0.02	0.05
From LUC (direct or indirect)	0	0.23	0.15	0.15
From LU	0.21	0.19	0.26	0.26
Total	0.22	0.26	0.26	0.25
Total, including LUC	0.22	0.48	0.41	0.40
Total, including LUC and LU	0.44	0.67	0.67	0.66
Land use, m^2/ kg ECM	0.90	1.01	1.07	1.06

[1] GHG emissions related to feed production (only for cows) calculated per kg milk produced

4. Discussion

From the present study it becomes clear that choice of method strongly affects the calculation of CF of feed. Which LCA approach should be used depends on the goal and scope of the study (Thomassen et al., 2008). ALCA seeks to quantify the fraction of the global environmental impact related to the product, whereas CLCA seeks to capture change in environmental impact as a consequence of a certain activity. The present study, using ALCA, showed a huge impact contribution from LUC when feeding is based on feed import from areas where deforestation takes place. However, the choice of method to account for emissions from LUC is critical, and so far there is no consensus on which method is best for inclusion of GHG emissions from LUC. Therefore, it is recommended to present the results in relation to LUC separately. The use of the CLCA method clarifies that for planning animal feeding in the future, a key measure is to reduce the land use rate per unit of product output, i.e. to increase land use efficiency. Land is a limited resource so that an increase in demand for land would cause indirect LUC elsewhere. In addition, it is important to consider soil carbon storage or loss potential of different land use types.

5. Conclusion

In conclusion, there were large variations in CF of different feedstuffs and especially between concentrated feed and home grown roughage. However, when calculating CF of a complete feed ration for cows using an attributional approach, the main reason for the difference was related to contribution from transport, and in particular from direct LUC. However, if calculated based on consequential LCA. there was only a minor difference in total GHG emissions between a ‘local’ and an ‘import’ feeding strategy, as the burden of indirect LUC was attributed to any type of feed, no matter it is local or imported.

6. Acknowledgement

This study was funded through the ÖSK Interreg IV A Programme project ‘REKS – Regional nöt och lammköttsproduktion – en tillväxtmotor’ funded by the European Union.

7. References

Audsley, E., Brander, M., Chatterton, J., Murphy-Bokern, D., Webster, C., and Williams, A. (2009). How low can we go? An assessment of greenhouse gas emissions from the UK food system and the scope for to reduction them by 2050. How low can we go? WWF- UK.

BSI, 2008. PAS2050. Specification for the assessment of life cycle greenhouse gas emissions of goods and services.

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W.A. 2008. LCA of soybean meal. Int. J. LCA. 13, 240-254.

Ecoinvent Centre, 2010. Ecoinvent Data v2.2. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland. http://www.ecoinvent.ch.

Elsgaard, L. 2010. GHG emission from cultivation of winter wheat and winter rapeseed for biofuels. Report requested by the Danish Ministry of Food, Agriculture and Fisheries. The faculty of Agricultural Sciences, Aarhus University. 34 pp.

Gyldenkærne, S., Albrektsen, R. 2008. Revurdering af ammoniakemissionen 2003-2007. Baggrundsnotat til vandmiljøplan III. Danmarks Miljøundersøgelser, Aarhus Universitet, Denmark.

Hermansen, J.E., Kristensen, T., 2011. Management options to reduce the carbon footprint of livestock products.Animal Frontier. 1 (1), 33-39.

IPCC, 2006. IPCC Guidelines for national greenhouse gas inventories. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

Kristensen, T, Mogensen, L., Knudsen, M.T., Hermansen, J.E., 2011. Effect of production system and farming strategy on greenhouse gas emissions from commercial dairy farms in a life cycle approach. Livestock Sci 140, 136-148.

Mikkelsen M.H., Gyldenkærne, S., Poulsen, H.D., Olesen, J.E., Sommer, S.G., 2006. Emission of ammonia, nitrous oxide and methane from Danish Agriculture 1985 – 2002. Methodology and Estimates. National Environmental Research Institute, Denmark. 90 pp –Research Notes from NERI No. 231. http://www.dmu.dk/Pub/AR231.pdf.

Nielsen, P.H., Nielsen, A.M., Weidema, B.P., Dalgaard, R., Halberg, N. 2003. LCA Food data Base. www.LcaFood.dk

Nielsen, O.-K., Lyck, E., Mikkelsen, M.H., Hoffmann, L., Gyldenkærne, S., Winther, M., Nielsen, M., Fauser, P., Thomsen, M., Plejdrup, M.S., Albrektsen, R., Hjelgaard, K., Vesterdal, L., Møller, I.S. & Baunbæk, L. 2009. Denmark's National Inventory Report 2009. Emission Inventories 1990-2007 - Submitted under the United Nations Framework Convention on Climate Change. National Environmental Research Institute, Aarhus University. 826 pp. – NERI Technical Report No. 724.

Plantedirektoratet, 2010. Vejledning om gødskning og harmoniregler. Ministeriet for Fødevarer. Landbrug og Fiskeri. 118 pp.

Thomassen, M., Dalgaard, R., Heijungs, R., de Boer, I., 2008. Atrtributional and Consequential LCA of milk production. International Journal of Life Cycle Assesment. 13, 339-349.

Vleeshouwers, L.M., Verhagen, A., 2002. Carbon emission and sequestration by agricultural land use: a model study for Europe. Global Change Biologi. 8, 519-530.

Environmental life cycle assessment of milk in Canada

Mia Lafontaine[1,*], Rosie Saad[1], Diane Gilbert[2], Manuele Margni[3]

[1] Quantis Canada, 395 Laurier Ouest, Montreal, QC, H2V 2K3, Canada
[2] Groupe AGECO Inc. 2014 Rue Cyrille-Duquet Bureau 307. Quebec, QC G1N 4N6, Canada
[3] CIRAIG, Department of Chemical Engineering, P.O. Box 6079, École Polytechnique de Montréal, QC, H3C 3A7, Canada
[*] Corresponding author. E-mail: *mia.lafontaine@quantis-intl.com*

ABSTRACT

As part of its initiatives to improve the environmental performance of dairy production in Canada, the Dairy Farmers of Canada commissioned in 2010 a full environmental and socio-economic life cycle assessment (LCA) for the production of milk across Canada. This paper summarises the environmental LCA methodology, results and conclusions. In line with existing publications, the environmental LCA found that the main sources of greenhouse gases were emitted at the farm, with enteric fermentation, manure management and feed production. Potential impacts on ecosystem quality were mostly caused by feed production, linked to eutrophication, land use, and potential ecotoxicity. Potential burdens on human health were dominated by ammonia emissions. The water footprint was strongly linked to geography and the use of irrigation. To understand the impact of the choices in practices, the analysis compared regionalised provincial average results. Scenarios based on practices were modelled to understand the scale of impact reduction possible.

Keywords: environmental impact, dairy, water footprint, regionalisation

1. Introduction

A Canadian milk life cycle assessment (LCA) study took place over two years, with the support of the Canadian Dairy Farmers, funding from Agriculture and Agri-Food Canada, and collaboration of many stakeholders, including provincial associations, researchers and stakeholders in the dairy sector. It was innovative in different ways, mainly by its scale and regionalisation of impact, and also by integrating the first Social LCA (SLCA) in the dairy sector. With a large inventory of farm datasets, environmental impacts were calculated based on provincial average inventories, allowing for interpretation of variable farming practices and their impact on overall performance, such as understanding how impacts were related to geo-physical conditions. Geographical coordinates and production volumes of 13,331 farms in Canada allowed for impact regionalisation considering location (ecoregions, watershed, etc.) to facilitate accurate estimation of potential impacts (potential acidification, eutrophication, ecotoxicity, human toxicity, land use and water use) in the different Canadian provinces. Results were calculated per province and weighted into a single environmental performance, while allowing for analysis of different practices.

2. Methods

The current study followed ISO 14040/14044 standards, as well as the International Dairy Federation (IDF)'s guidelines on LCA (IDF, 2010). As per the guidelines, the functional unit chosen was 1 kg of fat and protein corrected milk (FPCM). The boundaries of the study stopped at processing-plant front gate (farm gate + transportation). As per the guidelines, carbon sequestration in soils was excluded, and allocation between milk and meat was calculated according to a physicochemical equation, resulting in an average allocation to milk of 82%.

2.1. Life Cycle Inventory

A majority of activity data was sourced from on-farm surveys (Table 1).

Table 1. Data and their main sources

Data	Source
Diet percentages, manure storage practices	Cost of Production Surveys (Ontario, Quebec, New Brunswick, Nova Scotia, Prince Edward Island)
Manure spreading information, energy	Sheppard et al., (2011) survey on 500 farms
Equipment and energy used in feed production	Mail-in surveys (Alberta, Ontario)
Fertiliser types and concentrations	Sheppard et al., (2009) NH_3 emissions from fertilisers
Housing, energy & equipment in feed production	Ecoinvent models
Manure spreading tendencies	Provincial federations (most)
Crop yields, herd size, farm area	Statistics Canada

In total, cost of production surveys as well as mail-in surveys collected information from more than 300 farms in Alberta, Ontario, Quebec, and the Atlantic Provinces. A previous survey (Sheppard et al., 2011) supplied additional information on farm practices for all provinces. When no specific site data were available, or the contribution to impact was known to be minimal, life cycle inventory databases were used, mainly ecoinvent 2.2 (SCLCI, 2010). In the last resort, when assumptions were necessary because activity data were not available, expert judgements were used for validation. Models based on ecoinvent are summarised below, with adaptation described where relevant.

2.2. Life Cycle Inventory Assessment

The global framework adopted in this study (Fig. 1) is based on the peer-reviewed and internationally recognised LCIA method IMPACT 2002+ (Jolliet et al., 2003, updated by Humbert et al., 2011) with several novelties inspired by the work done for the development of IMPACT World+, including consistent spatially explicit levels and improved impact category modelling. Eighteen impact categories are included in this study. While they can be reported and interpreted separately, they can be modelled up to the five damage indicators: Climate Change, Natural Resources, Human Health, Ecosystem Quality and Water footprint, allowing their respective contribution to be put into perspective. IMPACT 2002+ grouping methodology was used to aggregate the midpoint indicators.

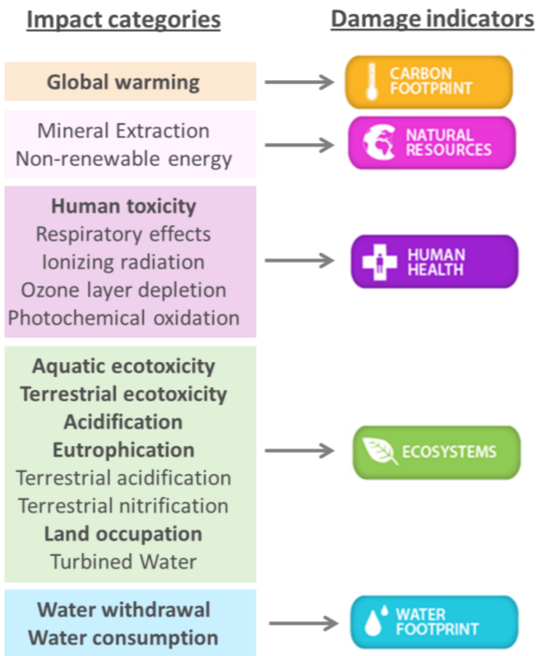


Figure 1. The LCIA framework used (regionalised impact categories in bold)

2.3. Regionalisation

Nine of the 18 impact categories were regionalised (bold, Fig. 1). Having a specific geographic context, this study considers a multi-scale spatially explicit life cycle approach for both inventory and impact assessment levels. Indeed, Canada is divided into distinct regions showing differences in land covers, vegetation patterns, climate and hydrological systems, and soil types. Spatial differentiation is important when quantifying the environmental footprint at each life cycle stage of Canadian milk production for regional impacts (e.g., acidification, eutrophication, smog) and local impacts (e.g., ecotoxicity). However, potential impacts at the global level (e.g., ozone depletion, global warming) are not affected by an emission's location.

Moreover, to perform regionalised assessment that accounts for spatial differentiation, impact indicators that address environmental problems in the agricultural sector and are highly sensitive to regional characteristics were included: water use, land use, acidification, eutrophication, toxicity and ecotoxicity. The framework used in this study and the methods underlying these regional-specific impact indicators are based on the IMPACT World+ LCIA method (CIRAIG et al., 2012).

3. Results

The average footprint of 1 kg of Canadian FPCM was calculated for each province using provincial averages of inventory flows and provincial weighted averages of characterisation factors based on farm locations and their production.

Table 2. Sources of impact and allocated profile

Damage Indicator	Main Contributor	Value	Unit
Climate Change	Enteric fermentation, feed production, manure	1.01	kg CO_2e
Ecosystem Quality	Land use, phosphorus eutrophication, emissions from coal for power, where applicable	14.95	PDF.m^2.yr
Human Health	NH_3 emissions (land use), NO_x, SO_2 (energy & diesel)	8.3e-7	DALY
Resource Depletion	Feed production, energy on farm, transportation	3.98	MJ
Water Consumption	Irrigation (when applicable), evaporation in energy production	61	L

3.1. Climate change

The distribution of greenhouse gas emissions (Fig. 2) was similar to that of other studies. While energy, transportation and buildings and equipment had little impact (8% of the total), the most important emissions were caused by CH_4 and N_2O emissions, occurring, in decreasing order, from enteric fermentation, manure storage and feed production.

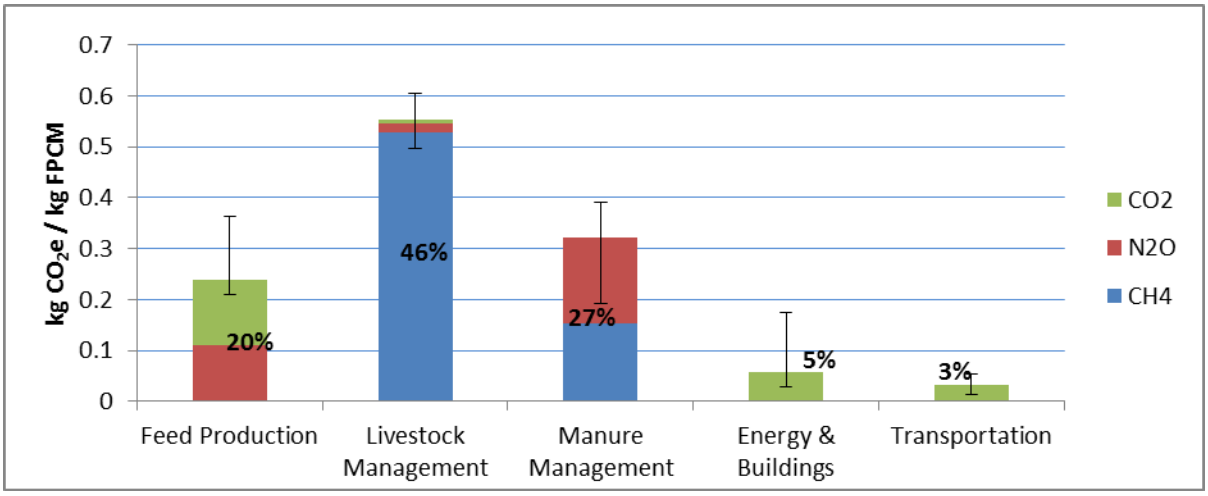


Figure 2. Distribution of average greenhouse gas emissions per kg of fat-and-protein-corrected milk (FPCM) across life cycle steps, with spread of provincial average results (error bars).

Variability in provincial results were highest (0.025-0.17 kg CO_2e/kg FPCM) for energy emissions, due to the variability of the electrical-grid-mix among provinces (14-293 g CO_2e/kWh). Variability was likely underestimated in feed production, due to the assumption that fertilisation recommendations were always followed. Still, based on soils and crops, manure spreading practices varied among provinces, while emission factors also varied based on geography, a result mostly linked to humidity (Rochette et al., 2008). Feed-production emissions were lowest in Alberta and Saskatchewan, because of their dry climate, while they were highest in British Columbia, due to a moist climate and high N recommendations for hay (200-300 kg N/ha, compared to less than 150 kg/ha elsewhere), followed by the Atlantic provinces, also because of moist climates, average yields and lower milk production per animal.

Variability in emissions from livestock management were linked to changing replacement-animal ratios (since their feed, digestion and manure is also included in the milk production system), as well as digestibility of feed, with concentrates, for example, having higher digestibility than forage. Finally, different types of manure storage contributed to variability in manure-management emissions.

3.2. Water footprint

Water use was influenced by two distinct scenarios: irrigated crops in western Canada and non-irrigated crops elsewhere. Water withdrawal, based on low-resolution statistics, reached 550 L/kg FPCM in the worst-case scenario (highest provincial irrigation per hectare) but was as low as 29 L/kg FPCM elsewhere. In terms of consumed water (removed from watershed through evapo(transpi)ration or incorporation in product), the this footprint ranged from 12-336 L/kg FPCM. With only 20% of milk produced in irrigated areas, the overall weighted average amounts to 61 L/kg FPCM.

The potential impacts on human health, ecosystem quality and resource depletion were assessed following different methodologies (including Maendly et al., 2010, Pfister et al., 2009,van Zelm et al., 2010,Verones et al., 2010) yet were shown to be negligible overall (< 1%). With regards to water scarcity, it is only critical in a specific location in the prairies that is not well represented in provincial averages yet still targets an important concern.

3.3. Ecosystem quality

In this category, potential impacts were highest for aquatic ecotoxicity, eutrophication and land use. In the first case, it was mainly the result of possible heavy-metal discharges to water associated with energy production for different life cycle stages. Zinc in mineral feed supplements also contributed. The two other sources of impact were linked to phosphorus fertilisation and arable land use.

Interestingly, all contributions displayed "ecosensitivity" based on farm location, as a function of geographic variation in characterisation factors (CFs), for example those for land use (Fig. 3). The effect is obvious when provincial values calculated using regional-level CFs are compared to those calculated using national-level CFs (Fig. 4). The regionalised provincial results range from 3-33 PDF*m^2*yr (median = 8), while the weighted mean equals 15 PDF*m^2*yr, illustrating that while half the provinces are below the median of 8 PDF*m^2*yr, most milk is produced in areas that are more ecosensitive, thereby driving up the weighted mean.

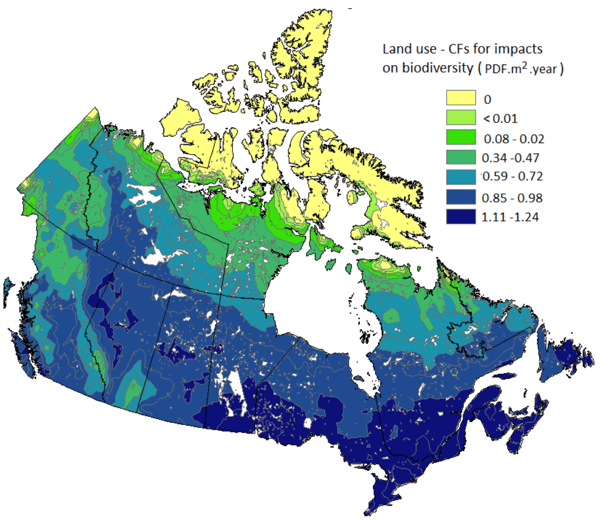


Figure 3. Map of regionalised land use characterisation factors (CFs)

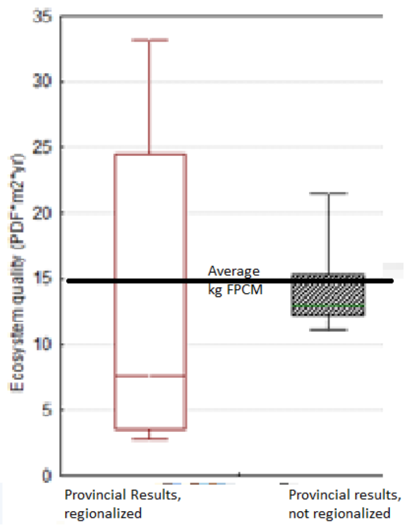


Figure 4. Provincial distribution of impact on ecosystem quality using regional-level ("regionalised") and national-level ("not regionalised") characterisation factors.

3.4. Human health

Potential effects on human health were modelled with six different impact categories, but only toxicity was regionalised. While toxicity plays a role in the overall impact (around 7%), ammonia emissions have the greatest influence (63% of potential impact). Ammonia emissions, from fertilisers, in housing and from manure storage, are linked to respiratory difficulties. The remainder of impacting substances also falls in the category of respiratory inorganics, related to fossil-fuel combustion (emissions of NO_x, SO_2, and hydrocarbons), electricity production and direct use. Potential toxicity impacts are also due to the heavy metal content of manure spread on crops not used in feed. Zinc, most notably, bio-accumulates, and excess zinc consumption (through crops) can interfere with the absorption of other essential minerals (ODS, 2011).

3.5. Hotspot assessment

Where hotspots were found, variability among provincial averages was analysed to understand the underlying trends. For climate change, the largest source of variability was the carbon footprint of energy used at the farm. This, in turn, was linked to the electrical-grid mix used in each province (0.025-0.17 kg CO_2e/kg FPCM), with provinces supplied by hydroelectricity having the smallest footprint.

Next in range of variability was manure management and feed production. For the former, the percentage contribution of CH_4 and N_2O varied furthermore, where solid storage caused higher N_2O emissions and liquid storage was dominated by CH_4 emissions. The province with the lowest manure footprint had 55% of farms using solid storage and tanks with a natural crust for liquid storage. Meanwhile, the province with the highest manure footprint had many liquid lagoons (37% of farms overall), driving its contribution much higher.

Table 3. Comparison of manure storage practices and their emissions (FPCM = fat-and-protein-corrected milk).

	Averaged Manag't	Solid Storage	Solid on Pasture	Liquid with Crust	Liquid with Cover	Liquid with No Cover	Liquid Lagoon
% of manure	Canadian avg:	34%	13%	37%	5%	8%	3%
kg CO_2e/kg FCPM	0.32	0.17	0.30	0.31	0.34	0.35	0.96
% CH_4	47%	15%	4%	40%	52%	61%	86%
% N_2O	53%	85%	96%	60%	48%	39%	14%

A similar evaluation of contributions was done for the feed-production stage (Table 4), especially since it was a hotspot in all impact categories. Rations in the Eastern provinces are more corn by-product rich while rations in the West use canola meal.

Table 4. Contributions of the feed production stage to impacts of crops and rations

	Hay	Corn Silage	Dry Corn	Small Grains	Soybeans	Rations East	Rations West
Overall weight	46%	20%	11%	10%	8%	6%	3%
Climate change	33%	10%	17%	15%	4%	13%	7%
Ecosystem quality	42%	10%	11%	8%	19%	6%	3%
Human health	39%	19%	14%	15%	8%	4%	2%

There was variability in contribution based on type of crop. Corn grain and small grains had relatively more impact on climate change, mainly to fertilisation rates, while soybeans had less. The same trend was observed for impact on human health, as ammonia emissions also depended on fertilisation rates. With potential impacts on ecosystem quality, there were different factors in play. Corn silage, for example, benefitted from the highest yield per hectare (approximately 13 t of dry matter (DM)/ha) while soybeans were in the lower range, with a yield around 2 t DM/ha, each affecting impacts on land use.

In evaluating sensitivity of certain parameters, it was important to consider the variable geopolitical and socio-economic context that influences practices, while remembering that agriculture is a complex system with many inter-related cause-effects chains that are difficult to model. With this in mind, and to perform a meaningful scenario analysis, a few "what-if" scenarios were modelled based on current practices and well-known alternatives (as opposed to marginal and emergent practices). These tested animal-replacement practices, alternative fertiliser types, fat supplements in feed, and manure management practices. While animal-replacement ratios affect the three main hotspots (feed production, enteric fermentation and manure storage), most of these options were limited in their overall impact and targeted different hotspots (data not shown).

4. Discussion

Applying LCA to production across Canada required a method that allowed and facilitated representation of differing provincial contexts, both in terms of practices (inventory) and geophysical conditions (CFs). Results showed that variability was driven by both aspects, depending on the indicator. By separating the two, it was easier to understand where reductions are possible and where observance of best practices is even more important (sensitive areas based on location).

Achieving consistency in data collection and interpretation across Canada, however, was a challenge. Additionally, while great uncertainty exists in modelling emissions from soils, variability is also great due to organic (manure) and synthetic fertiliser application dosages and techniques. For the most part of dosages, only recommendations exist, from which assumptions were derived.

5. Conclusion

The main contributions to impact came from feed production for four of the five endpoint indicators. For climate change, as confirmed in the literature, enteric fermentation is the most important source of emissions (46%), followed by manure management (27%) and feed production (20%).

The main sources of variability in climate change impacts among provinces were linked to on-farm energy use (due to different grid mixes), followed by manure management and feed production. Variability in results is expected to be underestimated in feed production, however, due to the assumption that fertilisation recommendations were followed.

The main sources of potential impact to ecosystem quality were caused by land use, eutrophication, and aquatic ecotoxicity from aluminium emissions derived from energy production and high copper concentrations in mineral supplements. This category was also the most sensitive to geographic locations of farms. For human health, impacts were driven by respiratory inorganics, mostly NH_3 emissions from fertilisers, in housing and from manure storage. They were followed by NO_x and SO_2 emissions associated with fossil-fuel use for energy production in the different stages of lifecycle.

Many ongoing research projects are evaluating mitigation options that would be worth modelling in "what-if" scenarios. Meanwhile, there are many aspects to consider when evaluating agricultural practices, and some economic or social trade-offs may require much more analysis. The current study helped outline the interchangeability of practices and the sensitivity of the geographical context to help guide best practices. This study acts as a first step in this direction, mapping the road to sustainable milk production in Canada.

6. References

CIRAIG, University of Michigan, Quantis and DTU, 2012. IMPACT World +: A new global regionalized life cycle impact assessment method. http://www.impactworldplus.org/en/index.php

IDF (International Dairy Federation), 2010. A Common Carbon Footprint Approach for Dairy: The IDF Guide to Standard Lifecycle Assessment Methodology for the Dairy Sector, http://www.idf-lca-guide.org/Files/media/Documents/445-2010-A-common-carbon-footprint-approach-for-dairy.pdf

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G and Rosenbaum R, 2003. IMPACT 2002+: A New Life Cycle Impact Assessment Methodology. Int J LCA 8(6): 324-330.

Humbert S, Margni M and Jolliet O, 2011. IMPACT 2002+ User Guide: Draft for version 2.2". Quantis, Lausanne, Switzerland. Available at: sebastien.humbert@quantis-intl.com

Pfister, S., Koehler, A., and Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption. Environmental Science and Technology, 43(11), 4098-4104.

Office of Dietary Supplements, 2011. Dietary Supplement Fact Sheet: Zinc, Office of Dietary Supplements, National Institutes of Health, USA Gov, http://ods.od.nih.gov/factsheets/Zinc-HealthProfessional/

Rochette, P., Worth, D., Lemke, B., McConkey B., Pennock, D., Wagner-Riddle C., Desjardins, R., 2008. Estimation of N2O emissions from agricultural soils in Canada. I. Development of a country-specific methodology, Canadian Journal of Soil Science, 2008, 88(5): 641-654

Sheppard S, Bittman S, and Bruulsema W, 2009. Monthly ammonia emissions from fertilizers in 12 Canadian ecoregions, Can. J. Soil. Sci, 90(1), 113-127

Sheppard SC, Bittman S, 2011. Farm survey used to guide estimates of nitrogen intake and ammonia emissions for beef cattle, including early season grazing and piosphere effects. Animal Feed Science and Technology, 166-167, 688-698.

Swiss Center for Life Cycle Inventories (SCLCI) (2010) ecoinvent database v2.2. Available at http://www.ecoinvent.org/home/.

van Zelm, R., Schipper, A., Rombouts, M., Snepvangers, J., and Huijbregts, M. (2010). Implementing groundwater extraction in Life Cycle Impact Assessment: characterization factors based on plant species richness for the Netherlands. Enivronmental Science and Technology, 45, 629-635.

Verones, F., Hanafiah, M., Pfister, S., Huijbregts, M., Pelletier, G., and Koehler, A., 2010. Characterization factors for thermal pollution in freshwater aquatic environments. Environmental Science and Technology, 44, 9364 - 9369.

Quantification of the reduction potential of GHG mitigation measures in Swiss organic milk production using a life cycle assessment approach

Christian Schader[1,*], Katja Jud[1,2], Matthias S. Meier[1,*], Till Kuhn[1,3], Bernadette Oehen[1], Andreas Gattinger[1]

[1] FiBL - Research Institute of Organic Agriculture, Frick, Switzerland
[2] Institute for Environmental Decisions, ETH Zürich, Switzerland
[3] Institute for Food and Resource Economics, University of Bonn, Germany
[*] Corresponding author. E-mail: c*hristian.schader@fibl.org*

ABSTRACT

Quantitative assessments of the effectiveness of GHG mitigation measures at farm level are scarce. Hence, the aim of this study was to quantify the GHG mitigation potential of selected measures on two typical organic farms in Switzerland. We built a single-farm model which enabled us to calculate the GHG emissions and energy consumption at farm level using a life cycle assessment approach. The model was used to calculate the effect of 13 different mitigation measures on a Swiss organic dairy and a Swiss organic mixed farm. At the dairy and mixed farm, respectively, 5.4% and 5.5% of mitigation can be realized by technical means and 15.4% and 12.9% with agronomic measures, including conversion to full-grazing systems, composting livestock manure, and the use of dual-purpose cattle breeds, where losses in productivity may occur. Total mitigation potential of the measures analysed is 20.8% (dairy farm) and 18.4% less emissions (mixed farm), respectively.

Keywords: global warming potential, greenhouse gas mitigation, organic agriculture, carbon footprint, dairy production

1. Introduction

The environmental benefits of organic farming compared to conventional farming include a reduced toxicity potential and mitigation of greenhouse gases (GHG) (Schader et al., 2012b). However, from a LCA perspective, i.e. when relating emissions to food production, organic farming frequently performs worse than conventional or integrated management due to reduced productivity (e.g. Williams et al., 2006). Furthermore, recent studies found a high variability between farms, even if they are of the same farm type or region (Hersener et al., 2011). This implies a high potential for optimising farm management with respect to environmental performance on organic farms. Currently, different mitigation measures for reducing GHG emissions from agriculture are discussed by farmers, policy makers and researchers (BLW, 2011). However, quantitative assessments of these measures at farm level are scarce. Hence, the aim of this study was to quantify the GHG mitigation potential of selected measures for two organic farms in Switzerland.

2. Methods

2.1 Selection of typical farms

Two Swiss organic farms (Table 1) were selected based on their farm type, size, production portfolio and location, for modelling the impacts of 13 mitigation measures in a typical farm-specific context. The real farms where adapted for improving the transferability of the results with respect to the impacts of the measures.

Due to the dominant role of milk production for the Swiss organic sector two dairy farms were selected. The first farm is a typical organic farm located in the mountain areas on rather marginal land. 20 dairy cows including offspring are kept on about 25 hectares. The average milk yield (ECM) per cow and year is 5,300 kg. Organic manure, mostly slurry, is applied via pipes. The second farm is a typical organic mixed farm as prevalent in the Swiss lowlands. 40 dairy cows including offspring are kept on about 50 hectares. The average milk yield (ECM) per cow and year is 6,425 kg.

2.2 Selection and specification of GHG mitigation measures

Based on a literature review Bischhofberger and Gattinger (2011) identified 21 measures applicable on Swiss organic farms which are potentially effective in mitigating GHG. Based on this study, 13 potential GHG mitigation measures were selected for quantification on the two specific farms (Table 2). The measures were chosen according to a) their presumed mitigation potential, b) the absence of trade-offs with other environmental and ethological impact categories and c) their applicability on Swiss organic farms. The farm specific characterisations of the measures were defined according to the local conditions of the selected farms and in consent with the farmers (Table 2).

Table 1. Overview of selected farms.

Characteristics	**Specialised dairy farm**	**Mixed farm**
Region	Mountain region	Lowlands
Elevation above sea level (m)	800	500
Summer feeding period (days)	200	215
Winter feeding period (days)	165	150
Farm size (ha)	25.1	51
Share permanent grassland	100%	52%
Share arable crops	0%	48%
Crops	Permanent meadows and pastures	Permanent meadows and pastures, ley, silage maize, triticale, peas, potatoes, winter wheat
Livestock (LU[a])	24.77	52.16
Stocking density (LU[a]/ha)	0.98	0.99
Number of dairy cows	20	40
Milk production[b] per cow	5300 kg	6425 kg
Cattle housing system	Loose housing system	Loose housing system

[a] Swiss livestock units
[b] fat and protein corrected milk (FPCM)

Table 2. Farm-specific characterisations of selected GHG mitigation measures.

Measure	Specification of measure
Composting livestock manure	Due to better aeration of manure, CH_4 and N_2O emissions can be reduced. However, additional use of machinery and fleece is necessary for compost preparation.
Increased number of lactations of dairy cows	An increased number of lactations (up to 3.6) decreases the share of cattle in an unproductive stage and thus total methane emissions
Use of dual-purpose cattle breeds	Cattle race ‘Original Braunvieh” is used instead of ‘Swiss Braunvieh”. Changes in milk yield were estimated according to Mahrer (2011)
Use of photovoltaics	Photovoltaics are used on the total roof area (Dairy farm: 500 m^2, mixed farm: 900 m^2) of the farms under consideration of local solar radiation levels.
Conversion to full-grazing system	Enhance the share of pastures for allowing the cattle herd to graze outside during the summer feeding period. This leads to a reduction of manure management emissions and energy use due to grassland management
Machine use during technical life time	Machines and tractors were assumed to be used beyond the amortisation period up to their technical life time.
Shade trees on pastures	Planting of walnut trees (*Juglans regia*) on pastures (5 on dairy farm; 20 on mixed farm)
Energy-efficient milk cooling devices	Installation of waste heat recovery devices from milk cooling for warm water heating
Concentrate-free feeding rations	Concentrate use in feeding rations of dairy cows and offspring is substituted by grass-based fodder
Application of Eco drive mode	Studies show that an application of ‘eco-efficient driving” can save 10-20% fuel compared to standard driving practice
Optimisation of machines and tractors	We assumed that farmers purchase and use the more energy efficient tractor (instead of an average tractor)
Use of solar heat	Solar heat is used for heating process water
Reduced tillage	Ploughing is replaced by cultivator, except after ley (measure only on the mixed farm). Fuel can be saved and soil carbon stocks is likely to be increased but was assumed to remain constant as reliable figures were unavailable for a situation where ploughing is reduced instead of completely stopped.

2.3. Farm model

We built a single-farm model based on a LCA approach which enabled us to calculate the GHG emissions and energy use of a farm in different conditions, using ecoinvent and other data sources as well as data from own assessments (Schader et al., 2012a).

Main components of the model are the plant and the livestock production module which take into account all processes and inputs relevant for defining the production inventories (Fig. 1). Inventories are based on ecoinvent (Nemecek und Kägi, 2007) and own inventories of organic production processes. The model is able to reflect interactions between plant and livestock production including emissions of purchased inputs, in line with life cycle assessment methodology as defined in ISO14040 and 14044.

GHG emissions were calculated based on the production inventories according to the IPCC guideline (2006) and PAS2050. For calculating on-farm N_2O emissions a model was employed, which specifically takes into account the mode of action of organic fertilisers when applied to the soil (Meier et al., 2012). Enteric fermentation was calculated using a model by Kirchgessner (1995), while the IPCC model (2006) was used for sensitivity analysis.

GHG emissions are calculated for the functional units 'ha of cultivated area' and 'kg FPCM (fat- and protein-corrected) milk produced'. For calculating the milk production-related impacts the emissions directly associated with cash crops were excluded, while emissions for dairy production were allocated economically between milk, meat and live animals output. Changes in productivity due to GHG mitigation measures have to be determined exogenously. Each farm was assessed a) without implementation of the measures and b) with each of the measures implemented individually. The difference in GHG emissions between both states of a farm is interpreted as the effectiveness of the measure for mitigating GHG emissions on the specific farm.

3. Results

3.1 GHG emission profiles without implementation of GHG mitigation measures

Without any mitigation measure implemented the dairy farm (DF) emits 139 t CO_2-eq, while the mixed farm (MF) 278 t CO_2-eq annually (Fig. 2; columns). Per hectare, this means almost equal emissions of 5.54 t CO_2-eq (DF) and 5.50 t CO_2-eq (MF), respectively. Both farms have a similar stocking density. However, the MF has a higher production. Therefore, the MF is more GHG efficient (0.89 kg CO_2-eq/kg milk) compared to the DF (1.08 kg CO_2-eq/kg milk) (Fig. 2; diamonds). The main reason for this difference lies in the more favourable climatic conditions for the MF as it is located in the lowlands and benefits from a longer vegetation period and higher temperatures.

The by far greatest share of GHG emissions is associated with enteric fermentation of both dairy cows and offsprings (DF: 56.1%; MF: 55.7%) (Fig. 2). Further important factors are manure management with 18.6% (DF) and 20.7% (MF), respectively, and fodder production (DF: 18.7%; MF: 13.7%). While the DF does not purchase any fodder from outside, the mixed farm imports concentrates (4.4%) which contribute 1.9% to the total emissions.

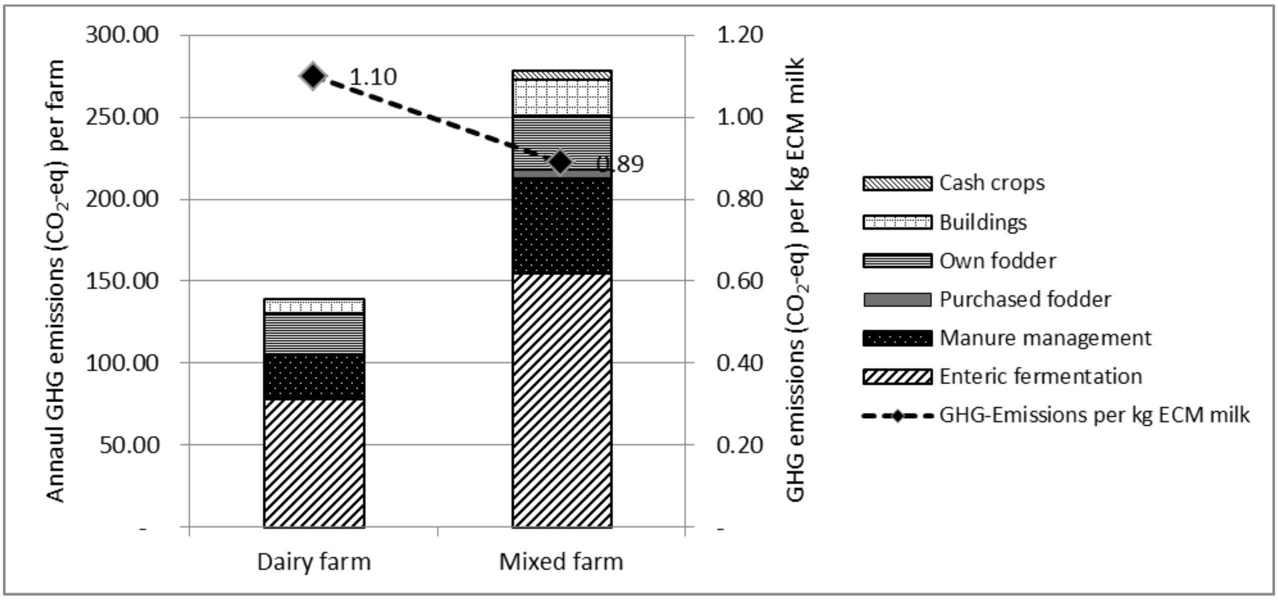


Figure 2. Annual GHG emissions (CO_2-eq) on a typical Swiss organic dairy and mixed farm

3.2 Mitigation potential of the measures

Effective measures are conversion to full-grazing systems, use of dual-purpose cattle breeds, increased number of lactations of dairy cows, composting livestock manure, use of photovoltaics, use of solar heat, and optimisation of machine life. At the dairy and mixed farm, respectively, 5.4% and 5.5% of mitigation can be realised by technical means (use of photovoltaics, optimisation of machine life, energy-efficient milk cooling devices, application of eco drive mode, optimisation of machines and tractors, use of solar heat) and 15.4% and 12.9% with agronomic measures (composting livestock manure, increased number of lactations of dairy cows, use of dual-purpose cattle breeds, conversion to full-grazing system, shade trees on pastures, concentrate-free feeding rations) where losses in productivity may occur. Total mitigation potential of the measures analysed is 20.8% (dairy farm) and 18.4% less emissions (mixed farm), respectively.

A limited mitigation potential of below 1% of total GHG emissions (Table 3) can be associated with the measures 'shade trees on pastures', 'energy-efficient milk cooling devices', 'concentrate-free feeding rations', 'application of eco drive mode', 'optimisation of machines and tractors', 'use of solar heat' and 'reduced tillage'. The relative effects of these measures are limited, as they address predominantly energy use, while CH_4 emissions from enteric fermentation dominate the emission profile of the selected farms. Furthermore, some of the measures may perform better in a different context. For instance, we assumed that under local farm conditions on the mixed farm, a full conversion to reduced tillage is not feasible due to weed control problems. However, from field and on farm-trials in other locations in Switzerland we know that if a full conversion is undertaken, soil carbon stocks build up, improving the GHG balance of production (Berner et al., 2008; Gadermaier et al., 2010).

The low performance of the measure 'concentrate-free feeding rations' depends on the fact, that organic farms in Switzerland are restricted already to use only 10% concentrates in their feeding rations. Furthermore, soy used in Swiss organic feeding rations at present does not originate from regions where direct land use change from rain forest or savannah to arable land is relevant.

Table 3. Optimisation potential of selected GHG-reduction measures on two typical Swiss organic farms (kg CO_2-eq / per farm and year)

	Specialised dairy farm		**Mixed farm (dairy/arable)**		
Potential reduction of GHG emissions	**Absolute**	**Relative**	**Absolute**	**Relative**	**Potential impact on productivity**
Total GHG emissions	139,066	100.00%	277,911	100.00%	
Composting livestock manure	-4429	-3.18%	-12,128	-4.36%	
Increased number of lactations of dairy cows	-7,788	-5.60%	-8,677	-3.12%	slightly reduced productivity expected
Use of dual-purpose cattle breeds	-3,977	-2.86%	-7,357	-2.65%	reduced milk production but increased meat production expected
Use of photovoltaics (on total roof area)	-4,073	-2.93%	-6,153	-2.21%	
Conversion to full-grazing system	-4,672	-3.36%	-6,128	-2.21%	Estimated reduction in milk production 11-19%
Optimisation of machine life	-2,206	-1.59%	-4,237	-1.52%	
Shade trees on pastures	-226	-0.16%	-753	-0.27%	< 1% reduced pasture productivity expected
Energy-efficient milk cooling devices	-235	-0.17%	-518	-0.19%	
Concentrate-free feeding rations	-371	-0.27%	-343	-0.12%	0-10% reduced productivity expected (Notz *et al.*, 2012)
Application of Eco drive mode	-728	-0.52%	-2,206	-0.79%	
Optimisation of machines and tractors	-111	-0.08%	-1,935	-0.70%	
Use of solar heat (for process water on farm)	-139	-0.10%	-262	-0.09%	
Reduced tillage*	-		-564	-0.20%	+/- 10% productivity changes expected (Berner *et al.*, 2008)
Potential GHG savings if all measures are implemented	-28,955	-20.82%	-51,261	-18.45%	

* calculations do not take into account potential gains in carbon stocks

4. Discussion and conclusions

Our model results demonstrate that the 13 measures for GHG mitigation have a cumulative potential of about 20.8% (dairy farm) and 18.4% (mixed farm), respectively. However, the effectiveness of the optimisation measures depends on farm-specific characteristics. For instance, the effectiveness installing a photovoltaic plant may be limited if the roof exposition is sub-optimal. Furthermore, if the mitigation potential is calculated for the functional unit 'milk production', implementing measures that reduce productivity could decrease the effectiveness of the measures or even lead to negative impacts on the global warming potential. Possible productivity losses depend, however, on farm-specific characteristics and are difficult to anticipate.

It is important to understand that the mitigation potential must not be interpreted as optimisation potential. The comparison of the two farms – one on favourable and one on marginal land – illustrates that geographical conditions can influence the GHG efficiency of production substantially. As we have shown, the mitigation potential of these two farms is limited, even if all measures are applied. Especially on organic farms, the input-side is in most cases already optimised to a large extent. Optimisation potential on organic farms with respect to GHG efficiency lies largely in improving farm management in order to increase productivity.

Finally, it is important to note, that we only analysed the impacts of the measures on GHG emissions. We regard the occurrence of trade-offs to other environmental impact categories with implementing the measures as unlikely. Therefore, the implementation of the analysed optimisation measures can be recommended from an ecological perspective, in particular if substantial productivity losses can be avoided on farm. In fact, there is a high potential of co-benefits in terms of other environmental and ethological impacts associated with some measures. For instance, composting of livestock manure is known to have a positive influence on soil fertility. Therefore, when ranking the measures against each other, these co-benefits should be considered. The farm model proved to be a useful tool for assessing farm- and product-specific GHG emissions and modelling the potential of measures for mitigating greenhouse gases.

5. Acknowledgements

The authors thank the many experts that contributed to the project underlying this paper, particularly Anet Spengler, Silvia Ivemeyer, Peter Klocke, Barbara Früh, Christian Winkler, Hansueli Dierauer, Paul Mäder, Christophe Notz, Eric Meili, Pamela Staehli, and Christoph Fankhauser.

6. References

Berner, A., Hildermann, I., Fliessbach, A., Pfiffner, L., Niggli, U. and Mäder, P. (2008). 'Crop yield and soil fertility response to reduced tillage under organic management'. Soil & Tillage Research 101, pp. 89-96.

Bischofberger, N. and Gattinger, A. (2011). 'Klimaschutz auf Biobetrieben - Merkblatt', Forschungsinstitut für biologischen Landbau (FiBL), Frick, Schweiz.

BLW (2011). 'Klimastrategie Landwirtschaft. Klimaschutz und Anpassung an den Klimawandel für eine nachhaltige Schweizer Land- und Ernährungswirtschaft', Bern, Bundesamt für Landwirtschaft (BLW).

Gadermaier, F., Berner, A., Fließbach, A., Friedel, J.K. and Mäder, P. (2010). 'Impact of reduced tillage on soil organic carbon and nutrient budgets under organic farming'. Renewable Agriculture and Food Systems 27: 1, pp. 68-80.

Hersener, J.-L., Baumgartner, D. and Dux, D. (2011). 'Zentrale Auswertung von Ökobilanzen landwirtschaftlicher Betriebe (ZA-ÖB). Schlussbericht', Agroscope Reckenholz-Tänikon (ART).

IPCC (2006). '2006 IPCC Guidelines for National Greenhouse Gas Inventories - Volume 4: Agriculture, Forestry and Other Land Use', Intergovernmental Panel on Climate Change (IPCC).

Kirchgessner, M. (1995). Tierernährung, Frankfurt/Main: DLG-Verlag.

Mahrer, D.D. (2011). 'Unterschiede zwischen F1 und F2 Kreuzungstieren von Original Braunvieh (OB) x Braunvieh (BV) und Tieren der Elternpopulation'. Bachelor Thesis. Berner Fachhochschule (BFH), Schweizer Hochschule für Landwirtschaft (SHL). Zollikofen.

Meier, M., Schader, C. and Gattinger, A. (2012). 'Modeling N2O emissions from organic fertilisers for LCA inventories', 8th International Conference on LCA in the Agri-Food Sector (LCAfood2012) 2-4 October 2012, Rennes, France.

Nemecek, T. and Kägi, T. (2007). 'Life Cycle Inventories of Agricultural Production Systems. Data v2.0. Ecoinvent report No. 15', Zürich and Dübendorf, ecoinvent centre. Swiss Centre for Life Cycle Inventories. Agroscope Reckenholz-Tänikon Research Station (ART).

Notz, C., Böhler, D., Hecht, J., Ivemeyer, S., Maeschli, A., Meier, M., Meili, E., Klocke, P., Kuhn, T., Schader, C., Schumann, T., Staehli, P., Walkenhorst, M. and Winkler, C. (2012). 'Feed no Food – Einfluss der kraftfutterminimierten Rinderfütterung auf Tiergesundheit, Ökologie und Ökonomie - Projektabschlussbericht 2009-2011', Frick, Research Institute of Organic Agriculture (FiBL).

Schader, C., Meier, M., Kuhn, T., Jud, K., Oehen, B., Gattinger, A. and Stolze, M. (2012a). 'Modell zur Quantifizierung des Energieberbrauchs und der Treibhausgasemissionen von Biobetrieben. Diskussionspapier. Version 30.3.2012', Frick, Research Institute of Organic Agriculture (FiBL).

Schader, C., Stolze, M. and Gattinger, A. (2012b). 'Environmental performance of organic agriculture'. in Boye, J. and Arcand, Y. (eds.), Green Technologies in Food Production and Processing, New York: Springer, pp. 183-206.

Williams, A.G., Audsley, E. and Sandars, D.L. (2006). 'Energy and environmental burdens of organic and non-organic agriculture and horticulture'. Aspects of Applied Biology. What will organic farming deliver? 79: COR 2006, pp. 19-23.

Produce beef or biodiversity? The trade-offs between intensive and extensive beef fattening

Thomas Nemecek*, Frank Hayer, Martina Alig, Philippe Jeanneret, Gérard Gaillard

Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zürich, Switzerland,
* Corresponding author. E-mail: *thomas.nemecek@art.admin.ch*

ABSTRACT

According to the concept of multi-functionality, agriculture has to fulfil several functions. In addition to the production of food, feed, fibre and fuels, it aims at maintaining a high level of biodiversity, maintaining agricultural production in marginal areas, and ensuring living for a rural population. The productive function requires highly productive systems, whereas to increase biodiversity, extensive management is needed. This trade-off between high productivity and high biodiversity potential is highlighted in this paper, using a case study with different beef production systems in Switzerland.

The SALCA-biodiversity method (Jeanneret et al., 2008) is used to assess potential impacts of agricultural management on the overall species diversity. The method uses 11 indicator organisms groups to describe the potential impacts on biodiversity. Three different beef production systems were evaluated using model farms: an integrated beef fattening (IBF), an integrated suckler cow system (ISC), and an organic suckler cow system (OSC), all located in the lowlands. First an inventory of all used agricultural areas (for feedstuff production, grazing, rearing animals, etc.) was established. Then, an inventory of all management activities influencing biodiversity was established and the potential impacts on the indicator organism groups were assessed. The average biodiversity scores were calculated as area and time weighted averages of all used agricultural areas. Within a given crop the differences were relatively small, whereas the differences due to different management intensities of grassland were very large. The final result is thus determined by the composition of the feed ration, the yields of the different feed raw materials, which determine the area required for their production, and the management intensity and type of use (grazing, cutting) for grassland.

The integrated beef fattening system uses two to three times less area to produce 1 kg of beef compared to the suckler cow systems. This is due to the fact that the suckler cow has to be fully allocated to the beef production, whereas in an intensive beef fattening system, the calf can be considered as a by-product of dairy production and the mother cow is allocated mainly to the milk. Furthermore, the suckler cow systems use partly extensively managed grassland with lower yields. The OSC needs slightly more area to produce the same amount of beef than ISC, due to lower yields of organic crops and grasslands.

The overall species diversity was estimated to be lowest for the IBF and higher for the suckler cow systems (Fig. 1). This difference is explained by the fact that overall species diversity is generally higher in grassland than in crops. In particular less intensively and extensively managed grassland had higher scores for overall species diversity. Due to the ban of pesticides, the OSC had a slightly higher biodiversity score than ISC.

Now we have to consider that the areas used are very different: the integrated beef system uses less area; this leaves area for other uses. If we assume that the excess area is used for ecological compensation areas, the integrated system would have the highest overall biodiversity score. If however, the excess area is managed in the same way as the rest of the system, the extensive systems would have the highest scores, but the overall production would not be the same.

Two scenarios have been calculated to estimate the theoretical potentials for biodiversity and productivity of beef fattening: one scenario assesses the maximal biodiversity potentials, which could be achieved by extensive management (Fig. 1). The difference to the studied systems is considerable. This shows that the current beef production systems have a high potential for development both in terms of higher productivity or higher biodiversity. These two goals are in conflict. Research is needed to develop beef fattening systems which better reconcile a good productivity with a high level of biodiversity (the green arrow "sustainable development").

Acknowledgement

These research results were developed by ART with the support of COOP Switzerland.

Reference

Jeanneret, P., Baumgartner, D.U., Knuchel, R.F. & Gaillard, G., 2008. A new LCIA method for assessing impacts of agricultural activities on biodiversity (SALCA-Biodiversity). In: Proceedings of the 6th International Conference on LCA in the Agri-Food Sector – Towards a sustainable management of the Food chain, Nemecek, T. & Gaillard, G. (eds.). November 12–14, 2008, Zurich. Agroscope ART, 34-39.

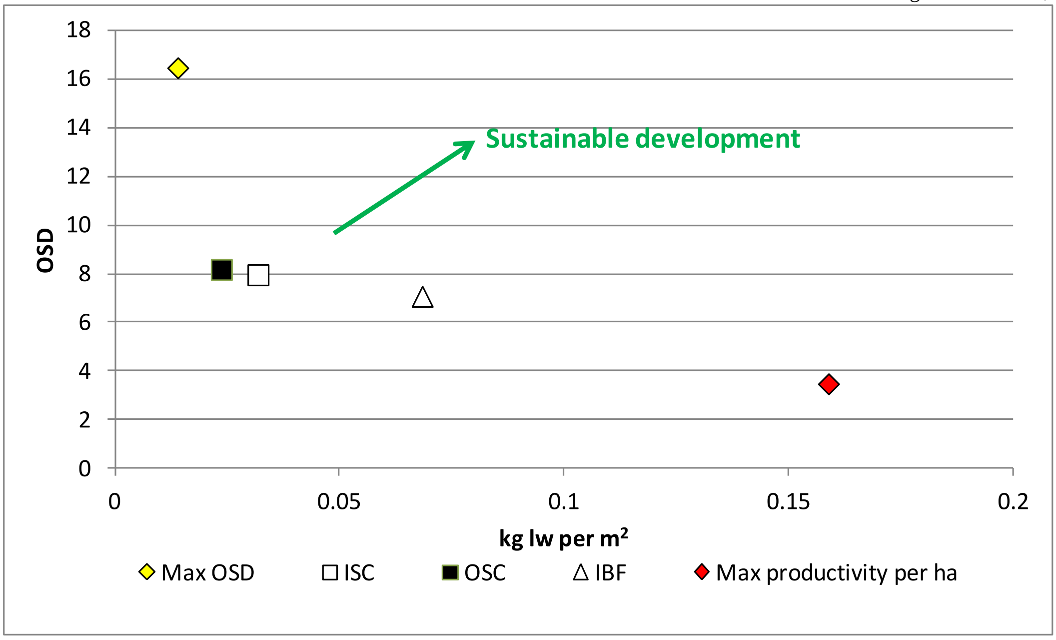


Figure 1. Comparison of overall species diversity (OSD) of three beef fattening systems: integrated beef fattening (IBF), integrated suckler cow system (ISC), organic suckler cow system (OSC). "Max OSD" shows the theoretical potential for highest biodiversity in a beef production system, "Max productivity per ha" shows the theoretical potential for high productivity.

Assessing land use impacts on biodiversity on a regional scale: the case of crop production in Kenya

Laura de Baan[1*], Michael Curran[2], Stefanie Hellweg[2], Thomas Koellner[3]

[1] NSSI, Institute for Environmental Decisions, ETH Zurich, Universitätstr. 22, 8092 Zurich, Switzerland
[2] ESD, Institute for Environmental Engineering, ETH Zurich, Schafmattstr. 6, 8093 Zurich, Switzerland
[3] Professorship of Ecological Services, University of Bayreuth, Universitätsstr. 30, 95440 Bayreuth, Germany
Corresponding author. E-mail: *laura.debaan@ethz.env.ch*

ABSTRACT

In current Life Cycle Impact Assessment (LCIA), impacts of land use on biodiversity are often assessed at a local (e.g. field) scale. The relevance of these local impacts is often limited, as species that are not able to survive on agricultural land may persist in adjacent undisturbed areas. However, if habitat is scarce on a regional scale, the risk of regional or even global extinction increases. Here, we present a method to upscale local land use impacts in LCIA to the ecoregional scale using an adapted species-area relationship model. This model is applied to land use in Kenya, where conversion of natural habitats to small and large-scale farms is threatening areas of high biodiversity value. As we use globally available data as a starting point, this method is potentially applicable to other world regions and provides a more environmentally relevant assessment of the land use impacts of agricultural products.

Keywords: biodiversity, land use, regional assessment, Kenya, spatial scales

1. Introduction

Agricultural production occupies about one third of the global terrestrial surface. This causes major impacts on ecosystems and biodiversity at multiple spatial scales (e.g. local, regional and global). To evaluate options for reducing these impacts, decision-support tools such as life cycle assessment (LCA) should help decision-makers to obtain the relevant environmental information related to their product or product system. In many life cycle impact assessment (LCIA) methods for land use, *local* impacts on biodiversity are assessed (e.g. de Baan et al., 2012), which reflect the direct impacts on the used piece of land (e.g. by removing forest-species to cultivate crops). These local impacts are traditionally assessed as potentially disappeared fraction of species (PDF), representing *relative* changes in species richness between a used piece of land and a reference situation (e.g. undisturbed natural vegetation). However, these local assessment methods do not provide the information relevant for decisions-makers concerned about conservation of species at larger scales, i.e. the *absolute* loss of species in a region or globally. Local extirpation of species from a used piece of land can be fully reversible if the species can survive in adjacent wild areas, and thus recolonize the field after its abandonment. As local conversion of habitat continues, extinction risk increases to an extent that there is not enough habitat remaining at a regional or global level, leading to regional or global extinction, respectively. Global extinction is fully *irreversible* and thus of highest environmental concern. An assessment that incorporates both local and regional factors requires additional information on the landscape surrounding the agricultural field, and is thus more data demanding.

In this paper, we present a method to upscale impacts from local to regional scale and to transform relative into absolute species losses. We apply the method in a preliminary case study of agricultural production and mammal species diversity in Kenya, using globally available data.

1.1. The case: agricultural production in Kenya

Previous land use LCIA methods have been mostly developed for the European (e.g. De Schryver et al., 2010; Koellner and Scholz 2008) or North American context (e.g. Geyer et al., 2010), and to our knowledge, only one study has focused on the African continent (Burke et al., 2008). We chose Kenya as a case study to assess regional impacts on biodiversity because it harbours biodiverse areas at threat from human encroachment. Kenya hosts two global biodiversity hotspots that have already lost more than 70% of their natural habitats due to human activities. This largely results from an overlap in agricultural suitability and biological value, which are both concentrated in South-Central and Western Kenya, and have attracted a dense and growing population causing continued land use change or intensification and encroachment of protected areas. Large shares of species rich ecosystems have been converted to small-holder and large-scale agricultural land to produce subsistence or export-orientated cash crops such as tea, coffee, cashews, and flowers. A strong dependence of the often poor rural populations on ecosystems for food, fuel and grazing land, coupled with increasing population densities puts further pressures on ecosystems (Biggs et al., 2004). A more recent threat to biodiversity comes from an increase in land excisions and sales to foreign investors for the purpose of large-scale, export-oriented farming for food and biofuels production (Odhiambo 2011).

2. Methods

We calculated land use impacts on biodiversity according the UNEP/SETAC Life Cycle Initiative Framework (Koellner et al., 2012a; Milà i Canals et al., 2007), and distinguished between occupation (land use), transformation (land use change) and permanent impacts (irreversible changes in ecosystems, see Fig 1). Occupation and transformation impacts were calculated based on regional extinction risk of non-endemic species. Permanent impacts were based on the extinction risk of species endemic to that ecoregion. Below, we outline how the number of potentially extinct species was estimated and how we derived regional characterisation factors for the three land use impacts (occupation, transformation, and permanent impacts).

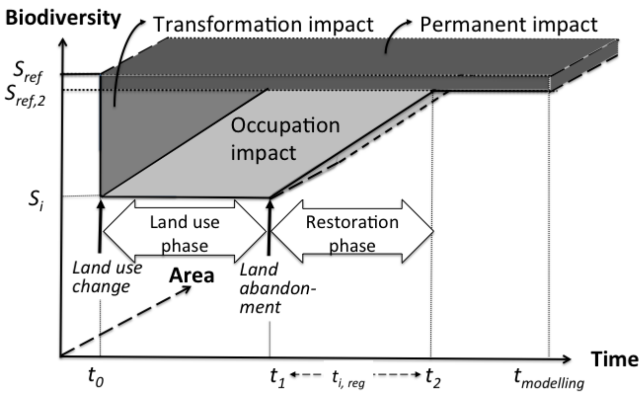


Figure 1. Graphical illustration of occupation, transformation and permanent impacts (adapted from Koellner et al., 2012a)

2.1 Calculating potential regional mammal species extinctions due to land transformation and occupation

As in earlier land use LCIA studies, we based our assessment on changes in the predicted species numbers according to the species-area relationship (SAR), a model often used to predict extinction rates due to habitat loss. The SAR is derived from island biogeography theory, which describes a non-linear (often power) function between species richness S and area A of island ecosystems (Rosenzweig 1995):

$$S = cA^z \qquad \text{Eq. 1}$$

with two constants c (species richness of area A=1) and z (species accumulation rate). If the area of primary habitat is reduced from an original area A_{org} to an area A_{new}, and if we assume $c_{new} = c_{org}$, then the relative number of species S remaining can be calculated as (Koh and Ghazoul 2010):

$$\frac{S_{new}}{S_{org}} = \left(\frac{A_{new}}{A_{org}}\right)^z \qquad \text{Eq. 2}$$

The number of species lost in an ecosystem S_{lost} due to habitat loss is given as (Koh and Ghazoul 2010):

$$S_{lost} = S_{org} - S_{org}\left(\frac{A_{new}}{A_{org}}\right)^z \qquad \text{Eq. 3}$$

As SAR models tend to overestimate extinction rates, we base our method on an adapted SAR model, the matrix-SAR developed by Koh and Ghazoul (2010), which gave more realistic estimates of species extinction rates than the conventional SAR model. Unlike the conventional SAR, the matrix-SAR model considers that the human-modified area (i.e. the land use matrix) is not void of species, but provides a certain habitat quality for different species groups. This is accounted for by adapting *z*, the exponent of Eq. 3, depending on the composition of the matrix (i.e. the area share of different land use types p_i) and the sensitivity σ_i of each taxonomic group to different land use types *i* (Koh and Ghazoul 2010):

$$z' = z^* \sum_i^n p_i \sigma_i \text{; with} \qquad \text{Eq. 4}$$

$$p_i = \frac{A_i}{A_{org} - A_{new}} \qquad \text{Eq. 5}$$

The taxon sensitivity s_i is calculated as the relative decrease in species richness on a human modified land compared to an undisturbed reference *ref*. Finally, the predicted number of species that might get regionally extinct can be calculated as (Koh and Ghazoul 2010).

$$S_{lost} = S_{org} - S_{org}\left(\frac{A_{new}}{A_{org}}\right)^{z\sum_i^n p_i s_i} \qquad \text{Eq. 6}$$

From the regionally extinct species, some species might not occur in any other world region (i.e. species endemic to one ecoregion), and thus get globally extinct. This is considered a global land use impact and is fully irreversible. We thus split the calculations into non-endemic and endemic species. The first are used to calculate occupation and transformation impacts (which are considered to be reversible) and the latter are used to calculate permanent impacts, which are considered to be irreversible. The total number of species regionally extinct is equal to the number of endemic and non-endemic species lost.

We calculated the potential regionally or globally extinct mammal species using data from WWF ecoregions (Olson et al., 2001) as spatial units. The area shares of different land use types were derived from the Globcover land cover map v2.3 (ESA 2009). Values for *z* were derived from Drakare et al., (2006) and for the taxon sensitivity σ we used data from de Baan et al., (2012).

2.2 Calculation of characterisation factors

Above, we used the matrix SAR to estimate the number of endemic and non-endemic species lost due to all accumulated land use activities within one ecoregion. For application in LCA, the impact per land use type and area is required. Therefore, the total regional impact had to be allocated to the different land use activities occurring within the region, to get an estimate of the number of species lost due to a specific land use type. In a first step, we divided the number of lost endemic and non-endemic species by the used area. Then, we allocated the impacts per area to the different land use activities, whereby the land use activity with the lowest habitat value and the largest area share got the highest share of the impacts.

Finally, we calculated characterisation factors for land occupation and transformation based on impacts on non-endemic species (considered as reversible impacts, see Fig 1). Impacts on endemic species were used to calculate permanent (irreversible) impacts.

The regional characterisation factor for ***occupation impacts*** were calculated by dividing the number of lost non-endemic species by the total used area and multiplying it with the allocation factor.

Transformation impacts were then calculated by multiplying the CF of occupation with half the time required for species to recolonize an area after the land was abandonment. As empirical data on regeneration times are missing, we assumed a constant regeneration time of 150 years for artificial area and 100 years for agricultural land.

Permanent impacts were calculated as the number of endemic species lost due to land use. These species are not occurring in any other global ecosystem and are thus permanently extinct, if they cannot persist in the ecoregion. In order for the three impacts (transformation, occupation and permanent) to be aggregated in a final step, they must be in the same units, i.e. [species loss* area * time], see Fig 1. Thus, we assessed the permanent impacts over a certain modelling period. Here we used a 500 years modelling period as suggested by Koellner et al., (2012a).

For calculating occupation impacts of agricultural products, the characterisation factor for occupation is multiplied by the land use inventory flow of land occupation, given as [area * time]. The transformation and permanent impacts are multiplied by the inventory flow of land transformation, given as [area].

3. Results

We calculated the habitat loss in each Kenyan ecoregion due to the total current land occupation and past transformation and modelled the potential loss of endemic and non-endemic mammals per ecoregion. The degree of habitat conversion varied substantially across ecoregions, with very little use in desert ecosystems and up to 85% converted land in agriculturally suitable areas (Fig 2). The total mammals species richness varied between ecoregion from 27 to 221 with zero to 6 endemic mammals per ecoregion. For nonendemic mammals, zero to 14 species were predicted to be regionally extinct due to all land use activities (Fig 3). For endemic species, zero to 0.4 species per ecoregion were predicted at risk of global extinction due to past habitat conversion (Fig 4).

A selection of the resulting regional characterisation factors for occupation, transformation and permanent impacts is displayed in Table 1. Compared to local (relative) characterisation factors calculated by de Baan et al., (2012), the regional (absolute) characterisation factors were several orders of magnitude smaller. The regional occupation and transformation impacts ranged between ecoregions by a factor 10 to over 180, whereas the regional permanent impacts ranged from zero (ecoregions with no endemic mammals) to $2*10^{-08}$ with a factor 80 from lowest to highest impact.

Table 1. Local and regional characterisation factors calculated for selected regions and land use types. The respective units are indicated in brackets. The land use types follow the classification suggested in Kollner et al., (2012b).

	Biome	5 Agriculture	5.1 Annual crops	5.2. Permanent crops	6. Agriculture, mosaic 50-70%	6. Agriculture, mosaic 20-50%	7. Artificial areas
Local occupation impacts [potentially disappeared *fraction* of species (PDF)] (from de Baan et al., 2012)							
World average		**0.56**	**0.6**	**0.42**	**0.2**	**0.2**	**0.44**
(Sub)-tropical moist broadleaf forest	1	0.51	0.54	0.42	0.18	0.18	-
(Sub)-tropical grassland, savannas, and shrublands	7	0.65	0.65	-	-	-	-
Regional occupation impacts [potentially regionally extinct non-endemic species]							
East African montane forests (AT0108)	1	5.5E-10	5.9E-10	4.6E-10	2.0E-10	2.0E-10	4.8E-10
Eastern Arc forests (AT0109)	1	1.1E-09	-	8.8E-10	-	3.8E-10	9.2E-10
Northern Zanzibar-Inhambane coastal forest mosaic (AT0125)	1	2.8E-10	3.0E-10	2.3E-10	9.8E-11	9.8E-11	2.4E-10
Northern Acacia-Commiphora bushlands and thickets (AT0711)	7	1.3E-10	1.3E-10	8.7E-11	4.1E-11	4.1E-11	9.1E-11
Somali Acacia-Commiphora bushlands and thickets (AT0715)	7	3.0E-11	3.0E-11	-	9.3E-12	9.3E-12	2.1E-11
Southern Acacia-Commiphora bushlands and thickets (AT0716)	7	1.9E-10	1.9E-10	-	5.8E-11	5.8E-11	1.3E-10
Victoria Basin forest-savanna mosaic (AT0721)	7	3.8E-10	3.8E-10	2.5E-10	1.2E-10	1.2E-10	2.6E-10
Regional transformation impacts [potentially regionally extinct non-endemic species * years]							
East African montane forests (AT0108)	1	2.8E-08	2.9E-08	2.3E-08	9.8E-09	9.8E-09	3.6E-08
Eastern Arc forests (AT0109)	1	5.3E-08	-	4.4E-08	-	1.9E-08	6.9E-08
Northern Zanzibar-Inhambane coastal forest mosaic (AT0125)	1	1.4E-08	1.5E-08	1.1E-08	4.9E-09	4.9E-09	1.8E-08
Northern Acacia-Commiphora bushlands and thickets (AT0711)	7	6.7E-09	6.7E-09	4.3E-09	2.1E-09	2.1E-09	6.8E-09
Somali Acacia-Commiphora bushlands and thickets (AT0715)	7	1.5E-09	1.5E-09	-	4.7E-10	4.7E-10	1.5E-09
Southern Acacia-Commiphora bushlands and thickets (AT0716)	7	9.3E-09	9.3E-09	-	2.9E-09	2.9E-09	9.5E-09
Victoria Basin forest-savanna mosaic (AT0721)	7	1.9E-08	1.9E-08	1.2E-08	5.8E-09	5.8E-09	1.9E-08
Regional permanent impacts [potentially globally extinct endemic species * years]							
East African montane forests (AT0108)	1	4.5E-09	4.7E-09	3.7E-09	1.6E-09	1.6E-09	3.8E-09
Eastern Arc forests (AT0109)	1	1.7E-08	-	1.4E-08	-	5.9E-09	1.4E-08
Northern Zanzibar-Inhambane coastal forest mosaic (AT0125)	1	4.1E-09	4.4E-09	3.4E-09	1.5E-09	1.5E-09	3.6E-09
Northern Acacia-Commiphora bushlands and thickets (AT0711)	7	3.2E-10	3.2E-10	2.1E-10	9.8E-11	9.8E-11	2.2E-10
Somali Acacia-Commiphora bushlands and thickets (AT0715)	7	5.7E-10	5.7E-10	-	1.8E-10	1.8E-10	3.9E-10
Southern Acacia-Commiphora bushlands and thickets (AT0716)	7	0.0E+00	0.0E+00	-	0.0E+00	0.0E+00	0.0E+00
Victoria Basin forest-savanna mosaic (AT0721)	7	2.6E-09	2.6E-09	1.7E-09	8.0E-10	8.0E-10	1.8E-09

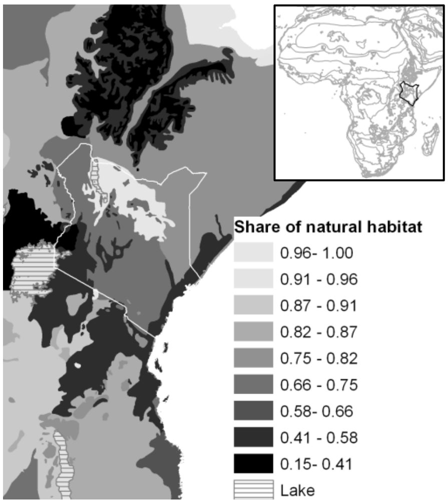


Figure 2. Remaining share of natural habitat in and around Kenya (country marked as white line)

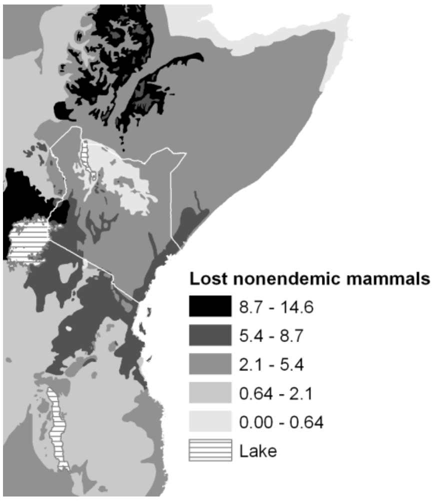


Figure 3. Modelled loss of non-endemic mammals

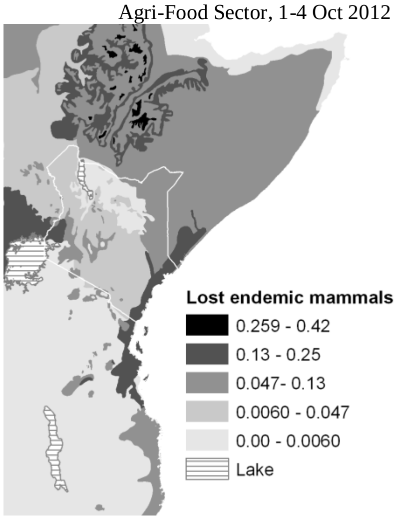


Figure 4. Modelled loss of endemic mammals

4. Discussion

In this paper we assessed *absolute regional* land use impacts (i.e. numbers of species that potentially disappear regionally due to land use) by upscaling *local relative* impacts (i.e. the potential *fraction* of locally disappeared species). Absolute and regional characterisation factors are better suited to compare land use impacts across ecosystems and across land use types. In contrast to local relative impacts, the presented regional absolute characterisation factors reflect differences of land use impacts occurring in species rich or species poor ecosystems with high or low levels of endemism, or of land use impacts occurring in highly disturbed or pristine ecosystems. In addition, they directly reflect conservation targets that aim to prevent regional (occupation and transformation impacts) and global extinction (permanent impacts).

The presented approach used globally available data to allow transferring the approach to other world regions. However, the accuracy of such global data in the regional context still has to be further assessed. Better regional land use maps, more region- and taxon-specific habitat suitability scores and better data on restoration times would improve the credibility of the results. Modelling species extinction at regional and global scales for use in LCA improves the interpretability of results, but also introduces many uncertainties. The validity of the matrix-SAR model still has to be further tested, as many important factors driving extinction risk were not considered here (e.g. heterogeneity within ecoregions, temporal dynamics, connectivity of habitats).

5. Conclusion

Upscaling local land use impacts to regional impacts provides more environmentally relevant information to policy makers on the effects of land use on biodiversity. The presented method can potentially help to better evaluate the environmental impacts of globally traded agricultural goods originating from Kenya or other world regions. As we use globally available data as a starting point, this method can be applied to other world regions. However, the robustness and validity of the presented characterisation factors still needs to be evaluated and tested against more detailed data and techniques in a variety of settings.

6. References

Biggs, R., Bohensky, E., Desanker, P. V., et al., , 2004. Nature Supporting People: The Southern African Millennium Ecosystem Assessment. Integrated report. Pretoria, South Africa: Council for Scientific and Industrial Research.

Burke, A., Kyläkorpi, L., Rydgren, B., Schneeweiss, R., 2008. Testing a scandinavian biodiversity assessment tool in an African desert environment. Environ. Manage. 42: 698–706.

de Baan, L., Alkemade, R., and Koellner, T., 2012. Land use impacts on biodiversity in LCA: a global approach. Int. J. Life Cycle Assess. (online first).

De Schryver, A. M., Goedkoop, M. J., Leuven, R. S. E. W., and 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.

Drakare, S., Lennon, J., and Hillebrand, H., 2006. The imprint of the geographical, evolutionary and ecological context on species-area relationships. Ecol Lett 9: 215-227.

ESA, 2009. GlobCover land cover map, v2.3. European Space Agency.

Geyer, R., Lindner, J.P., Stoms, D.M., Davis, F.W., and Wittstock, B., 2010. Coupling GIS and LCA for biodiversity assessments of land use: Part 2: Impact assessment. Int J Life Cycle Assess 15: 692-703.

Koellner, T. and Scholz, R.W., 2008. Assessment of land use impacts on the natural environment. Part 2: Generic characterization factors for local species diversity in Central Europe. Int. J. Life Cycle Assess. 13: 32-48.

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., Milà i Canals, L., Saad, R., de Souza, D. M., and Müller-Wenk, R., 2012a. UNEP-SETAC Guideline on Global Land Use Impact Assessment on Biodiversity and Ecosystem Services in LCA. Int. J. Life Cycle Assess. (accepted).

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Goedkoop, M. J., Margni, M., Milà i Canals, L., Müller-Wenk, R., Weidema, B., and Wittstock, B., 2012b. Principles for Life Cycle Inventories of land use on a global scale. Int. J. Life Cycle Assess. (online first).

Koh, L.P. and Ghazoul, J., 2010. A Matrix-Calibrated Species-Area Model for Predicting Biodiversity Losses Due to Land-Use Change. Conserv. Biol. 24: 994–1001.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., and Rydgren, B., 2007. Key Elements in a Framework for Land Use Impact Assessment Within LCA. Int. J. Life Cycle Assess. 12: 5-15.

Odhiambo, M.O. 2011. Commercial pressures on land in Africa: A regional overview of opportunities, challenges and impacts. Rome: The International Land Coalition.

Olson, D., Dinerstein, E., Wikramanayake, E., et al., 2001. Terrestrial ecoregions of the worlds: A new map of life on Earth. BioScience 51: 933-938.

Rosenzweig, M.L., 1995. Species diversity in space and time. Cambridge University Press.

Modelling effects of river water withdrawals on aquatic biodiversity in LCIA

Danielle Tendall[1,2,*], Stephan Pfister[2], Stefanie Hellweg[2], Gérard Gaillard[1]

[1] Agroscope Reckenholz-Tänikon research station ART, 8046 Zürich, Switzerland
[2] Ecological Systems Design, ETHZ, 8093 Zürich, Switzerland
[*] Corresponding author. E-mail: *danielle.tendall@art.admin.ch*

ABSTRACT

Water abstraction for crop production is expected to increase during the next century in Switzerland (Fuhrer et al., 2009) and certain regions of Europe (EEA 2009) due to climate change. Rivers are a significant source of water for irrigation, and therefore impacts of river water withdrawals should be included in LCIA of crop production. The impact of river water withdrawals on freshwater biodiversity has been modelled in LCA (Hanafiah et al., 2011) by using a relationship between fish species richness aggregated within river basins to average discharge at the mouth of the basins, based on statistical regression. However the parameterisation used, developed for latitudes below 42° and near-natural rivers, is not applicable to Switzerland and much of Europe. Furthermore, local effects are not addressed.

We therefore developed species-discharge parameterisations for Switzerland and Europe, including macro-invertebrate taxa in addition to fish for Switzerland, so as to reflect more ecosystem functionality. We show that the goodness of fit can be highly improved for certain regions if a higher spatial resolution is used (for example subsets based on eco-region): such spatial subsets for Europe revealed Pearson's R^2 of up to 0.59 compared to 0.35 for the whole dataset (Fig. 1), and for Switzerland R^2 was improved from 0.69 to 0.9. We assessed the sensitivity of results between such "improved-fits" and the initial generic model: predictions of PDF are similar, although the parameterisations with higher spatial resolution predict species richness more accurately (the slopes of the parameterisations show similar trends, whereas absolute values such as maxima differ). The parameterisations for Switzerland show that the potential fraction of species lost (PDF) due to marginal withdrawals of river water is higher for smaller rivers and suggest threshold behaviour between small and large rivers. Uncertainty in the species-discharge relationship was observed to be high for discharges below $20 m^3/s$ for Switzerland, in which range other sources of variability should be explored.

Generalised relationships between discharge and aquatic species richness are not confirmed by experimental studies, as effects are highly site-specific (Poff and Zimmerman 2010). Additionally, the major hypothesis of the reference model is debated, namely: inferring causality between changes in discharge and aggregate species richness in the watershed ("aggregated-level"). Using site species richness ("site-level") rather than aggregated-level richness in the regression may be an improvement, as we show with the example of a small watershed. In keeping with the site-specific nature of such relationships (as mentioned above), correlations tested for larger spatial scales were found to be weak, illustrating their limited use for larger spatial scales and generic application.

We conclude that our aggregated-level parameterisations can be used to calculate characterisation factors specifically for Switzerland and for eco-regions of Europe, however with high uncertainties for small rivers. For adequate species richness prediction, the use of such spatially-differentiated parameterisations is recommended, whereas the generic parameterisation is sufficient for estimates of PDF. Site-level models may improve ecological meaningfulness in localised assessments, but additional drivers should be included to improve model strength and reduce uncertainties.

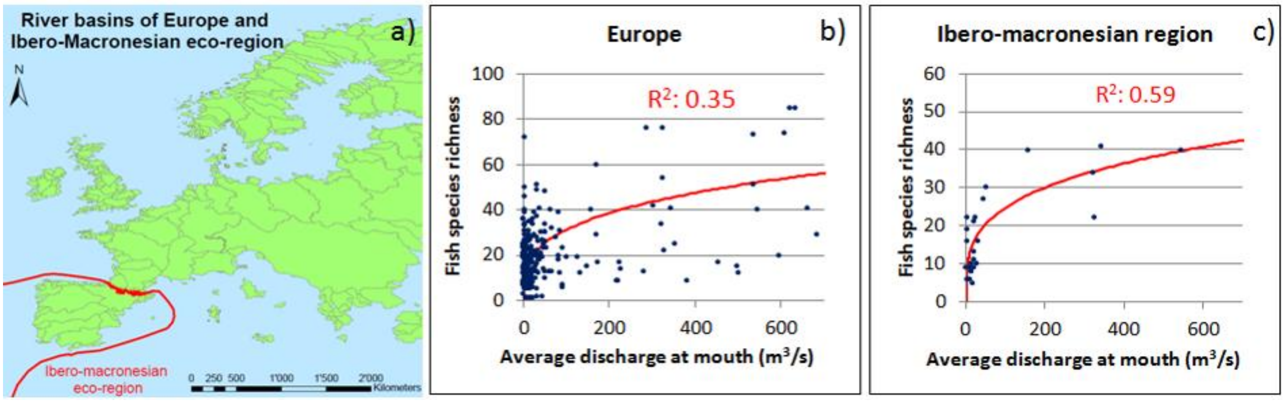


Figure 1. Improvement of correlation strength using regionalised species-discharge relationships: an example for Europe: a) map of river basins of Europe, with ibero-macronesian eco-region highlighted in red, b) species-discharge relationship for Europe, c) species-discharge relationship for ibero-macronesian region.

References

European Environment Agency (EEA), 2009. Water resources across Europe - confronting water scarcity and drought. Copenhagen, 55 p.

Fuhrer, J., Jasper, K., 2009. Bewässerungsbedürftigkeit in der Schweiz. Agroscope Reckenholz-Tänikon, Swiss Federal Office for Agriculture, 74 p.

Hanafiah, M.M., Xenopoulos, M.A., Pfister, S., Leuven, R.S.E.W., Huijbregts, M.A.J., 2011. Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. Environ. Sci. Technol. 45, 5272-5278.

Poff, N.L., Zimmerman, J.K.H., 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 55, 194-205.

Soil-quality indicators in LCA: method presentation with a case study

Emmanuelle Garrigues[1,2,*], Michael S. Corson[1,2], Christian Walter[2,1], Denis A. Angers[3], Hayo van der Werf[1,2]

[1] INRA, UMR1069, Soil Agro and hydroSystems, F-35000 Rennes, France
[2] Agrocampus Ouest, UMR1069, Soil Agro and hydroSystems, F-35000 Rennes, France
[3] Agriculture and Agri-Food Canada, 2560 Blvd. Hochelaga, Quebec City, Quebec, G1V 2J3, Canada
[*] Corresponding author: *emmanuellegarrigues@hotmail.com*

ABSTRACT

Impacts on soil quality should be included in life cycle assessments because of the essential role that soils play in ecosystem functioning. We propose a method that integrates impacts on quality of agricultural soils (erosion, soil organic matter, and compaction) of each stage of an agricultural product as a function of the soil and climate contexts of its agricultural processes. Input data must be as site-specific and accurate as possible, but if measured data are missing, the method has a standardised framework of rules and recommendations for estimating or finding them. We present a case study focused on the soil-quality impacts of producing pig feed in Brittany, France. The framework allows for incremental improvement of the method through the inclusion of new soil-quality impacts.

Keywords: soil quality, life cycle assessment, indicators, site dependence

1. Introduction

Soils are an essential resource in both managed and natural systems, and maintaining soil quality is critical to the sustainable development of human activities, in particular agriculture. The difficulty in representing impacts on soil quality remains an unresolved problem in Life Cycle Assessment (LCA) because of soil's spatial and temporal variability and the complex interactions among soil properties. Current status of soil quality in LCA in presented in (Garrigues et al., 2012). It is crucial to consider soil quality in the environmental assessment of products, especially those with a majority of their life cycle in bio-based processes (such as agriculture and forestry).

Soil is defined herein as naturally occurring, unconsolidated mineral or organic material at least 10 cm thick that occurs at the earth's surface and is capable of supporting plant growth. In this definition "naturally occurring" excludes displaced materials such as gravel dumps and mine spoils, but "unconsolidated material" includes that compacted or cemented by soil-forming processes. Soil quality can be defined by its capacity to function (Karlen et al., 1997) and/or its fitness for use (Larson and Pierce, 1994; Letey et al., 2003).

The objective of this study was to establish a framework for quantifying indicator(s) of impact on soil quality in a life cycle perspective, valid for all soil and climate conditions, and considering both on-site and off-site agricultural soils. The method developed answers needs identified by Garrigues et al., (2012) for LCA indicators of impacts on soil quality. It includes the impact categories erosion, soil organic matter (SOM) and compaction. Erosion and SOM impacts already exist in LCA approaches (Milà i Canals et al., 2007; Nuñez et al., 2010), but compaction impacts have yet to be quantified in detail in LCA. Cowell and Clift (2000) provided some ideas but excluded soil compactibility of their indicator. We applied the method to a case study of soil-quality impacts of producing pig feed in Brittany, France.

2. Method Presentation

2.1. General framework

Integrating soil-quality impacts throughout the life cycle of an agricultural product requires a global approach to assess impacts on soil quality that can be adapted to individual soil and climate contexts. Input data must be as site-specific and accurate as possible, but if measured data are missing, the method has a standardised framework of rules and recommendations for estimating or finding them.

Soil-quality impact assessment within LCA is quantified with midpoint indicators describing processes that can degrade or improve the soil. Soil physical, chemical and biological properties and function are excluded as indicators because of the difficulty in determining how they influence the system functions reflected in the functional units. Pathways were selected to link elementary flows of the inventory (LCI) to the midpoint impact indicators, which result from the combination of soil, climate, and management characteristics (Fig. 1).

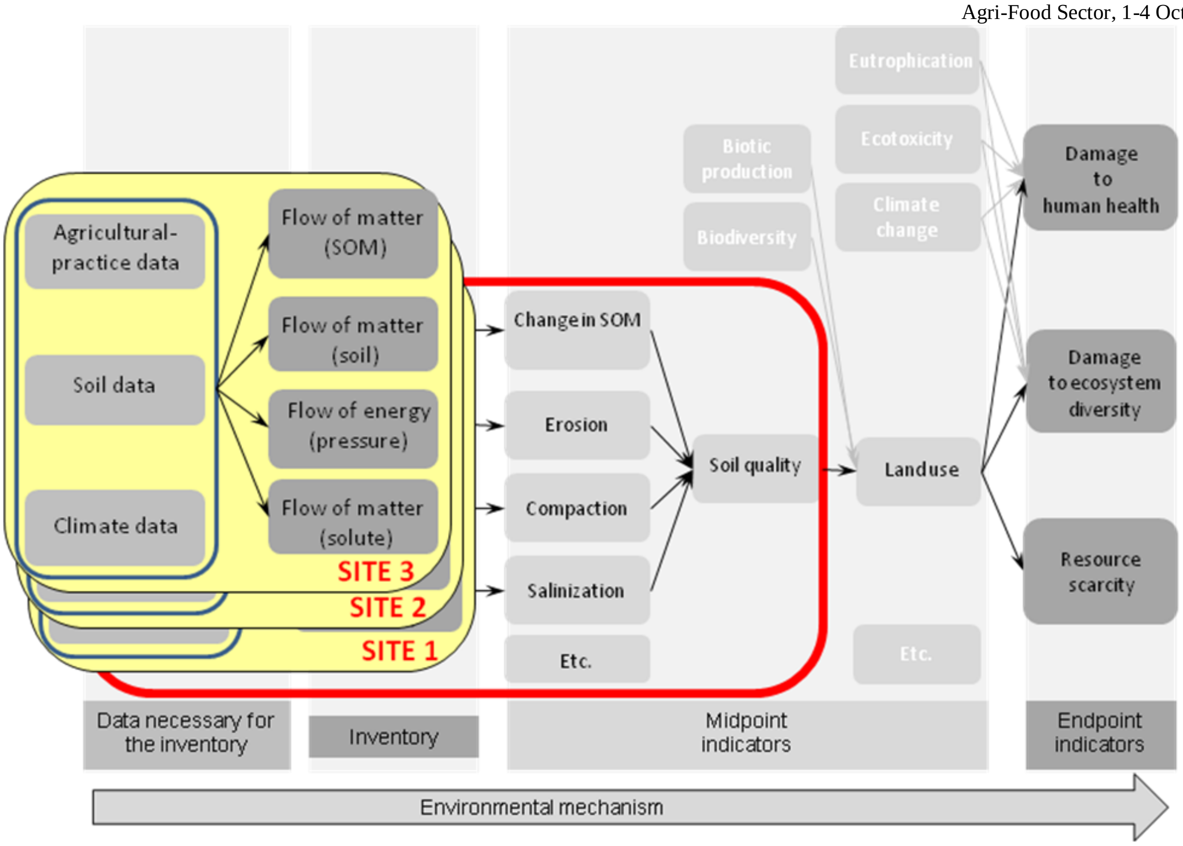


Figure 1. Steps for assessing impacts on soil quality (outlined) (adapted from Garrigues et al., 2012).

The LCI and impact assessment (LCIA) are based on simulation modelling, using models simple enough for use by non-experts, general enough to parameterise with available data at a global scale, and already validated: RUSLE2 for erosion (Renard and Ferreira, 1993); RothC for SOM (Coleman et al., 1997; Coleman and Jenkinson, 2008), and COMPSOIL for compaction (O'Sullivan et al., 1999). Most of the input data necessary for establishing the LCI are common to the three midpoint indicators. For each indicator, total impact is estimated by summing the impacts from individual upstream agricultural sites together. Thus, the method currently has no regionalised characterisation factors for LCIA, assuming that a given erosion, SOM, or compaction impact has equal impact regardless of location.

2.2. Inventory – input data requirements

2.2.1 *Soil data*

Soil characteristics such as texture, C content, bulk density, and slope are required. If necessary, national or international databases, such as the Harmonized World Soil Database (FAO et al., 2009), can provide the required data. In the future, the Global Soil Map project (http://www.globalsoilmap.net) will provide data at a finer resolution. Users can assume that agricultural processes in a region occur on its dominant soil type, a compromise between data precision and availability.

2.2.2 *Agricultural-practice data*

Crop data (e.g., yield, residues), management data, and vehicle characteristics (e.g., type and weight of vehicle, tyre size) are necessary.

2.2.3 *Climate data*

Monthly (erosion and SOM) and daily (compaction) temperature, precipitation, and potential evapotranspiration (PET) data are necessary. For SOM predictions, a time series up to 20 years is preferable. We used the TURC method (Federer, 1996) to estimate PET:

$$PET=0.313\ T_m\ (R_s + 2.1)/(T_m + 15) \text{ with } PET=0 \text{ when } T_m<0 \qquad (1)$$

where T_m is mean daily air temperature (°C) and R_s is daily solar radiation (MJ/m²).
If climate data are not found from national sources, international climate databases exist (NASA, 2012).

2.3. Simulation models used

2.3.1 *Erosion: RUSLE2*

The RUSLE2 (Revised Universal Soil Loss Equation) model (Renard and Ferreira, 1993) improves upon the original USLE model. The fundamental equation is:

$$A = R \times K \times LS \times C \times P \quad (2)$$

where A is the computed annual soil loss, R is the rainfall-runoff erosivity factor, K is the soil erodibility factor, LS is a topographic factor combining slope length (L) and steepness (S), C is a cover-management factor, and P is a supporting-practice factor. Three input databases are required that describe climate, crops and field operations.

2.3.2 *Soil* o*rganic matter change: RothC*

RothC (version 26.3) simulates the dynamics of organic carbon (C) in soil (Coleman and Jenkinson, 2008). The effects of soil type, temperature, moisture content and plant cover are considered in the turnover process. It uses a monthly time-step to calculate total organic carbon (TOC, t/ha) and microbial biomass carbon (t/ha) over one to hundreds of years. The method simulates 20 years of the same management practice and divides the total change in SOM by 20 to provide the rate over one year.

2.3.3 *Compaction: COMPSOIL*

COMPSOIL (O'Sullivan et al., 1999) predicts the effect of an agricultural machine on soil bulk density using readily available machine and tyre data. Topsoil and subsoil compaction are reported both separately (0-30 and 30-50 cm, respectively) and together. Initial dry bulk density comes from the SOTWIS database (ISRIC, 2012) from which soil texture is divided into five classes (coarse to very fine, according to the FAO texture triangle), each associated with an initial bulk density. Soil water content, a required input, is predicted from soil and precipitation data with the two-reservoir BILHY model (Jacquart and Choisnel, 1995). The method assumes uniform initial bulk density and water content profiles.

2.3. Indicators of impact on soil quality

The erosion indicator represents a loss of soil (t), while the SOM indicator represents an increase or decrease in the stock of soil C (t C). The compaction indicator represents a loss of soil porosity (m^3/ha) and distinguishes topsoil from subsoil compaction because the former is more easily reversible. Estimates of these soil processes in an inventory level are already informative enough to serve as indicators impact without requiring characterisation factors. A single indicator of impact on soil quality has not yet been developed because of the difficulty in aggregating diverse impacts into a single measure.

2.4. Case study

The case study was selected to illustrate impacts of a composite product formed from crop-based ingredients produced with widely differing soils, climates, and crop-management practices. It focused on the global soil-quality impacts of producing pig feed in Brittany, France, with ingredients coming from Brittany, Brazil, and Pakistan (Table 1).

Table 1. Ingredient composition (by mass) and sources of representative pig feed produced in Brittany.

Ingredient	Maize	Wheat	Triticale	Barley	Pea	Rapeseed cake	Soya cake	Soya oil	Molasses
Soil type	Loam	Loam	Loam	Loam	Loam	Loam	Clay		Loam
Country	France (Brittany)	France (Brittany)	France (Brittany)	France (Brittany)	France (Brittany)	France (Brittany)	Brazil (Santa Catarina)		Pakistan
Source crop	Maize	Winter wheat	Triticale	Barley	Pea	Rapeseed	Soya		Sugar-cane
Yield (t DM/ha)	9.0	7.0	7.0	6.5	4.2	3.3	2.8		35.0
Pig feed ingredient (%)	3.1	34.5	14.6	4.3	16.3	8.8	1.1	7.8	3.6
Economic allocation (%)	100	100	100	100	100	23.8	65.4	34.6	18.1

The system boundary for crop products used as feed ingredients was set at the farm gate, while that for feed ingredients was set at the factory gate. For each crop, the temporal boundary included the inter-crop period (if any) that occurs just before the crop. Impacts were predicted per ha of each crop and then converted to impact per tonne of feed ingredient based on crop yields and economic allocation for rapeseed cake, cane molasses and soya oil and cake. Impacts per tonne of ingredient in pig feed were added together to calculate total impact per tonne of pig feed produced in Brittany.

3. Results and discussion

Table 2 shows predicted erosion, SOM change, and compaction impacts per t of feed; note that SOM change is the only indicator that can have a negative value.

Table 2. Erosion, change in soil organic matter (SOM), and compaction impacts per tonne of pig feed in Brittany.

EROSION	SOM CHANGE	COMPACTION
0.177 t soil/t feed	-0.026 t C/t feed	Topsoil: 17.6 m^3/t feed Subsoil: 5.9 m^3/t feed Total: 23.5 m^3/t feed

For erosion, despite constituting only 9% (by mass) of pig feed (Fig. 2 left), soya-based ingredients (cake + oil) contributed 69% of the impact (Fig. 2 right). Agriculture-related erosion in Brazil tends to be higher than in Europe, especially in the location where we assumed soya to originate: Santa Catarina state, where precipitation and mean slopes are high. The erosion model used, RUSLE2, represents well the high sensitivity of erosion to precipitation and mean slope.

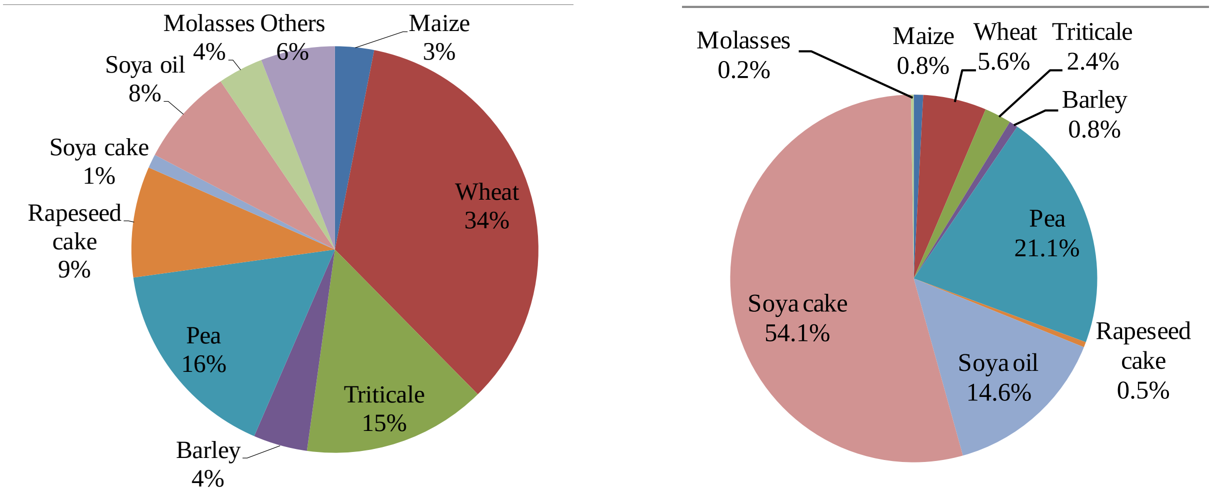


Figure 2. Left: Ingredient composition by mass of representative pig feed produced in Brittany. Right: Contribution of crop-based ingredients to erosion impact per tonne of pig feed in Brittany (total soil loss: 0.177 t soil/t feed).

For SOM change, rapeseed cake and pea contributed most to the net negative impact (Fig. 3). In Brittany pea is mostly cultivated with only mineral fertilisation and has few residues left on field. The C dynamics simulate by Roth C is sensitive to manure and plant-residues supplies. Furthermore, the soil of Brittany has a high C content (2.5% TOC), which requires high C input over the 20 years of simulation of pea cultivation to be able to maintain it.

For compaction, as crops grown in Brittany require similar agricultural practices, the relative impact of each ingredient is similar to its relative mass in the feed (Fig. 4). The impact of wheat is relatively higher because of the highest number of passes in field than the others crops. In Brazil, machines are heavy, but reduced-tillage practices result in fewer passes than in Brittany. Furthermore, Brazilian soils have high clay contents, which decrease their sensitivity to compaction (unlike the loamy soils of Brittany).

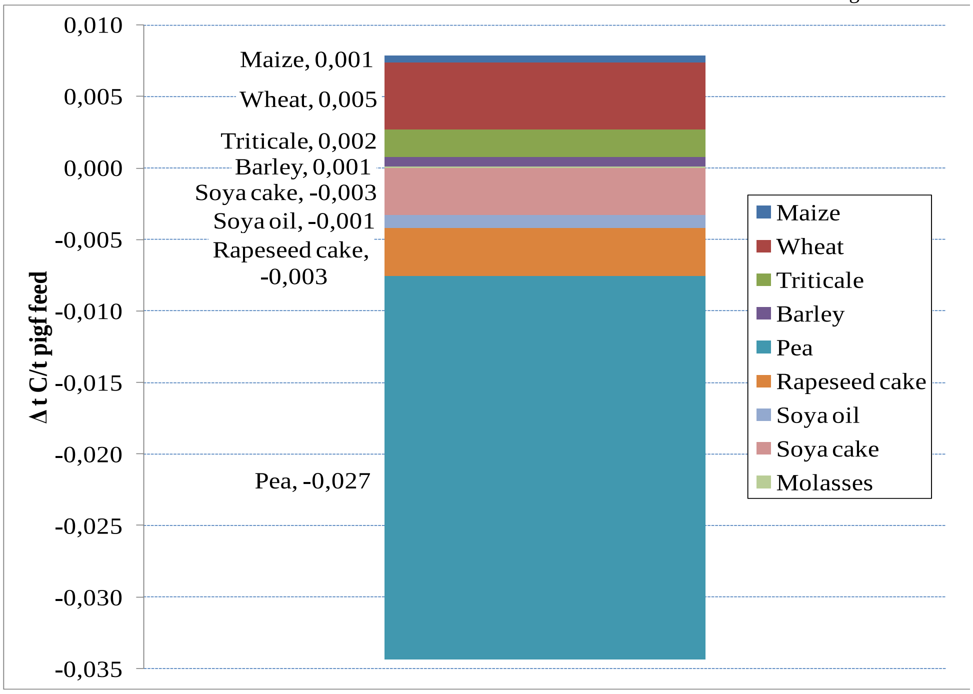


Figure 3. Contribution of crop-based ingredients to impact of change in soil organic matter (SOM) per tonne of pig feed in Brittany (net SOM change: -0.026 t C/t feed).

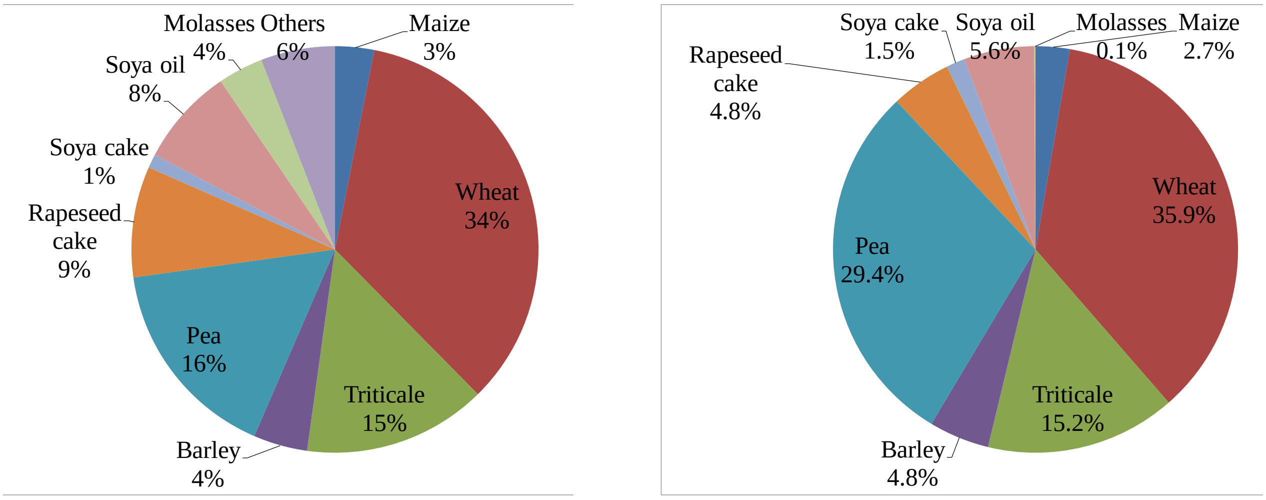


Figure 4. Left: Ingredient composition by mass of representative pig feed produced in Brittany. Right: Contribution of crop-based ingredients to compaction impact in the upper 50 cm of soil (topsoil + subsoil) per tonne of pig feed in Brittany (loss of porosity: 23.5 m^3/t feed). The subsoil compaction is irreversible.

These soil-quality impact indicators can be used in LCAs of bio-based materials (e.g., plants, wood, food, industrial bio-based materials) for cultivation processes or for waste-management processes if considering composting. Although, the impact of non-cultivation processes on soil quality could be included, many of them, such as soil sealing with concrete, transform soil into nonsoil, which has zero soil quality. Thus, we believe that non-cultivation processes are better included with impacts of land use and land-use change.

In a life cycle perspective, soil-quality impact indicators can interact with other impact categories, such as climate change, in which SOM changes influence net C emissions into the atmosphere. Also, soil-quality impact indicators complement other impact categories, providing increased ability to identify "impact swapping" or trade-offs between transport distance and agricultural soil quality.

All the input data necessary for establishing the LCI (approximately 30 parameters) will be presented in the project report in late 2012 and can be found in the international databases cited. Most input data are common to the three indicators. As the framework allows for incremental improvement, the inclusion of new soil-quality impact indicators (such as salinisation) will increase input-data requirements little. A forthcoming users' guide will describe each step of the method.

5. Conclusion

Impacts on soil quality should be taken into account into a life cycle perspective because of the essential role of soils in ecosystem functioning. We have developed a framework for quantifying indicators of impact on soil quality, valid for all soil and climate conditions, and considering both on-site and off-site agricultural soils. These indicators can be used in LCAs of bio-based materials or the waste-management stage when considering composting.

The first indicators developed represent the most prevalent threats on soil: erosion, SOM change and compaction. Overall impact estimates result from the combination of soil, climate, and management characteristics. Results to date can begin to fill a database of soil-quality impact indicators for crops and crop-based products from a several regions. Most of the input data necessary for establishing the LCI are common to the three midpoint indicators and can be found in existing databases.

The framework allows for incremental improvement of the method through the inclusion of new soil-quality impacts. Improvement efforts will focus first on developing robust impact indicators for individual soil processes before considering whether to aggregate them into a single indicator. Nonetheless, a variety of aggregation approaches can be explored (Garrigues et al., 2012).

6. References

Coleman, K., Jenkinson, D.S., 2008. ROTHC-26.3 A model for the turnover of carbon in soil. Model description and windows users guide. Harpenden, Herts, UK, p. 47.

Coleman, K., Jenkinson, D.S., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter, D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma 81, 29-44.

Cowell, S.J., Clift, R., 2000. A methodology for assessing soil quantity and quality in life cycle assessment. Journal of Cleaner Production 8, 321-331.

FAO, IIASA, ISRIC, ISSCAS, JRC, 2009. Harmonized World Soil Database (version 1.1). FAO, Rome, Italy and IIASA, Laxenburg, Austria.

Federer, C., 1996. Intercomparison of methods calculating potential evaporation in regional and global water balance models. Water Resources Research 32, 2315-2321.

Garrigues, E., Corson, M.S., Angers, D.A., van der Werf, H.M.G., Walter, C., 2012. Soil quality in Life Cycle Assessment: Towards development of an indicator. Ecol. Indic. 18, 434-442.

ISRIC, 2012. Harmonized continental SOTER-derived database (SOTWIS) (http://www.isric.org/projects/harmonized-continental-soter-derived-database-sotwis).

Jacquart, C., Choisnel, E., 1995. Un modèle de bilan hydrique simplifié à deux réservoirs utilisable en agrométéorologie. La Météorologie 8ième série - n°9 - mars, 29-44.

Karlen, D.L., Mausbach, M.J., Doran, J.W., Cline, R.G., Harris, R.F., Schuman, G.E., 1997. Soil quality: a concept, definition, and framework for evaluation. Soil Science Society of America Journal 61, 4-10.

Larson, W.E., Pierce, F.J., 1994. The dynamics of soil quality as a measure of sustainable management. Defining soil quality for a sustainable environment. Soil Science Society of America, pp. 37-51.

Letey, J., Sojka, R.E., Upchurch, D.R., Cassel, D.K., Olson, K.R., Payne, W.A., Petrie, S.E., Price, G.H., Reginato, R.J., Scott, H.D., Smethurst, P.J., Triplett, G.B., 2003. Deficiencies in the soil quality concept and its application. Journal of Soil and Water Conservation 58, 180-187.

Milà i Canals, L., Romanya, J., Cowell, S.J., 2007. Method for assessing impacts on life support functions (LSF) related to the use of 'fertile land' in Life Cycle Assessment (LCA). Journal of Cleaner Production 15, 1426-1440.

NASA, 2012. Climatology Resources for Agroclimatology Daily Average Data (http://power.larc.nasa.gov/cgi-bin/cgiwrap/solar/agro.cgi?email=agroclim@larc.nasa.gov).

Nuñez, M., Civit, B., Muñoz, P., Arena, A.P., Rieradevall, J., Anton, A., 2010. Assessing potential desertification environmental impact in life cycle assessment. The International Journal of Life Cycle Assessment 15, 67-78.

O'Sullivan, M.F., Henshall, J.K., Dickson, J.W., 1999. A simplified method for estimating soil compaction. Soil & Tillage Research 49, 325-335.

Renard, K.G., Ferreira, V.A., 1993. RUSLE model description and database sensitivity. Journal of Environmental Quality 22, 458-466.

Finnish consumer understanding of carbon footprinting and food product labelling

Hanna Hartikainen*, Juha-Matti Katajajuuri, Hannele Pulkkinen, Taneli Roininen

[1] MTT Agrifood Research Finland, 00790 Helsinki
* Corresponding author. E-mail: *hanna.hartikainen@mtt.fi*

ABSTRACT

Production and consumption of food products have significant impacts on climate change. Carbon labelling of food products represents a major means of communicating the impacts of food products on climate. However, there is little knowledge on how consumers perceive carbon labelling. The aim of the Climate Communication 2 (2011-2013) project was to study how Finnish consumers perceive the communication of carbon footprints for food products. The study comprised 5 semi-structured focus groups and an online-survey. The focus groups showed that the term 'carbon footprint' was familiar to many, but there was substantial misunderstanding of its meaning. There were positive attitudes towards carbon labels, but the participants indicated that the information became meaningful only when other key purchasing criteria (such as price and taste) were satisfied. Furthermore, there was divergence on perceived needs for carbon label content.

Keywords: carbon footprint, carbon labelling, food, consumer behaviour, survey

1. Introduction

In Finland around 25% of greenhouse gas emissions (GHG-emissions) from private consumption originate from the production and consumption of food, including household food preparation, food preservation, journeys to shops and meal services (Regina et al., 2011; Seppälä et al., 2009). Communication of climate impacts for food products is highlighted by there being no great reductions made solely by adapting low carbon technology (Weidema et al., 2008). Therefore, the greatest potential for reducing GHG-emissions from food products lies in consumer behaviour.

One way to inform consumers about food product GHG-emissions is carbon labelling of food products, which has expanded steadily in Finland during recent years. The first carbon label appeared in 2008, and to date six Finnish food companies include carbon labels on their product packages. Overall more than 40 Finnish food products are carbon labelled and more will be labelled in the future. Some Finnish food companies are also communicating product carbon footprints only on their websites. Additionally, some state that they compensate for their product carbon footprints. Consumer perceptions of product carbon footprint and attitudes toward climate-friendly products remain challenging nonetheless. It is unclear whether Finnish consumers seek information on climate-friendly food products or not. From recent international studies it is indicated that there is a growing need among consumers for accurate information about the impacts of food and its production on climate (i.e. European commission 2009, The Climate Group 2006). However, there is also a need for a deeper and more up-to-date study.

In the Finnish Climate Communication 2 project (2011-2013) Finnish consumer understanding about carbon footprinting and information needs are studied. Appreciating the complexity and broad scope of the topic, the aim of the study is to establish the nature of:

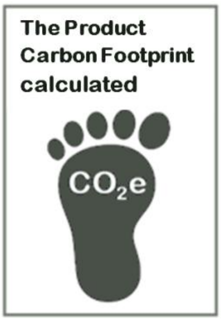
- Consumer perception of product carbon footprints and labels
- Consumer understanding of the message transmitted by carbon labels
- Consumer perception of the information content in the carbon label

2. Methods

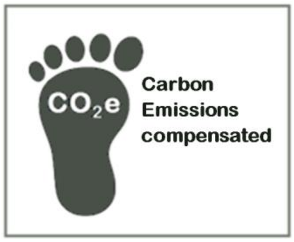
There were five focus groups in the group discussions and 33 participants in total. The key criterion for consumer recruitment was that the consumer stated that environmental friendliness was at least quite important when grocery shopping (by giving a score of at least 3 on a scale of 1 to 5, 5 being very important), and thus being more concerned than the average consumer in that respect. The participants were grouped according to their ages. There was a group of young adults (ages 24-28), two middle-aged groups (ages 31-44) and an elderly group (ages 53-65). Additionally, the members of the fifth group (ages 28-47) considered themselves even more environmentally conscious in comparison with other focus groups when grocery shopping.

Each group discussion lasted around two hours: first the participants discussed their own criteria for grocery shopping, then they discussed how environmentally conscious they were in general and how their consciousness related to food consumption. Lastly the groups discussed carbon footprinting and carbon labelling of food products. The participants were also shown various carbon labels from which they chose their favourites (Figures 1a-f). Each of the labels emphasised one of the following information points:

- The product carbon footprint is calculated (Figure 1a).
- The product GHG-emissions are compensated for by emission reductions elsewhere (Figure 1b).
- The company is committed to reducing its product carbon footprint by xx% per year (Figure 1c).
- The product has a low carbon footprint compared with other food products in its category (e.g. cereal products) (Figure 1d).
- The product carbon footprint is indicated on a scale (on a scale with five ranges: 200 or less, 200-400, 400-800, 800-1200, 1200 or more grams CO_2-equivalents per 100 gram of product). The scale is broad enough for all food products, and thus it enables rough comparisons between different food product categories to be made (Figure 1e).
- The product carbon footprint is indicated numerically: xx CO_2-equivalents per 100 gram of product (Figure 1f).



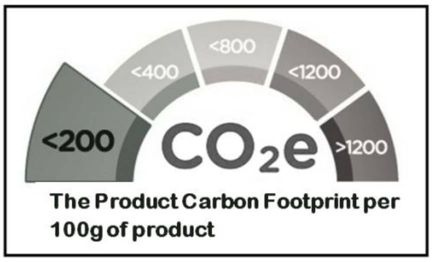
1a.



1b.

1c.

1d.



1e. (edit. Raisio 2011)

1f.

Figures 1a-f. Carbon labels.

3. Results

In the discussions all the participants (when grocery shopping) regarded many other buying criteria, such as taste, price, and healthiness, as more important factors than environmental friendliness. Only a couple participants regarded food as one of the main environmental stress factors, whereas housing and transportation were mentioned often. However, when asked directly: *Does food impact the environment?,* the participants replied that food had an at least some impact on the environment. All groups mentioned food packaging waste, food waste, food transport, and meat production as having an impact on the environment. Also, in the elderly group (ages 53-65) there was more talk about food scarcity.

A surprising finding was that while the participants had heard of the term 'product carbon footprint', only two respondents were able to describe the term at least somewhat accurately. About 50 per cent of respondents thought that the term referred to environmental impacts in general, and 6 respondents thought that the carbon footprint would be calculated by taking into account only the energy needed to produce the product.

Furthermore, interestingly, terms like ‘climate impact’ and ‘global warming’ were not mentioned in the context of environmental impacts of food. Similar findings were also made in a UK study where 89 per cent of respondents had confusion in interpreting and understanding carbon labels (Gadelma and Oglethorpe, 2011).

After being provided with the correct definition of ‘product carbon footprint’ the participants were single-minded that communicating product carbon footprints was positive. However, the majority agreed on the focus being quite narrow. Many claimed that other environmental impacts should also be taken into account. Participants also questioned whether communicating product carbon footprints would really have any impact on their buying behaviour. For instance, they stated that at their current level of understanding of carbon footprint it would be hard to put the information into perspective and say whether a carbon footprint was low or high. However, several participants stated that a carbon label could have a positive impact on their purchase decisions, but only when the choice was to be made between two otherwise comparable food products.

The most preferred carbon label among the participants (12 of the 33 respondents preferred this carbon label) was that providing the carbon footprint according to a scale (Figure 1e). Many considered the label to be illustrative because it indicated an approximate product carbon footprint in comparison with that for other food products. Meanwhile, some found the label to be quite confusing.

Other favourites were the label showing that the product had low GHG-emissions in comparison with other products in its product category (6/33) (Fig. 1c), and the label giving the carbon footprint as an exact number (5/33) (Fig. 1f). The former label (Fig. 1c) was considered good because its message was easy to understand, although it did not allow comparisons between different product categories. The latter label (Fig. 1f) was considered good because if a company were to use such a label on its product packages it would seemingly require considerable input from the company, and which could lead to reductions in the product carbon footprint. However, the information of such a label was seen as being rather abstract without adequate an understanding of the magnitude of the figure. Additionally, some participants stated that the exact figure should be on all products in order to make valid choices based on it.

The remaining carbon labels did not get much support. ‘The carbon label has been calculated’ label (Fig. 1a) was quite poorly received because it did not indicate magnitude, and thus the footprint could be relatively high. Additionally, some participants saw a potential risk of ‘green washing’ on a label stating that the company was committed to reducing its product carbon footprint on an annual basis (Fig. 1c).

4. Discussion and conclusion

Overall group discussions deepened understanding of how some consumers currently perceive carbon footprinting and labelling, and how unclear, but interesting, the issue is and could be in the future. Firstly, the participants stated that the issue is at least somewhat interesting and important, but there seem to be diverse requirements for the type of information to be included in a carbon label. The most definite outcome from the discussions was that there is a clear need to educate consumers to understand better the concept of product carbon footprint. This would allow better understanding of the importance of the product carbon footprint information to consumers. Again, it is also quite interesting to see how consumers perceive carbon footprinting and labelling given their current level of understanding. Whether accurate understanding of carbon footprinting would increase demand for carbon labelled food products is still uncertain.

The focus groups gave only a narrow view on the consumer thoughts on the issue, and therefore to get a broader view and perhaps get answers to some unanswered questions and issues raised in the group discussions, the next step was an extensive quantitative survey (held in spring 2012). The aim of the quantitative survey was to obtain around 1000 responses from a miscellaneous consumer group (e.g. different age groups, different levels of environmental consciousness etc.).

5. References

European Commission, 2009. Europeans’ attitudes towards the issue of sustainable consumption and production, Analytical report, Flash EB Series 256

Gadema Z., Oglethorpe D. 2011. The use and usefulness of carbon labelling food: A policy perspective from a survey of UK supermarket shoppers. Food Policy 36: 815–822

Raisio 2011. http://www.ekologia.fi/www/page/Ekologia_CO_merkki

Regina K., Lehtonen H., Perälä P., 2011. The role of agriculture as a greenhouse gas emissions producer and as a part of climate policy (in Finnish): Maatalouden rooli kasvihuonekaasupäästöjen tuottajana ja osana ilmastopolitiikkaa, Selvitys Maa- ja metsätalousministeriölle, Maa- ja elintarviketalouden tutkimuskeskus (MTT)

Seppälä J., Mäenpää I., Koskela S., Mattila T., Nissinen A., Katajajuuri J-M., Härmä T., Korhonen M-R., Saarinen M., Virtanen Y., 2009. The environmental impact assessment of material flows of Finland's national economy with ENVIMAT-model (in Finnish): Suomen kansantalouden materiaalivirtojen ympäristövaikutusten arviointi ENVIMAT-mallilla, Suomen Ympäristö 20/2009, Suomen ympäristökeskus (SYKE)

The Climate Group, Carbon footprint consumer survey (2006)

Weidema B. P., Wesnæs M., Hermansen J., Kristensen T., Halberg N., Eder P., Delgado L., 2008. Environmental improvement potentials of meat and dairy products. European Commission Joint Research Centre (JRC) – Institute for Prospective Technological Studies (IPTS) EUR 23491 EN 2008

Application of PAS 2050-1 supplementary requirements for horticultural products: carbon footprint of pumpkin and asparagus

Michael Blanke*, Florian Schaefer

INRES - University of Bonn, Auf dem Hügel 6, D-53121 Bonn, Germany
* Corresponding author. E-mail: *mmblanke@uni-bonn.de*

ABSTRACT

This contribution provides information on the new PAS 2050-1 supplementary requirements (formerly 'Product Category Rules – PCRs) for horticultural products released in March 2012. This is one of the first supplementary requirements and may be used as a guideline for industries other than horticulture such as the food industry. The data presented originate from one of the five international pilot projects. In the German application trial, asparagus, strawberry, rhubarb and pumpkin were employed. The new PAS 2050-1 includes recommendations for a "cradle to gate" or "business to business" approach. It provides an Excel tool for both land use change (LUC) and the nitrogen balance within a crop rotation. Examples will be given for issues to be excluded such as Capital goods or included such as biogenic carbon of the horticultural product. The objective of this contribution was to determine experiences (pro and cons) of the implementation of these new supplementary requirements ('SRs') for horticultural products to the PAS 2050: 2011 as part of this pilot study, while calculating the carbon footprint for the four crops, including autumn pumpkin and asparagus in Germany.

Three pumpkin farms with different pumpkin cultivation were chosen: a) a small scale organic (50 kg N/ha), b) a small scale integrated production (IP) (120 kg N/ha), and c) a large scale business enterprise (70 kg N/ha). Area *viz.* hectare was chosen as the first and mass i.e. kg saleable product as the second functional unit. System boundaries ranged from plantlet or seed acquisition to sale. The carbon footprint at the cultivation level (FCF) ranged between 157 kg CO_2eq/ha (organic) and 251 kg CO_2eq/ha (small scale (IP)). Taking the yield into account the mass specific Carbon Footprint was from 8 g CO_2eq/kg saleable pumpkin to 20 g CO_{2eq}/kg saleable pumpkin. The nitrous oxide emissions, which originated from the nitrogen fertilisation, were calculated based on 0.7% N_2O per kg applied N. They were the most relevant source of GHG emissions in the cultivation phase. Neither the form (organic or inorganic) nor the amount of applied nitrogen (2.5-fold difference) influenced the carbon footprint. However, carbon reduction potentials include use of nitrification inhibitors such as DMPP and DCD, which reduce the nitrous oxide emissions by ca. 47 or 40%, respectively, or the CULTAN fertilisation system. Plant protection (methiocarb in the IP) contributed less than 1% to the carbon footprint. The large specialised farm showed the best carbon footprint of 8 g CO_2eq/kg pumpkin due to the use of potassium fertiliser and 3-fold larger yields (18 t/ha versus 5.8 t/ha in the organic) and, to a lesser extent, its sheer scale. On the other two farms, cultivation is more extensive with the main income not from pumpkin; any increase in farm size or their pumpkin acreage would not improve their efficacy and cradle-to-gate carbon footprint.

The farm carbon footprint of asparagus (FCF), also corrected for biogenic carbon (0.198 t CO_2-e/ha/a), was 2.8 t CO_2-e/ha. The product carbon footprint for asparagus (PCF – B2C), calculated over an 11-year life-span of the orchard in Germany including the first unproductive years, was 801 g CO_2eq/kg saleable asparagus.

In conclusion, the present application trial led to 3-17% less product carbon footprints (PCF) due to the offset of the biogenic carbon in the harvested produce and confirmed the benefits and suitability of PAS 2050-1 for the horticultural industry.

Keywords: carbon footprint, PAS 2050-1, horticultural products, application trial, land use change (LUC)

1. Introduction

The PAS 2050-1 'Assessment of life cycle greenhouse gas emissions from horticultural products' was developed as one of the first product category rules (PCR) and for the agri-food sector. Other sectors may develop their own PCRs and use or modify the PAS 2050-1 in full or in part. The authors were part of the steering group for the development of this PAS 2050-1 and hosted one out of five pilot projects for its potential implementation in horticulture (Table1). This included pumpkin, asparagus, rhubarb and strawberry in Germany (BSI, 2012).

Table 1. Pilot projects for the PAS 2050-1 (horticulture) during winter 2011/2012

Pilot project institution	Country	Crop
Food Research	New Zealand	Kiwi (open cultivation)
Karen Fisher	Great Britain	Orange juice (product)
IRTA , Lleida	Spain	Protected apple under hailnet
Productshap Tuinbouw, Gouda	Netherlands	Greenhouse vegetables and flowers, nursery trees and container plants
University of Bonn, INRES – Horticultural Science	Germany	Pumpkin, rhubarb, asparagus, strawberry (all field production)

2. Materials and Methods

2.1 Features of PAS 2050-1 (March, 2012)

The PAS 2050-1 is now designated as 'Supplemental Requirements' (SR) to PAS 2050 rather than as 'PCR- Product Category Rule(s)'. It is the first, freely and publicly available guideline for all horticultural products, including plants, production, flowers, fruits, nuts, vegetables, nursery trees and orchards, for assessment of their inherent/associated greenhouse gas emissions. This PAS 2050-1 is suited for 'cradle-to-gate' assessment, formerly designated as "B-2-B" ("business to business") approach. For cradle-to-grave assessments of horticultural products/production to the final consumer, the first part of the assessment is supported by the PAS 2050-1 (hort; March, 2012) and the second latter part by the PAS 2050 (2011). Overall, the following life cycle processes are included (BSI, 2011).

- Seed or young plant production;
- Storage of young plantlets before planting
- Crop cultivation (e.g. fertiliser, plant protection, cultivation, harvest)
- Storage of crop products
- Transport (from and to the field);
- Waste management (at farm level)

Annual crops are defined as plants sown or planted during one production season, which is the minimum reporting period for carbon footprint assessment.

Perennial crops with harvests over several productive seasons require a reporting time of at least 3 years (rolling average), which includes all developmental stages in proportion. In fruit or nut trees (or sugar cane, bananas, asparagus), these include the juvenile phase without yield and ageing phase of lesser yields.

Carbon sequestration, as part of a horticultural production scheme, is calculated over the last 20 years and considered under land use change (LUC). An Excel tool for LUC calculations is provided, which includes the relevant worldwide data, e.g. for most horticultural crops as well as deforestation. LUC results are included in the overall carbon footprint values, but are not separated from the carbon footprint value as in the World Business Council for Sustainable Development/World Resource Institute (WBCSD/WRI) Standard, since they may have a great contribution to the calculated values.

Table 2. Features of the PAS 2050-1

Feature	Previous approach	New approach in PAS 2050-1
Wording/ designation [a]	Product Category Rule -PCR	Supplemental Requirements (SR)
Only B to B approach	PAS 2050 2011 for B to C	PAS 2050-1 (hort) for B to B
Perennial crops	n.a.	minimum 3 years (rolling average) to proportionally include all developmental stages (e.g. juvenile phase without yield)
Annual crops	n.a.	minimum 3 years or at least 3 productive cultivation cycles
Carbon sequestration	Separate entity	Integrated in L U C
LUC values	100 years	approach based on the last 20 years and integrated in overall CF value
Biogenic carbon (if for food or feed)	Not included	Offset against horticultural production
Biogenic carbon (if not for food or feed- e.g. tree trunk for timber/furniture production)	Included	Offset against horticultural production
Capital goods	Excluded	Excluded in carbon footprint (e.g. greenhouses and superstructures for polytunnels)
Consumables	Included	Materials (film and foils etc.) given as examples, which are replaced regularly are included
Fertiliser	Single crop assessment	Crop rotation
Calculation tools	Not provided	Calculation tools in Excel with data basis provided for both L U C and allocation of fertiliser (incl. organic) for crop rotation systems

2.2 Biogenic carbon

Biogenic carbon, i.e. the carbon contained in the harvested product exported outside the system boundary (out of the farm gate), is offset in the PAS 2050-1 against the carbon emissions during horticultural production, if used for food or feed. Similarly, biogenic carbon not used for food or feed, e.g. tree trunks for the timber or furniture industry, can be offset against horticultural production (BSI, 2012) (Table 2).

Capital goods in horticulture such as greenhouse support structures for poly-tunnels, buildings, grading facilities and cold stores are defined and excluded from the Carbon Footprint calculations in PAS 2050-1.

Consumables, which are replaced on a regular basis, such as plastic foils for plant covering and growing substrates, are included. Similarly, fertilisers and plant protection compounds are included in the product carbon footprint.

3. Results

3.1. Carbon footprint of pumpkin

The results presented here are part of one out of five pilot projects of the PAS 2050-1 (Table 1). Autumn pumpkin was chosen here as one of the crops from the pilot project. The objective of the study was to determine the carbon footprint for four farming and marketing systems, using primary data obtained on the farms for autumn pumpkin as an example and model crop for the pilot project to test the application of the PAS 2050-1 during its draft version. Acreage, viz. hectare for the crop cultivation and weight, i.e., kg saleable product for the marketing phase were employed as the two functional units with system boundaries from seed acquisition to disposal; offset was not used.

In the farm carbon footprint (FCF), pumpkins from the organic farm with 240 kg CO_2eq/ha scored best due to the least (50 kg N/ha) (organic) nitrogen application, compared with twice that value (448 kg CO_2 eq/ha) for those from the IP farm and large specialised farm. This was due to nitrous oxide emissions, as a consequence of N fertilisation, with 99% of FCF in the cultivation phase; it contributed *ca*. 10% and plant protection (methiocarb in the IP) and <1% of the product carbon footprint (PCF). Neither the form (organic or inorganic) nor the amount of applied nitrogen (2.5-fold difference) influenced the final product carbon footprint. The biogenic carbon of pumpkin is 42 g CO_2/kg pumpkin. This value is based on its 91% water content and the dry matter content of 90 g/kg with 50% carbon. The value is now offset against the pumpkin cultivation following PAS 2050-1 (March 2012)

The large specialised farm showed the best product carbon footprint with 146 g CO_2eq/kg pumpkin due to use of potassium fertiliser and 3-fold larger yields (18 t/ha versus 5.8 t/ha in the organic [728g CO_2eq/kg]) and, to a lesser extent, its sheer scale (Table 3). On the other two farms, cultivation is more extensive with the main income not from pumpkin; any increase in farm size or their pumpkin acreage would not improve their efficacy and product carbon footprint. The imported organic Argentinean pumpkin scored second best with 247 g CO_2eq/kg despite the long-distance transport due the lower energy consumption of bulk sea freight. The private consumer shopping in both cases (retail versus farm shop) amounted to as much as 89% of the product carbon footprint (Schaefer and Blanke, 2012).

Table 3: Carbon Footprint of pumpkins as dependent on cultivation and marketing systems

	Integrated Production (IP)	**Organic DEMETER**	**Large farm Business**	**Argentinean organic pumpkin**
From seeding to harvest (kg CO_2eq/ha)	448	240	264	240
From seeding to harvest (g CO_2eq/kg pumpkin)	35.8	41.3	14.6	41.3
From harvest to disposal (g CO_2eq/kg pumpkin)	687	687	131	207
Pumpkin yield (t/ha)	12.5	5.8	18.1	5.8
CO_2eq/kg pumpkin (g)	723	728	146	247
CO_2eq/kg pumpkin (g, after subtracting biogenic carbon)	681	686	104	205

3.2. Carbon footprint of asparagus

The carbon footprint of asparagus was calculated over the entire life-span including the productive and unproductive phases, i.e. 11 years. In this case study for the new PAS 2050-1 only one farm with one cultivation system was analysed to study the ease of implementation of the new rules. LUC was calculated with an Excel tool that was developed in the pilot project. The result of the LUC for asparagus following annual crop land was negative for our specific growing and environmental conditions. To avoid any criticism of offset it was assumed zero in our calculation. The biogenic carbon was calculated from its 6% dry matter with 47% carbon content based on average (11 year) yield (7t/ha) including the initial unproductive phase of the cultivation. The result for the Farm Carbon Footprint (FCF) was 2.208 t CO_2eq/ha asparagus, after the biogenic carbon was subtracted. Including the cooling, grading and packaging the business to business carbon footprint according to the new PAS 2050-1 was 401 g CO_2eq/kg asparagus (Table 4). The transportation to retail and the use phase (shopping tour, fridge and cooking) at the consumer amounted to 400 g CO_2eq/kg asparagus using the guideline of the PAS 2050:2011. The overall business to consumer (B 2 C) carbon footprint of 801 g CO_2eq/kg asparagus shows the result over the all life-cycle stages of asparagus.

Table 4. Carbon footprint of asparagus cultivated in an integrated production system

	Integrated Production (IP) of Asparagus
Farm carbon footprint per area [t/ha]	
Average asparagus yield per ha and per year (over 11 years)	7.020 t/ha
Yearly cultivation, tillage and planting (per ha) without biogenic carbon	2.406 t CO_2eq/ha/a
LUC (asparagus after annual cropland)	0.0 kg CO_2eq/ha/a*
Bio-genic carbon per ha (6% dry matter of asparagus)	-0.198 t CO_2eq/ha/a
From planting to harvest (over 11 years) (ha)	2.208 t CO_2eq/ha/a
Farm carbon footprint [g per kg]	
From planting to harvest (over 11 years) (kg)	315 g CO_2eq/kg asparagus
Cooling, grading and packaging (5 kg cardboard carton)	086 g CO_2eq/kg asparagus
Carbon footprint from seedling to harvest according to PAS 2050 -1 (B 2 B)	**401 g CO_2eq/kg asparagus**
Product carbon footprint [g per kg]	
Overall transportation farm to retail	096 g CO_2eq/kg asparagus
Use phase (shopping tour, fridge and cooking)	304 g CO_2eq/kg asparagus
Product Carbon Footprint from harvest via use phase to disposal	**400 g CO_2eq/kg asparagus**
Product Carbon Footprint (PCF) CO_2eq/kg asparagus **(B 2 C)**	**801 g CO_2eq/kg asparagus**

*LUC result was negative but assumed as zero.

4. Discussion- Carbon reduction potential

The pilot project PAS2050-1 sponsored by Productshap Tuinbow was successful and established guidelines, with examples and references for the horticultural sector, and appears the first product category rule (PCR). Carbon reduction potential for autumn pumpkin, chosen as part of the associated pilot project, appears in three sectors i) potassium fertilisation, ii) reduction of nitrous oxide (N_2O) emissions and iii) consumer behaviour. The first carbon reduction potential lies in improving *viz.*, increasing the potassium fertilisation in all farming systems, except for the large farm, which would increase yields and hence reduce the product carbon footprint (PCF). The second carbon reduction potential lies in the reduction of N_2O emissions: With 99%, nitrous oxides had the largest share of the carbon footprint during cultivation. Use of the global IPCC values of 1.0% (IPCC, 2006) or 1.25% (IPCC, 2007) N_2O-N per kg applied fertiliser-N would nearly double the N_2O from 0.8% during cultivation (Kuikman et al., 2006). By contrast, the use of nitrification inhibitors such as DMPP with a 49% reduction in nitrous oxide emissions from regional soils (Weiske et al., 2001) could halve the carbon footprint during cultivation. Alternatively, depot nitrogen fertilisers like the ammonium-based CULTAN system (Bacher and Blanke, 1996; Sommer and Scherer, 2009) could reduce the amount of applied nitrogen by ca. 25%.

The third carbon reduction potential is the consumer behaviour regarding the means of transport for shopping. The use of bicycles, public transport or non-fossil-based cars could reduce the product carbon footprint.

5. Conclusion

Overall, the application trial showed PAS 2050-1:2012 is suitable for the horticultural industry. These supplementary requirements enable the user in the horticultural business to have a clear and structured approach for the special issues in this specific sector. The new supplementary requirements, which offset biogenic carbon in the horticultural/harvested product (Table 2), result in lesser carbon footprint values than in previous studies. In the present pilot study, the product carbon footprint (PCF, cradle to grave, or B 2 C) was 3% (asparagus), 8% (rhubarb) and 17% (strawberry) less due to subtracting the biogenic carbon in the harvested produce. Hence, larger yields and produce with a large carbon contents play an increasing (positive) role in the carbon footprint; yields e.g. in pumpkin cultivated in the same region (soil and climate) with a different cultivation system (organic or IP) can vary by a factor of 3 (Table 3).

Our value of 0.801 kg CO_2eq/kg asparagus compares with another study on carbon emissions in the horticultural sector and 0.7-0.8 kg CO_2eq/kg asparagus in Switzerland (Stoessel et al., 2012), in which nitrogen fertiliser including related N_2O emission played the greatest role. The Swiss values may be 10-15% less, because the system boundary of the study ended at retail level. This section from retail to the consumer was 0.304 kg CO_2eq/kg asparagus in the present pilot study (Table 4), the PCF to retail amounted to 0.497 kg CO_2eq/kg asparagus in Germany.

6. References

Bacher, W., Blanke, M.M, Pring, R. Baker, E.A., 1996: Ammonium nutrition enhances chlorophyll and glaucousness in Brassica kohlrabi. Annals of Botany 78, 559-604.

BSI, 2011. PAS 2050:2011. Specification for the assessment of life cycle greenhouse gas emissions of goods and services. British Standards Institute. London, England

BSI, 2011. PAS 2050 Review 2011: Summary of the Changes to PAS 2050 and rationale

BSI, 2012. PAS 2050-1:2012: Assessment of life cycle greenhouse gas emissions from horticultural products - Supplementary requirements for the cradle to gate stages of GHG assessments of horticultural products undertaken in accordance with PAS 2050, British Standards Institute. London, England

Intergovernmental Panel on Climate Change (IPCC), 2007. Climate Change 2007 Synthesis Report.

Intergovernmental Panel on Climate Change (IPCC), 2006. Guidelines for National Greenhouse Gas Inventories. Published by the Institute for Global Environmental Strategies (IGES) for the IPCC. ISBN 4-88788-032-4

Intergovernmental Panel on Climate Change (IPCC), 2007. Climate Change 2007 Synthesis Report.

Kuikmann, P.J., van der Hoek K.W., Smit, A., Zwart, K., 2006. Update of emission factors for nitrous oxide from agricultural soils on the basis of measurements in the Netherlands. Alterra report 1217. 40 pp.

Schaefer, F., M.M. Blanke, 2012. Farming and marketing system affects carbon and water footprint - a case study using Hokaido pumpkin. J. Cleaner Production 28, 113-119.

Sommer, K., Scherer, H.W., 2009. Source/Sink-Relationships in plants as depending on ammonium as “CULTAN”, nitrate or urea as available nitrogen fertilisers. Acta Horticulturae 835, 65-88.

Weiske, A., Benckiser, G., Herbert, T., Ottow, J.C.G., 2001. Influence of the nitrification inhibitor 3,4-dimethylpyrazole phosphate (DMPP) in comparison to dicyandiamide (DCD) on nitrous oxide emissions, carbon dioxide fluxes and methane oxidation during 3 years of repeated application in field experiments. Biology and Fertility of Soils 34, 109-117.

Stoessel, F., Juraske, R., Pfister, S., Hellweg, S., 2012. Life cycle inventory and carbon and water foodprint of fruits and vegetables: Application to a Swiss retailer. Environ. Sci. Technol. 46, 3253-3262.

Environmental and nutritional assessment of Breton pâté production

Anne Himeno, Lori Gustavus, Ricardo Teixeira*

Bluehorse Associates, 12 rue Soyer, 92200 Neuilly-sur-Seine, France
* Corresponding author. E-mail: *ricardo.teixeira@bluehorseassociates.com*

ABSTRACT

As part of its commitment to corporate social responsibility, the Jean Hénaff company, a France-based pork meat processor, partnered with Bluehorse Associates and a leading French engineering school Ecole Centrale Paris to conduct a study to measure the carbon footprint of its country-style pâté line, which contains pork meat produced with different agricultural methods (conventional, organic, high-quality labelled meat, and meat from linseed-fed pigs) and various packaging materials. The study was done with the lifecycle assessment application Carbonostics and included nutritional data analysis. the analysis identified pork meat as the first source of carbon emissions with differences linked to the recipe. The second most important source of emissions was the processing stage. In addition, the results revealed that the Hénaff "country-style" pâtés had a carbon footprint lower than international benchmarks. The study included recommendations for the elaboration of new recipes, for marketing and for operations improvement.

Keywords: life cycle assessment, pâté, pork, carbon emissions, nutrition

1. Introduction

Product environmental footprint, and particularly product carbon footprint (PCF), is now driving sustainable business strategy. PCF is particularly important for food and agriculture companies since food products have high impacts throughout their lifecycle, particularly during the agricultural stage. As the true impact of these products is increasingly disclosed, consumers are empowered to make informed choices. Ensuring sustainability is today as important as ensuring nutritional quality of the product. This is a new reality to which companies must adapt.

The company Jean Hénaff (http://www.Hénaff.com/) has established itself over more than one hundred years of existence as a reference in southern Finistère, Brittany (western coast of France). The company is specialised in pork food products, most importantly pâté. For the past several years, Jean Hénaff has focused on its corporate social responsibility through its sustainability policy and via its local engagement with suppliers and commitment to nutrition and health. Hénaff was France's first SME and the first Breton company to have signed the voluntary nutritional progress commitment charter with the French Ministry of Health in November 2010.

In 2008 Hénaff concluded a Bilan Carbone, an inventory of Greenhouse Gas (GHG) emissions generated by all of the company's activities (ADEME, 2006). Following this initial company-level inventory, Hénaff wished to conduct a more in-depth analysis at the product level for their flagship line of pâtés. In line with its strong health and nutrition commitment, Hénaff wished to incorporate the nutritional aspect in parallel to GHG emissions into this study (Hénaff, 2012).

A project in partnership with Bluehorse Associates and Ecole Centrale Paris was thus initiated in 2011 to assess the carbon footprint of its line of country-style pâté products, integrating the nutritional profile into the analysis and recommendations.

2. Methods

2.1. Goals of the study

Besides the carbon footprint assessment of the considered products, the goals of this study for Jean Hénaff were:

- To identify the main sources of emissions for each product in its range of country-style pâtés, throughout their life cycle;
- To compare country-style pâtés together (impact of meat type, recipe, packaging, etc.) taking nutritional quality into consideration;
- To identify potential avenues for carbon footprint improvement.

2.2. Products analysed

First, the number and type of products to analyse was determined. Considering the objectives Hénaff posed for this study, the study targeted nine pork-based products, with net weights of between 78 and 200 g, distinguished by recipe, type of pork meat (conventional, organic and other quality certifications), and type of packaging.

We considered four meat types distinguished by the agricultural circuit from which they are obtained:

- Conventional agriculture: use of synthetic fertilisers and pesticides for the production of feeds is allowed, as is the use of standard antibiotics for pigs;
- Label Rouge: approved production that conforms to specifications for the production of superior quality meat according to a specific recipe;
- Organic farming: no synthetic fertilisers or pesticides, strict restriction of the use of conventional veterinary medicine;
- Bleu-Blanc-Cœur: pigs are fed a finishing phase feed with traditional and high-Omega 3 plant sources (grass, linseed, alfalfa, lupin, etc.).
- We considered also the following 3 packaging types:
- Tin can;
- Aluminium can;
- Glass jar with tin plate lid.

2.3. Scope and functional unit

Since the objective was to find hotspots throughout the supply chain, we chose a Life Cycle Assessment (LCA) methodology. The life cycle considered for pâté production is shown in Figure 1. The scope of the study included complete product life-cycle, from the production of raw materials in the agricultural phase (including the production of animal feed), all the raw material and finished product transport phases, the pâté production processes, consumption and end of life. The functional unit studied was a 78 g-equivalent pâté product eaten, with refrigeration for 24 hrs in the consumer's home included for products in excess of 180 g.

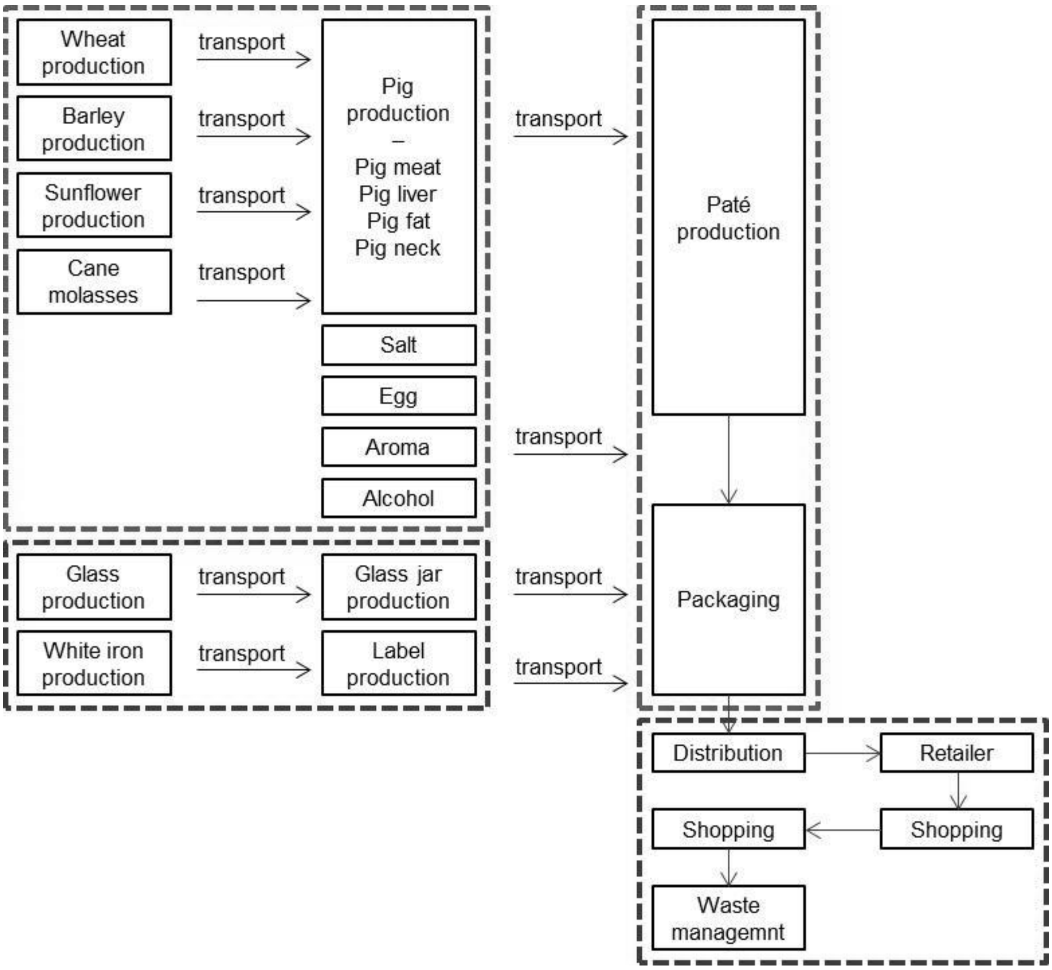


Figure 1. Representation of the life cycle studied.

2.4. Software and data used

The study was conducted by Bluehorse Associates in collaboration with Ecole Centrale Paris.

Primary data for the recipe used, packaging composition, processing energy consumption and transport was primarily obtained from the Bilan Carbone report or, where it was missing, collected by Jean Hénaff and its suppliers.

For GHG emission data, the Carbonostics (2012) online food LCA application was used. Carbonostics is a lifecycle assessment tool designed to pinpoint the hotspots of any food product or menu along three key criteria: cost + carbon + nutrition. Carbonostics' built-in database includes GHG emission factors to assess all the life cycle stages, as well as nutritional information on ingredients. Using Carbonostics enabled the integration of GHG and six nutritional indicators, namely calories, protein, lipids, carbohydrates, sodium and sugar - all in the same analysis.

Local data was also collected for the life cycle stages with higher influence in final results, namely the agricultural production phase. Basset-Mens and van der Werf (2005) provide GHG emissions factors at the farm outlet for pigs reared in Brittany in three agricultural circuits: conventional (2.30 kg CO_2e/kg live pig), Label Rouge (3.46 kg CO_2e/kg live pig) and organic (3.97 kg CO_2e/kg live pig). We then used local study (Chevillon et al., 2011) to allocate GHG emissions for rearing pigs amongst the different parts of the carcass (liver, throat, breast, skin, etc.). We used mass allocation, in line with the BPX 30-323 guidelines (AFNOR-ADEME, 2009), the reference frame for the experimentation on the environmental display of consumer products conducted in France from mid-2011 to the end of 2012.

Amongst the hypotheses used, the following should be noted:

- The emission factors for Bleu-Blanc-Cœur (BBC) pork are the same as those for conventional meat. Indeed, the specific feeding phase is of relatively short duration (2 months) and the proportion of high-Omega 3 plants in the food ration is low, circa 2%.
- Emissions ascribed to secondary and tertiary packaging are negligible when compared to those of the product. They were thus not included in the study.

Finally, a sensitivity analysis (study of the impact of modulating different input variables on end result variation) was conducted and the main results are presented in the following section. As part of the sensitivity analysis we also changed the procedure for allocating emissions to pork parts from mass to economic.

3. Results

3.1. GHG emissions for each pâté line

An initial observation is that the carbon footprint of pâté products is between 160 g and 260 g CO_2e for 78 g (Figure 2), i.e. between 200 and 330 g CO_2e for 100 g, which is below available international benchmarks. This can be attributed to the fact that the GHG emissions for pork production in Basset-Mens and van der Werf (2005) were lower than the equivalent emission factors for other countries in the Carbonostics (2012) database. Moreover, the organic country-style pâté has the highest carbon footprint, while Label Rouge and conventional/Bleu-Blanc-Cœur pâtés have a comparable footprint. This result is explained not only because organic pork production in Bretagne has higher emissions (Basset-Mens and van der Werf, 2005), but also because the proportions of the different pork cuts in the recipe vary.

An analysis of results per life-cycle phase for GHG shows that raw materials are responsible for over 80% of the total impact. Pork is the main hotspot amongst raw materials. The contribution of packaging to emissions reveals that glass has the greatest impact, followed by tin plate and aluminium. Although packaging emissions are not comparable to pork emissions, the difference between packaging materials is sufficient to differentiate between pâtés. Energy spent for processing, the only life cycle step that Hénaff controls directly, lag far behind with only 10% of the total impact. The transport-related impact appears negligible. These percentages are consistent with the Bilan Carbone findings for the bundle of Hénaff products.

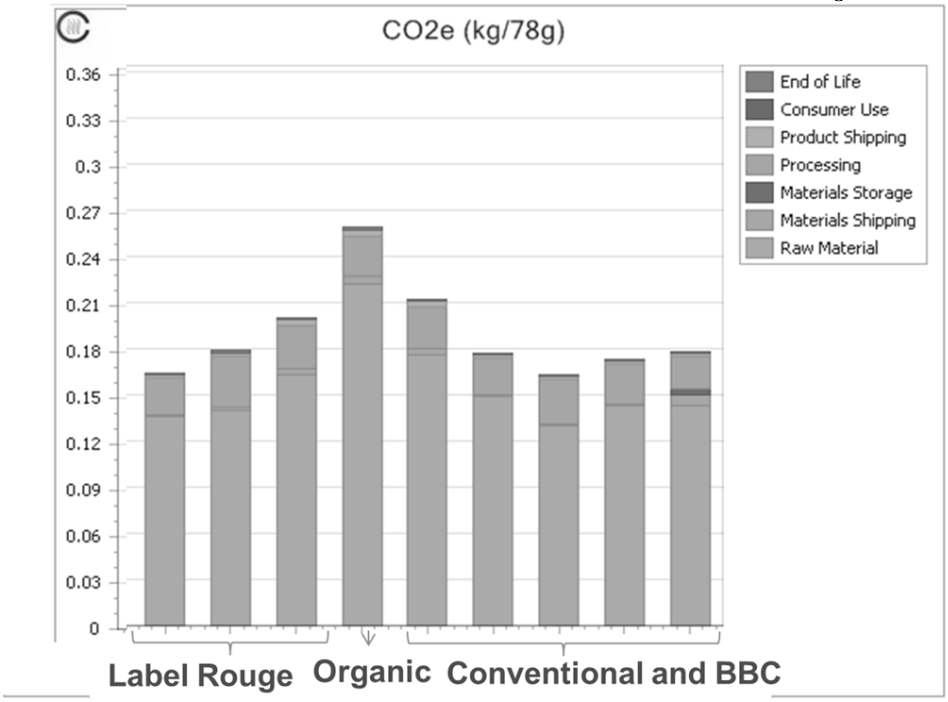


Figure 2. Results per life-cycle phase (pork production impacts allocated by mass to pork products). Raw material production is the hotspot (bar on the bottom). Second highest bar is for processing. BBC - Bleu-Blanc-Cœur pâté.

3.2. Combined analysis of GHG emissions and nutrition

Label Rouge pâtés cause higher emissions than conventional pâtés, despite providing less calories (Figure 3) and more protein (results not shown in this paper). Organic pâté falls between the two on the calories scale and has a carbon footprint similar to Label Rouge. These nutritional variations are mainly explained by recipe variations. This means that even if we had chosen a different functional unit (nutritional units instead of mass units), results would change, and Label Rouge would be the most beneficial.

Results obtained in Carbonostics for the nutritional indicators were validated using laboratorial measurements.

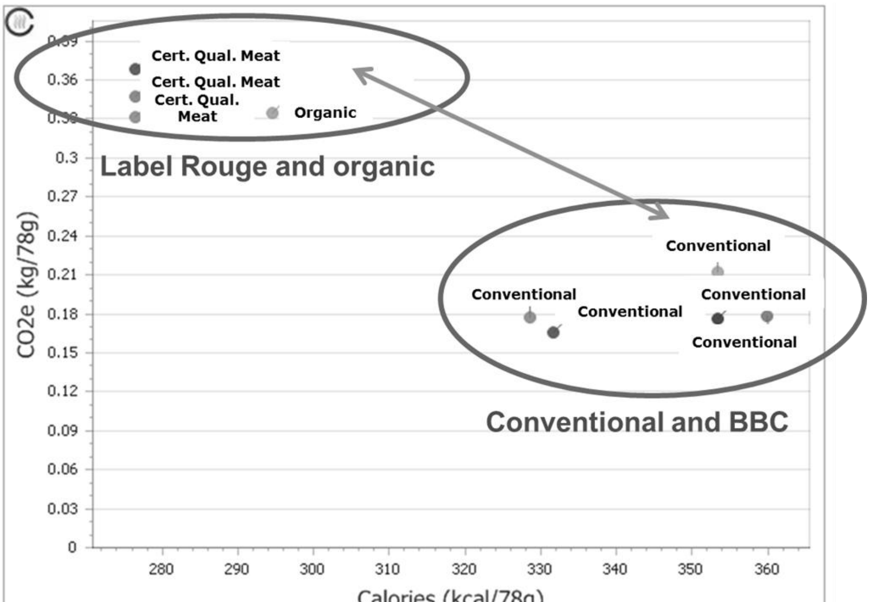


Figure 3. Results for carbon emissions vs. calories. Two clusters of products are shown – Label Rouge and organic pâtés on the left, conventional and Bleu-Blanc-Cœur pâté (BBC) on the right.

3.3. Sensitivity analysis

The following simulations were performed in the context of the sensitivity analysis. We list next the analysis made and the changes in results.

- Economic allocation of emissions for pork parts (instead of mass) - the magnitude of the diference in emissions between organic, Label Rouge and conventional pâtés increases very significantly (more than 10%), but the relative positioning and the qualitative results regarding hotspots remain the same.
- Economic allocation of energy consumption at the pâté production site – Hénaff measured the energy consumption of all machinery involved in the production of the pâté and then physically allocated the energy step by step to each unit produced. As an alternative, quicker route, we allocated the total energy bill to each pâté produced using economic allocation, ie, attributing a fraction of total energy consumed at the production site to each unit of product according to its contribution to total turnover. Overall, there was no signifficant difference in results (less than 1%).
- Using foreign emission factors for pork parts – we used other emission factors from the Carbonostics database, which correspond to records in international databases for different pork parts. For example, we used an emission factor from the LCA Food Denmark database (Nielsen et al., 2003) for pork neck, and a CLM (2010) database record for tenderloin. The final carbon footprint of all pâtés increased. However, the relative results did not change.
- Considering freezing of certain meat parts instead of refrigeration, and considering the impacts from the slaughtering process, not included at first due to low quality data: there was no significant impact in total emissions and no change in relative results.

4. Discussion

This work produced two types of results. For Jean Hénaff, results had operational significance. For Bluehorse Associates and Ecole Centrale Paris, the results were methodological very interesting.

Regarding operational results, Hénaff discovered that their pâtés have lower emissions than international benchmarks. This could be used as an export marketing strategy in particular to the United States, where Hénaff is already the only French meat manufacturer to be USDA-certified.

Hénaff also learned that the ingredients, and mainly pork, are the leading GHG source of emissions. The recipe, and more specifically the proportions of the various pork cuts used, have a significant influence on carbon footprint and of course on the nutritional profile. These choices are made primarily according to the desired organoleptic profile of the end product: its taste, texture, etc.

Equally important to Hénaff is to know where not to focus efforts. Transport, cold storage of ingredients and slaughterhouse processes did not have a significant impact for any of the 9 varieties of country-style pâté products studied. They thus do not appear to constitute an interesting avenue for achieving a rapid and significant reduction of the carbon footprint of country-style pâtés. This study thus raised questions concerning certain preconceived notions of the relative importance of the various sources of emissions (e.g., “food-miles”).

To effectively reduce the carbon footprint of its products, while maintaining or improving their nutritional properties, Jean Hénaff must therefore consider optimizing its choices in terms of recipe, type of meat and packaging, while pursuing its nutritional commitments. Hénaff now has a platform that allows the company to combine these different angles. Naturally, since this study was a first step using only one environmental indicator, namely carbon emissions, other indicators can be used in the future to draw additional conclusions.

For Bluehorse Associates and Ecole Centrale Paris, this difference in results when choosing different functional units was very important. We discovered that if the reference flow is unit of nutritional indicator (e.g. calories, protein, etc.) instead of mass of product, results can be inverted. This fact highlights the difficulty of choosing a functional unit for studies on food products. The function of a food product is to provide quality nutrition. Since there are many different nutritional indicators, LCA practitioners normally use simple comparisons between mass amounts. This may lead to biased results.

Pork meat was particularly challenging as a case study also because the choice of allocation method for pork parts has a dramatic effect on absolute results. Pâtés use parts such as pork neck and fat, which share the majority of the pork production impacts. We used economic and mass allocation, and results changed significantly (more than 10%). Some international standards like PAS2050 (BSI, 2010) in the UK recommend economic allocation, while others like BPX30-323 (AFNOR-ADEME, 2009) and the GHG Protocol (WRI/WBCSD, 2011) recommend mass allocation. This discrepancy is a strong limitation to inter-study comparability.

5. Conclusion

A life-cycle approach taking into consideration all life-cycle steps can identify the main sources of emissions and eliminate certain preconceived ideas concerning the relative significance of transport, packaging or processes, or concerning the compared impact of ingredients obtained from organic farming. The present studied displayed the power of LCA to improve business management. Besides the knowledge of the carbon footprint of its country-style pâtés, along with the main sources of emissions, Hénaff was empowered by this study to make educated decisions about the company's sustainability agenda and how to implement it when designing new products. Hénaff can also use the attributes discovered in the study, like the fact that the emissions per 100g of pâté are lower than international benchmarks, for marketing purposes, in particular for foreign sales of its top-of-the-line products. Finally, Hénaff can now use this precise consumption data per unit of product, delivered in the Carbonostics results, which could be leveraged to optimize its manufacturing processes.

The study also raised interesting methodological points that should be addressed by the LCA community. The dramatic changes in conclusions depending on the functional unit chosen are one example, as is the importance of the impact allocation procedure. For food products, environmental aspects should never be covered separately from nutrition. A combined analysis of carbon-related and nutritional aspects provides more extensive and reliable information.

6. References

ADEME, 2006. Bilan Carbone – Bilan des Emissions de Gaz a Effet de Serre d'une Entreprise Industrielle ou Tertiaire ("Bilan Carbone" – Bill of Greenhouse Gas Emissions of an Industrial or Tertiary-sector company", in English). ADEME, Paris.

AFNOR-ADEME, 2009. BP X 30-323: General principles for an environmental communication on mass market products. AFNOR, Paris.

Basset-Mens, C. and van der Werf, H.M.G., 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. Agr. Ecos. & Env. 105, 127–144.

BSI, 2010. PAS 2050:2010: Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution, London.

Carbonostics. 2012. http://www.carbonostics.com/. Accessed March 2012.

Chevillon, P., Nassy, G. and Gault, E., 2011. Impact carbone du maillon Abattage Déoupe sur la filière porcine ("Carbon emissions of linked slaughtering-dismantling on the pork industry", in English). IFIP (Insitut du Porc), Paris.

CLM. 2010. http://www.clm.nl/index_eng.php?id=methodology. Accessed January 2010.

Hénaff, 2012. http://www.henaff.com/fr/entreprise-responsable/nutrition-et-sante/. Assessed March 2012.

Nielsen, P.H., Nielsen, A.M., Weidema, B.P., Dalgaard R. and Halberg, N., 2003. LCA data on basic food products produced and consumed in Denmark. Danish Institute of Agriculture Sciences. Available from: http://www.lcafood.dk.

WRI/WBCSD, 2011. The Greenhouse Gas Protocol: Product Life Cycle Accounting & Reporting Standard. World Resources Institute/World Business Council for Sustainable Development, Washington DC.

8th Int. Conference on LCA in the Agri-Food Sector, 1-4 Oct 2012

Estimating carbon footprints of individual crops in organic arable crop rotations

Marie Trydeman Knudsen[1*], John E. Hermansen[1], Jørgen E. Olesen[1], Andreas Meyer-Aurich[2], Ngonidzashe Chirinda[1]

[1] Dept. of Agroecology, Aarhus University, Blichers Allé 20, Postbox 50, DK-8830 Tjele, Denmark
[2] Dept. of Technology Assessment and Substance Cycles,Leibniz-Institute for Agricultural Engineerign, 14469 Potsdam, Germany
[*] Corresponding author. E-mail: *MarieT.Knudsen@agrsci.dk*

ABSTRACT

Organic agriculture relies to a high degree on crop rotations, which interlinks the environmental impact of the crops. This constitutes a challenge when using LCA for analysing organic products, specifically with regard to allocation aspects. This was studied in an organic arable crop rotation experiment grown at three different locations in Denmark for three years. The aim was to compare the carbon footprint of the crops at farm gate in four different crop rotations. The carbon footprints were estimated based on either a) the full crop rotation or b) the individual crops. The last approach was done by splitting the environmental burden and benefits from e.g. the green manure equally among the crops on a per hectare basis. The study highlights the importance of analysing the whole crop rotation and including soil carbon stock changes when estimating carbon footprints of organic crops where green manure crops are included.

Keywords: life cycle assessment, greenhouse gas emissions, organic, crop rotation

1. Introduction

Organic agriculture relies to a high degree on recycling of nutrients and using crop residues as a means to fertilise crops and maintain soil fertility. Thus, the basket of products that originates from organic production is diverse and to a high degree interlinked. This constitutes a challenge when using life cycle assessment (LCA) for analysing organic agricultural products derived from the more complex systems. For example with regard to allocation aspects when green manure crops or catch crops are included in the crop rotation. Thus, while ISO standards for LCA (ISO 14040 and 14044) provide overall guidelines, there is still a need for further development of LCA methodologies for complex systems such as organic agriculture. Furthermore, the need to include soil carbon changes in the LCA of organic products is more urgent since organic farming practices on average increase soil carbon sequestration (FAO, 2011).

The aim of the present paper is to explore and suggest a method to estimate the carbon footprints of individual crops in organic arable crop rotations.

2. Methods

The challenge of estimating carbon footprints of organic crops was studied in an organic arable crop rotation experiment grown at three different locations in Denmark for three years (2006-8). The organic crop rotations were designed to explore ways to avoid the use of conventional manure. Organic arable production is often in practice dependent on conventional manure, which is considered problematic by parts of the organic sector in e.g. Denmark. Four different crop rotations were compared in the experiment in order to provide alternative solutions to secure nitrogen (N) supply for the crops without using conventional manure.

2.1 The crop rotations

The four scenarios are illustrated in Figure 1; three organic ('Slurry', 'No input' and 'Mulching') and one conventional crop rotation. The first organic scenario (Slurry) represents the present situation where slurry (often conventional) is imported and used on arable farms and all four crops in the crop rotation can be sold as sales crops. The second organic rotation (No input) represents a scenario where still all four crops can be sold as sales crops, but no organic fertiliser is used. The third organic scenario (Mulching) represents a solution where the faba beans are replaced by a green manure crop (grass-clover) that is incorporated in the soil and thus only three sales crops can be sold from the rotation. Furthermore, a conventional rotation was included that is similar to the 'Slurry' rotation, where all four crops can be sold as sales crops, but mineral fertiliser is used instead of slurry and in addition pesticides are used. Where possible, catch crops were included in the rotations (Figure 1).

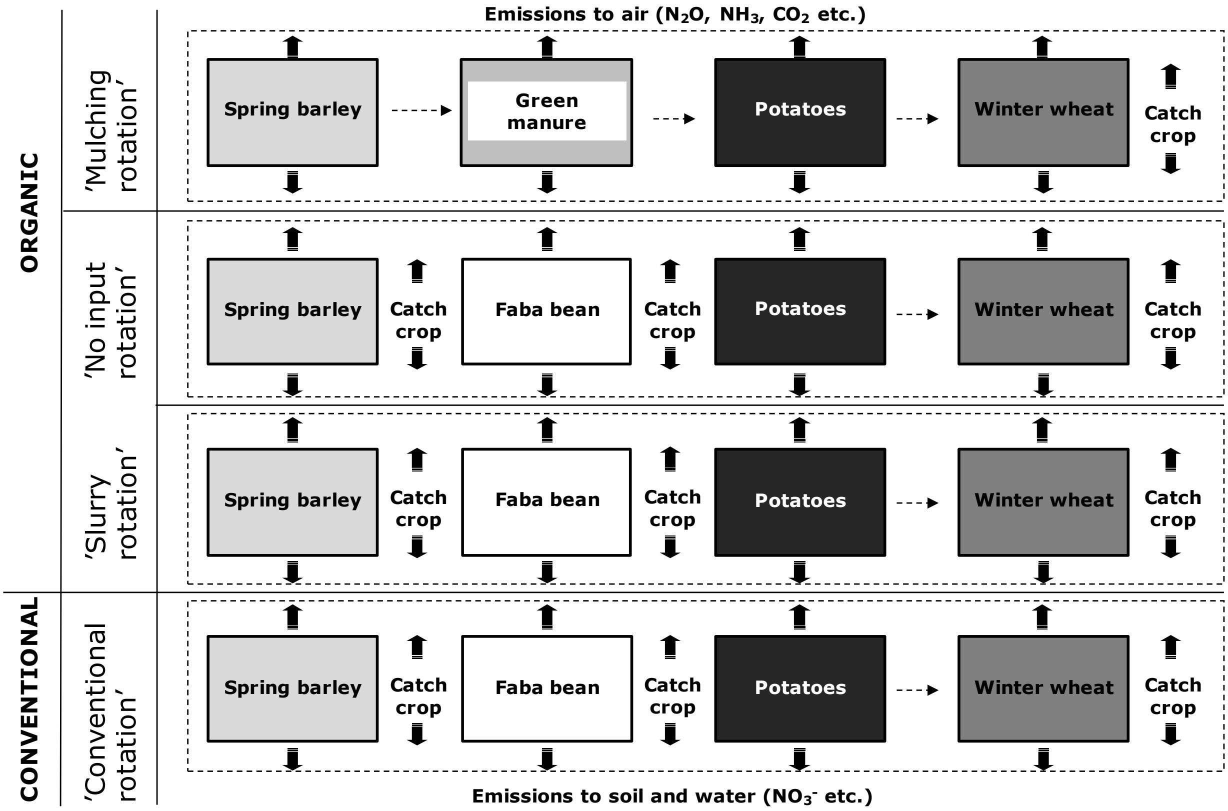


Figure 1. Illustration of the crop rotations analysed in the study. The studied crop rotations were grown at three locations in a randomised and replicate long-term experiment in Denmark (Jyndevad, Foulum and Flakkebjerg) during three years (2006-8).

2.1 Estimation of the carbon footprint in a life cycle assessment approach

A life cycle assessment (LCA) approach until farm gate was used in the present study focusing on greenhouse gas emissions. The carbon footprints are expressed in g CO_2 eq. per kg dry matter (DM) of the total sales crops or the specific crops. Thus, the functional unit is one kg crop DM.

The system boundaries in the studied systems included mainly the production of agricultural inputs and the agricultural production in the field, including soil carbon stock changes (Fig. 2).

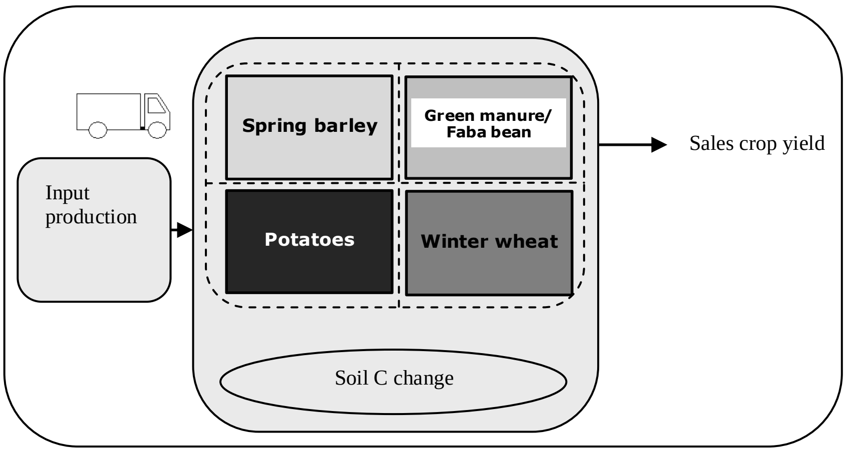


Figure 2. Illustration of the processes involved and system boundaries in the present study of crop rotations. The carbon footprints are calculated based on both the full crop rotation and separately for the specific crops.

The data are primarily based on the field experiment for the five treatments, two blocks, three locations and three years. Crop DM and nitrogen (N) yields were measured in the field experiment for both grain and straw. For the catch crops, total DM and N yield were measured. The above and below ground crop residues were estimated according to IPCC guidelines 2006 (IPCC, 2006). The N content in applied manure was measured in the experiment. The diesel and energy consumption was based on the operations in the experiment. GHG emissions related to diesel and energy consumption was based on data from the Ecoinvent database (Ecoinvent Centre, 2009).

Direct and indirect emissions of nitrous oxide (N_2O) were estimated according to the IPCC guidelines 2006 (IPCC, 2006). N leaching was based on measurements in the experiment as reported by Askegaard et al., (2011). The greenhouse gas emissions and characterisation factors were based on IPCC 2007 standards (IPCC, 2007).

2.2 Allocation of environmental burden in crop rotations and soil C change

At the agricultural production stage, the crops are interlinked in the organic crop rotations since the yields of the sales crops depend on the recycling of nitrogen (and carbon) from the green manure crop through its effects on soil fertility. Thus, the environmental impacts from the green manure crops need to be included in the carbon footprint of the sales crops. Two approaches are used in the present paper:

A) Assessment of the full crop rotation: The full crop rotation was assessed as a ‘black box’, where the emissions where based on all the inputs used for the crop rotation and related to the total sales crop DM yield from the crop rotation. This approach is represented by the dotted line surrounding the full crop rotation in Figure 2. Thus, the emissions from the green manure crop are included, but the carbon footprint result can only be estimated as an average over the full crop rotation per kg DM.

B) Assessment of single crops: allows to estimate carbon footprint results of the single crops in the crop rotation (e.g. for spring barley). However, in this approach it is assumed that the environmental burdens and benefits from green manure crops (incl. soil carbon changes) and catch crops are divided equally among the other sales crops in the crop rotation on an area basis. The carbon input and N_2O emissions from organic (and mineral) fertilisers were allocated to the specific crops that received the fertiliser. Likewise, carbon input and N_2O emissions from the above and below ground crop residues from the main crops were allocated to the specific main crop. The carbon input and N_2O emissions from the above and below ground crop residues from the catch crops and green manure crops were allocated equally on the crops in the crop rotation on a per hectare basis.

Long-term soil carbon changes were included in the study based on recorded carbon inputs from manure and estimated above and below ground crop residues (based on measured yields and IPCC guidelines 2006 (IPCC, 2006)) from main crops, catch crops and green manure crops. It was assumed that the conventional crop rotation would be the point of departure – and thus the relative soil carbon changes from the conventional crop rotation were estimated in a 100 years perspective. The relative difference relative to the conventional crop rotation in total carbon input was calculated for each organic scenario. It was then assumed that approx. 10% of the surplus of carbon would be sequestered (Petersen et al., 2011, Christensen 1986).

2.3 Statistical analysis

Statistical analyses by Proc Mixed were performed using SAS software. The carbon footprint values for the full crop rotation were estimated for each of the four treatments, three locations, two blocks and three years. The model included treatment and location as main effects and year and blocks as random effects. The significant difference between treatments was estimated using the Tukey’s Studentized range test with Pf 0.05 if a main effect or interaction was significant. The carbon footprint results are shown as an average over the years due to non-significant differences between the three years.

3. Results

The analysis of the full crop rotation showed no significant difference in carbon footprint of sales crops between the different crop rotations (Fig. 3). The higher N_2O emissions from grass-clover in the ‘Mulching’ rotation were counteracted by a higher soil carbon sequestration. The ‘No input’ rotation had lower yields and a negative soil carbon sequestration, which increased the carbon footprint of the sales crop from this rotation (Fig. 3).

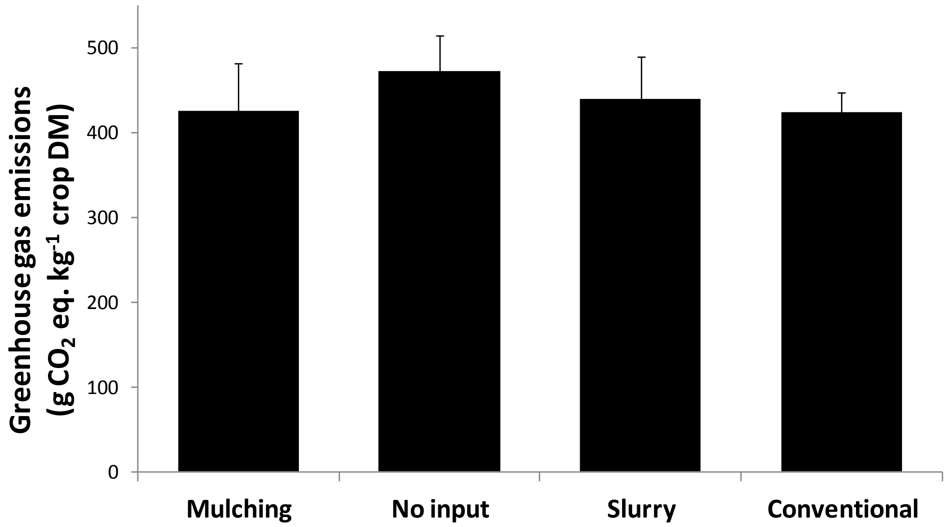


Figure 3. Carbon footprints per kg sales crop DM at farm gate based on the full crop rotations 2006-8. The values are the means over the three years ± S.E.

The analysis of the individual crops provided a specific carbon footprint value of e.g. spring barley grown in the different crop rotations (Figure 4). Figure 4 shows that the carbon footprint values for e.g. spring barley grown in the 'Mulching' rotation was affected by the soil carbon sequestration and the N_2O emissions caused by the green manure crop compared to the 'Slurry' or 'Conventional' rotations. These findings show that the contributions caused by the green manure crop to the carbon footprint of the organic crops are considerable.

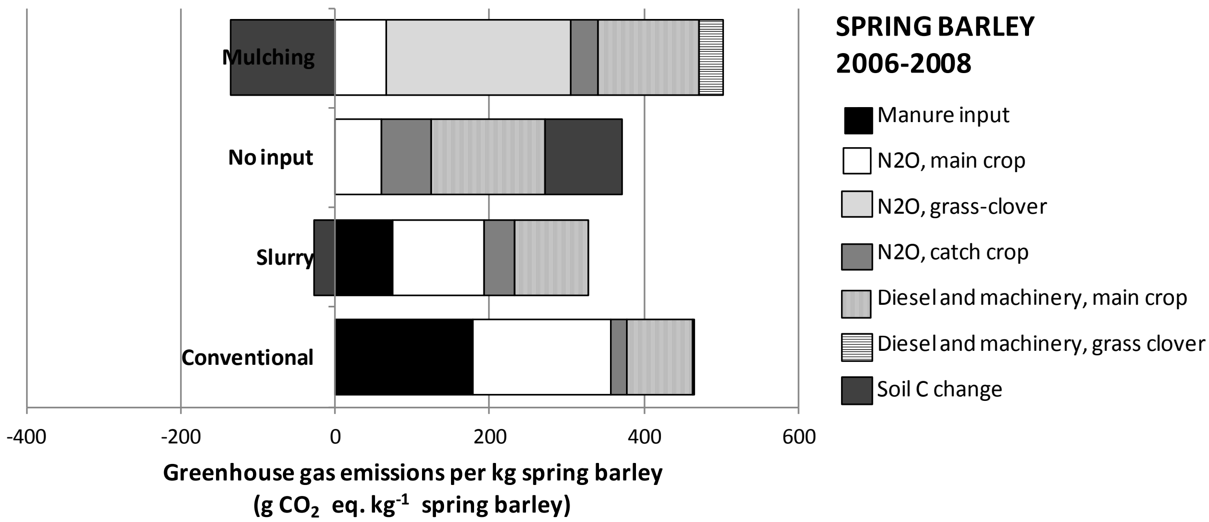


Figure 4. Carbon footprints of spring barley at farm gate in the different crop rotations 2006-8. The values are the means over the three locations and three years (2006-8).

4. Discussion

When calculating the carbon footprint of crops from crop rotations where green manure crops or catch crops are included, there is a need to include the environmental burdens and benefits from the green manure or catch crops and thus include the full crop rotation in the assessment.

One solution, as suggested here, is to assess the full crop rotation as a 'black box' based on kg sales crop DM from the different crops. The analysis over the whole rotation can be used to judge on different rotation designs. The evaluation could be based on kg DM, as suggested here or be related to the functions of agriculture as discussed by Nemecek et al., (2011). Another option is to use a functional unit that reflects the dietary needs by humans as discussed by Smedman et al., (2010).

However, there is a need for carbon footprint values for the specific crops. Furthermore, the full crop rotation implies several crop specific inputs and several outputs i.e. different sales crops from the rotation, which should be included in the assessment since these inputs might be very different between crops. Thus, the environmental burdens and benefits need to be divided and allocated to the specific crops. According to the ISO standards for LCA (ISO 14040 and 14044) allocation '...wherever possible should be avoided by dividing the unit process in two or more sub processes...'. In a conventional crop rotation, this is often no problem and the analysis can be focused on the crop specific inputs and outputs i.e. crop yields. Likewise, in simple organic rotations without green manure and catch crops (Knudsen et al., 2010) or organic perennial crops, that do not imply other crops (Knudsen et al., 2011), it is possible to focus on the specific crop. The advantage of this approach is that the crop-specific inputs will be reflected in the carbon footprint of the crop, such as e.g. high diesel consumption in the potatoes. If it is not possible to subdivide the unit process, i.e. the crop rotation, and economic or mass allocation is performed to allocate the environmental burden among the different sales crops, the crop specific inputs would not be reflected. This would mean that e.g. high diesel consumption in the potatoes, would also burden the e.g. spring barley that happens to be grown in the same crop rotation.

Therefore, in order to calculate carbon footprint values for the specific crops, we recommend subdividing the crop rotation in the specific crop productions. The challenge is then how to divide the environmental burdens and benefits from the green manure and catch crops. This could be done either on a per hectare basis or based on the N residual effect. The last approach would have the theoretical advantage that the crop following the green manure crop, here: potatoes, which might benefit more from the N in the green manure residues would also bear a higher share of the burden. However, the disadvantage is that exact numbers of the residual effect are needed and those would differ from study to study and increase the uncertainty. Furthermore, the beneficial residual effect is not only caused by N but also carbon (C) with an increased soil carbon level, which affects all of the crops in the crop rotation equally. Therefore, we recommend dividing the environmental burdens and benefits from these green manure and catch crops equally on the hectares used for sales crops in the crop rotation.

The study shows that both the carbon footprint results based on the full crop rotation and the results for the specific crops would had been very different if the environmental burden from the green manure and catch crops and the soil carbon changes had not been included.

5. Conclusion

In conclusion, the study highlights the importance of analysing the whole crop rotation and including soil carbon changes when estimating carbon footprints of organic crops especially where green manure crops are included. Two methods were presented, which enable an integration of green manure crops and catch crops in the overall analysis. The analysis over the whole rotation should be used to judge on different rotation designs. In order to calculate carbon footprint values for the specific crops, it is recommended to include the environmental impacts from green manure and catch crops equally on the hectares used for sales crops in the crop rotation.

6. References

Askegaard M, Olesen JE, Rasmussen IA, Kristensen K (2011) Nitrate leaching from organic arable crop rotations is mostly determined by autumn field management. Agriculture, Ecosystems & Environment 142(3–4): 149–160

Christensen BT (1986) Straw incorporation and soil organic matter in macro-aggregates and particle size separates. Journal of Soil Science 37: 125-135

FAO (2011) Organic agriculture and climate change mitigation. A report of the Round Table on Organic Agriculture and Climate Change. December 2011, Rome, Italy.

Ecoinvent Centre. 2009. Ecoinvent Database v.2.0. Swiss Centre for Life Cycle Inventories. Online at: http://www.ecoinvent.org/ .

IPCC (2006) 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel of Climate Change (IPCC), National Greenhouse Gas Inventories Programme. Online at: http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html

IPCC (2007) Climate Change 2007: the physical science basis. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.) Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA 996 p.

Knudsen MT, Yu-Hui Q, Yan L, Halberg N (2010) Environmental assessment of organic soybean (Glycine max.) imported from China to Denmark: a case study. Journal of Cleaner Production 18:1431–1439

Knudsen MT, de Almeida GF, Langer V, de Abreu LS and Halberg N. 2011. Environmental assessment of organic juice imported to Denmark: a case study on oranges (Citrus sinensis) from Brazil. Organic Agriculture 1: 167-185

Nemecek T, Dubois D, Huguenin-Elie O, Gaillard G (2011) Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agricultural Systems 104: 217-232

Petersen BM, Knudsen MT, Hermansen JE, Halberg N (2011) A methodological approach to include soil carbon changes in life cycle assessments. In Knudsen (2011) Environmental assessment of imported organic products – focusing on orange juice from Brazil and soybeans from China. ISBN 978-87-7611-408-4.

Smedman A, Lindmark-Månsson H, Drewnowski A, Edman A-KM (2010) Nutrient density of beverages in relation to climate impact. Food and Nutritional Research, 54: 5170. DOI: 10.3402/fnr.v54i0.5170.

The carbon footprint of Brazilian canary melon

Maria Cléa Brito de Figueirêdo[1,*], Imke J.M. de Boer[2], Carolien Kroeze[3], José Potting[3], Viviane da Silva Barros[1], Fernando Aragão[1], Rubens Sonsol Gondim[1]

[1] Embrapa Tropical Agroindustry, Ceará, Brazil
[2] Animal Production Systems Group, Wageningen University, the Netherlands
[3] Environmental System Analysis Group, Wageningen University, the Netherlands
[*] Corresponding author. E-mail: *mariacleabrito@gmail.com*

ABSTRACT

Emission of greenhouse gases (GHG) is an environmental concern that has been the focus of food producers and consumers worldwide. The objective of this study, therefore, is to quantify GHG emissions along the production chain of Brazilian melon, and, to identify improvement options. The product system encompasses (i) upstream processes, i.e. the production and transport of inputs, such as seeds, pesticides, diesel and plastics, (ii) melon processes in the Low Jaguaribe and Açu region, i.e. the production of seedlings, melons and packing, and (iii) downstream processes, i.e. transport of melons to Europe and solid waste disposal. The functional unit is one tonne of exported canary melon. Primary data related to inputs and solid waste generation was obtained from farmers and researchers at the Brazilian Agriculture Research Corporation (Embrapa) - Tropical Agroindustry branch, through a structured questionnaire, during the first semester of 2011. The emissions of carbon dioxide, nitrous oxide and methane are estimated from the amount of input used in different activities, applying emission factors proposed by IPCC and Brazilian GHG Inventories. GHG emissions from melon processes in the Low Jaguaribe and Açu region are considered, regarding the following activities: land use change (biomass loss, with cutting and burning, and soil organic matter mineralisation); nitrogen fertilisation (including incorporation of field residues in soils); and fossil fuel combustion by tractors. The impact on climate change is calculated in CO_2-equivalentes, considering the global warming potential of GHG in a time period of 100 years. GHG emissions from upstream and downstream processes are mainly taken from the Ecoinvent database. Considering all processes and the uncertainty in measurements, the total impact on climate change of canary melon, up to their distribution in the European market, can reach 710 kg CO_2-eq/t melon. Indirect emissions, from the production of inputs and transports, exceeded direct emissions from melon processes located in the Low Jaguaribe and Açu region. The major melon process in this region contributing to GHG emission is crop production, whereas the major upstream and downstream processes were fertiliser and plastic production. Moreover, scenario analyses showed that the carbon footprint of canary melon can be reduced by 24% if melon fields are located in former agricultural areas and by 6% if nitrogen fertilisation is reduced to best practice levels in crop production. The results of this study may support Brazilian melon producers when accounting melon carbon footprint and deciding about which management practices use to reduce it.

Keywords: canary melon, Brazil, carbon footprint, climate change

1. Introduction

Melon (*Cucumis melo* L.) is a cucurbit crop, whose fruit is rich in vitamins, minerals, and has low calorie content. It is produced mostly in tropical regions, and subsequently exported across the world. The high luminosity (about to 3,000 h/year) together with the low precipitation rate (from August to December) and humidity constitute excellent conditions for melon production (Silva and Costa, 2003).

Brazil was the second largest world melon exporter in 2010 (FAO, 2011). The main exporting melon producers in Brazil are clustered in the Low Jaguaribe and Açu region, in the Northeast States of Ceará and Rio Grande do Norte. In 2009, melon production in this region contributed to 99% of the country melon exports (MDIC, 2011). From 1999 to 2009, the area cultivated with melon in this region increased more than 60% (IBGE, 2011).

The type of melon commonly produced and exported belongs to the *Cucumis melo* inodorous Naud group, popularly known as canary yellow melon (Silva and Costa, 2003). Melon production occurs in open fields and relies on drip irrigation and fertirrigation (i.e. application of soluble fertiliser through an irrigation system). Fertirrigation is required because of insignificant rainfall during the production period (from July to December) (Miranda et al., 2008) and a low nutrient content of soils (Crisóstomo et al., 2002).

The release of new requirements related to carbon footprint (CF) certification has raised the attention of Brazilian melon producers. Examples of CF protocols are PAS 2050 (BSI, 2011) and the Product Life Cycle Accounting and Reporting Standard (WRI, WBCSD 2011). These standards consider emissions from processes related to the life cycle of a product, and facilitate identification of improvement options through product value chains. According to Pandey et al., (2011), the CF of a product is defined as the amount of GHGs expressed in terms of CO_2-eq or CO_2-equivalents emitted by that product during its life cycle, with specific system boundaries.

So far, few studies have assessed the CF of melon. Audsley et al., (2009) quantified the CF of melon produced in- and outside Europe in general terms, using proxy values from existing data related to similar food products. Cellura et al., (2012) quantified the CF of Italian melon produced in pavilion and tunnel greenhouses in a Sicilian agricultural district. These studies, however, do not address melon production in tropical countries such as the market-leader Brazil. They did not look into the emission reduction potential of possible improvement options nor analyse the uncertainties in the CF calculations of melons planted in open fields.

Our objectives, therefore, are to assess the CF of Brazilian exported canary melon, considering the uncertainties in the GHG emissions, and to evaluate reduction potentials of improvement options. Results give melon producers insight in the CF of their product, and in potential options to reduce it.

2. Methods

This study is based on life cycle assessment (LCA), according to ISO (ISO 14040 and 14044, 2006a, 2006b), but focuses on climate change.

2.1 System boundary and functional unit

The system boundary includes (i) upstream processes, i.e. the production and transport of inputs (seeds, pesticides, fertilisers, coconut substrate, cleaning materials, diesel, paper and plastics), (ii) melon processes in the Low Jaguaribe and Açu region, i.e. the production of seedlings, melons and packing, and (iii) downstream processes, i.e. transport of melons to Europe and solid waste disposal. Melon distribution by European retailers and the final consumption are outside the scope of this study.

The functional unit is one tonne of exported canary melon. Allocation of inputs and outputs is performed based on the market price of exported and nationally commercialised melons, considering that 99% of total revenue is due to exported melons.

2.2 Data collection

Primary data related to seedling, plant production and melon packing was collected at production units in the Jaguaribe/Açu region, using a structured questionnaire. As seed production occurs outside Brazil, data was collected from an experimental seed greenhouse, maintained by the Brazilian Agriculture Research Corporation (Embrapa). Data from the production of coconut substrate were obtained from Figueirêdo et al., (2009). Data regarding the other processes of the product system was obtained from Ecoinvent (Frischknecht and Jungbluth, 2007). The amount of inputs and solid wastes related to melon processes is presented in Table 1.

Table 1. Primary data related to one tonne of exported melon, obtained from farms and completed with experimental data from a melon seed greenhouse.

Inputs and outputs	Unit per production unit	Seed production	Seedling production	Crop production	Melon packing
Area	m^2	0.30	0.01	441.92	0.52
Seed	g	0.08	33.66	0.00	0.00
Seedling	g	9.03	0.00	2.471.75	0.00
Coconut substrate	g	1,011.11	3,564.00	0.00	0.00
Water	L	0.09	0.06	186.05	0.15
Electricity	kWh	11.49	0.46	72.60	18.15
Diesel	g	0.00	0.00	7,207.20	0.00
Cleaning products	g	0.00	0.00	0.00	648.10
Plastics	g	73.27	519.31	38,008.36	659.01
Papers	g	0.00	0.00	0.00	58,495.80
Wood (pallets)	g	0.00	0.00	0.00	11,965.80
Fertilisers					
Organic compost	g	0.00	0.00	123,684.66	0.00
N	g	4.05	1.65	5,548.72	0.00
P_2O_5	g	0.59	1.65	6,660.24	0.00
K_2O	g	7.47	0.00	9,613.66	0.00
Others	g	3.98	0.00	2,347.80	0.00
Pesticides					
Insecticide	g	1.28	0.01	765.72	0.00
Fungicide	g	0.55	0.02	480.19	2.66
Herbicide	g	0.46	0.00	0.00	0.00
Solid waste					
plastic	g	66.01	523.47	38,008.36	0.00
empty pesticide packages	g	0.16	0.00	643.50	0.31

2.3 Estimation of GHG emissions

To estimate GHG in seed, seedling, melon production and packing, the following activities are included: land use change (i.e. biomass loss from cutting and burning, and soil organic matter mineralisation); nitrogen fertilisation (including incorporation of field residues in soils); and fossil fuel combustion by tractors. These activities may release CO_2, CH_4 and N_2O. These GHG are estimated considering climate and soil characteristics of the Jaguaribe/Açu region, applying emission factors proposed by IPCC (2006). Carbon storage in biomass and soil were according to the Brazilian GHG Inventory (MCT, 2010).

GHG emissions from production of other inputs (agrochemicals, diesel, electricity, substrates and cleaning materials), from transport of inputs and melons, and from final disposal of solid wastes were accounted using the Ecoinvent database (Frischknecht and Jungbluth, 2007). Emissions from the production of coconut substrate were estimated based on Figueirêdo et al., (2009).

2.4 Impact assessment

The CF of each process is quantified considering the mass (kg) of GHG emitted in each activity and their respective GWP, over a timeframe of 100 years, according to IPCC (2007). The CF of canary melon is the sum of the footprints of all processes that integrate this product system applying economic allocation ratios for co-products.

2.5 Scenario and uncertainty analysis

A reference situation is defined to quantify the use of inputs, generation of solid wastes, GHG emissions, and CF of the canary melon. This reference situation has the following characteristics: i) the average amount of nitrogen fertiliser applied in melon production fields is 6 kg N/t melon and ii) savannah (Caatinga) forests previously occupied the area where seedling and melon production currently takes place, as well as where packing houses are now located (i.e. forest cutting and burning occurred just before melon fields were established), less than 20 years ago.

GHG emissions may vary among farms depending on farm practices. We, therefore, performed a sensitivity analysis to test the sensitivity of the CF calculated in the reference situation to variations in the production process. The most important variations in production practices in melon farms that potentially change GHG emissions are related to issues i) to ii), formerly described. These variations are used to define five different scenarios: 1) the amount of nitrogen fertiliser applied in melon fields is 4 kg N/t, as suggested by previous study in the study area (Crisóstomo et al., 2002) and 2) previous land use was agricultural land area.

The CF was calculated for each scenario, considering the average yield of 23 t/ha in crop production. In addition, the uncertainty in each scenario was analysed using Monte Carlo analysis, assuming log normal distributions of probability functions. The Pedigree matrix was used to determine the deviations of each parameter (Goedkoop et al., 2008).

3. Results and discussion

3.1 GHG emissions

The export of one ton of canary melon generates the average total amount of 509 g of CO_2, 1,430 g of CH_4, 482 g of N_2O and 30 g of other GHG (Table 2). Land conversion (from Caatinga vegetation to melon production fields) is a major source of CO_2 emissions in melon processes. Nitrous oxide and methane emissions mainly occur in the production of fertilisers that are applied in melon fields.

The overall emission from upstream and downstream processes is higher than the overall emissions from melon processes at the Jaguaribe and Açu region. Upstream and downstream processes together contribute 63% of CO_2, 85% of CH_4, 55% of N_2O and 100% of all other GHG emissions.

Considering individual processes in the canary melon chain, however, plant production generates the largest emissions of CO_2 (37%). The production of fertilisers is the largest source of CH_4 (71%) and N_2O (53%).

Table 2. Estimated GHG greenhouse of one tonne of exported canary melon in the reference situation.

Processes	CO_2 (g)	%	CH_4 (g)	%	N_2O (g)	%	Other GHG (g)	%
Seedling	3.96	0%	0.01	0%	0.04	0%	0.00	0%
Cropping	187,375.32	37%	215.82	15%	217.80	45%	0.00	0%
Packing	201.96	0%	0.30	0%	0.03	0%	0.00	0%
Total melon processes (Jaguaribe and Açu region)	187,581.24	37%	216.13	15%	217.87	45%	0.00	0%
Seed production	106.92	0%	0.10	0%	0.10	0%	0.00	0%
Transport of melon BR-NL	60,100.17	12%	0.35	0%	1.73	0%	67.67	0%
Paper production	52,058.96	10%	5.78	0%	3.70	1%	109.22	0%
Plastic production	89,301.73	18%	2.39	0%	0.97	0%	496.99	2%
Production of fertilisers	47,570.40	9%	1,013.54	71%	253.68	53%	2,378.72	8%
Production and distribution of electricity	17,885.73	4%	167.78	12%	0.81	0%	16.97	0%
Transport of materials to farms	22,544.48	4%	0.14	0%	0.63	0%	23.32	0%
Other processes	32,269.12	6%	23.98	2%	2.79	1%	27,144.80	90%
Total upstream and down-stream processes	321,837.51	63%	1,214.05	85%	264.42	55%	30,237.70	100%
Total emissions	509,418.75	100%	1,430.18	100%	482.29	100%	30,237.70	100%

3.2 CF in the reference situation and alternative scenarios

The average CF of canary melon in the reference situation is 710 kg CO_2-eq/t exported melon (Figure 1), ranging from 632 to 787 kg CO_2-eq/t melon. Among all processes in the product system, crop production has a relatively large share in the CF in most of the scenarios.

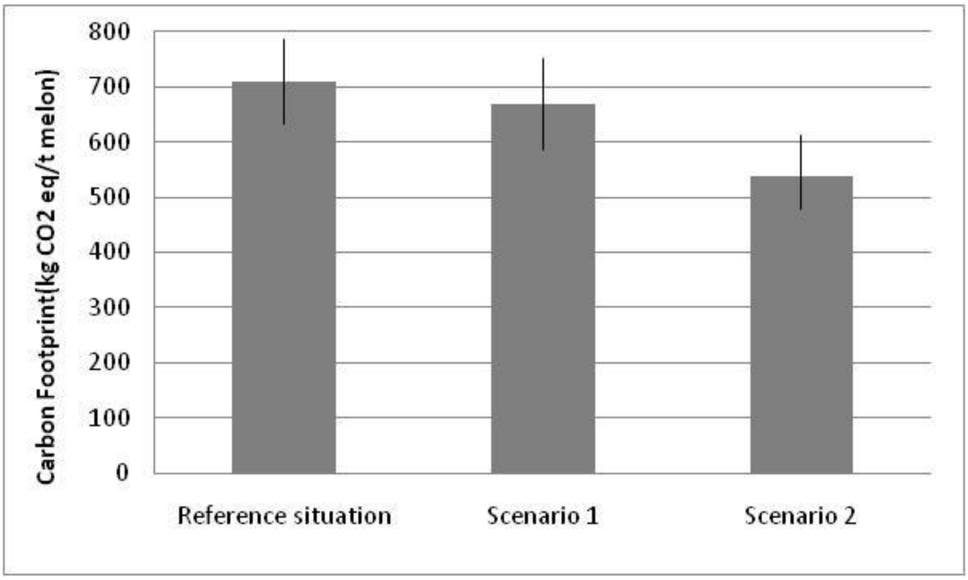


Figure 1. Canary melon CFs by scenario, including estimated uncertainty.

The average CF value is reduced by 6% if the nitrogen application is brought down from 6 to 4 kg N/t of melon (scenario 1). Locating melon processes in the Low Jaguaribe/Açu region in former agricultural areas reduces CF by 24% (scenario 2).

The distances between input production sites and melon fields and packing houses and the transport of melons from Brazil to The Netherlands are considerable. However, in none of the scenarios the GHG emissions from transportation make a large contribution to the CF of canary melon.

Uncertainty regarding CF values mainly concerns variations in carbon fractions of vegetation types and soil, in GHG emission factors and in transport distances. The carbon fraction of Savannah was estimated by MCT (2010), varying according to local physiognomies. The soil carbon content varies according to soil types that may be found in the studied region (low clay activity, high clay activity and sandy soils). Emission factors used to estimate GHG emissions were according to IPCC (2006) and may be two times more or less the average value (factors for nitrous oxide vary from -70% to +300%). Distances from sites where materials and fuels are produced to melon farms and packing houses may also vary from farm to farm. The estimation of these distances considering major exporting countries or national states is just an approximation of the real distances.

The CF result of 710 kg CO_2-eq/t of exported canary melon from the Low Jaguaribe/Açu region in the reference scenario is low compared to results reported in earlier studies (Cellura et al., 2012). The average CF of Italian melon cultivated in greenhouses in southern Sicilia was 1,427 kg CO_2-eq/t melon (Cellura et al., 2012). Main differences in melon production systems refer to the total amount of agrochemicals, plastics (greenhouses) and fuels used by Italian producers.

4. Conclusion

The CF for Brazilian canary melon in the reference situation is 710 kg CO_2-eq/t of exported melon (ranging from 632 to 787 kg CO_2-eq/t melon). Emissions from upstream and downstream of melon processes in the Low Jaguaribe and Açu region have a large share in this total footprint. Direct emissions from seedling, cropping and melon packing contributed less than 50% in all scenarios, including the reference situation. Melon cropping is the main process responsible for direct emissions among the processes in the Low Jaguaribe and Açu region.

Scenario analysis shows that the footprint can be reduced to 539 kg CO_2-eq/t of exported melon. This reduction is achievable when melons are produced on farms located in already existent agricultural areas.

5. References

British Standards (BSI), 2008. PAS 2050:2011 - Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. DEFRA, United Kingdom. Available at: <http://www.ifu.ethz.ch> (accessed 22.09.2010).

Cellura, M., Ardente, F., Longo, S., 2012. From the LCA of food products to the environmental assessment of protected crops districts: a case-study in the South of Italy. Journal of Environmental Management, 93, 194-208.

Crisóstomo, L. A., Santos, A. A., Raij, B., Faria, C. M. B., Silva, D. J., Fernandes, F. A. M., Santos, F. J. S., Crisóstomo, J. R., Freitas, J. A. D., Holanda, J. S., Cardoso, J. W., Costa, N. D., 2002. Adubação, Irrigação, Híbridos e Práticas Culturais para o Meloeiro no Nordeste [Fertilization, irrigation, hybrids and cultural practices to the melon plant in the Brazilian Northeast]. Circular Técnica 14. Embrapa Agroindústria Tropical, Fortaleza.

Figueirêdo, M. A., Rodrigues, G. S., Caldeira-Pires, A., Rosa, M. F., Aragão, F. A. S., Vieira, V. P. P. B., Mota, F. S. B., 2010. Environmental performance evaluation of agro-industrial innovations. Part 1: Ambitec-LifeCycle, a methodological approach for considering life cycle thinking. Journal of Cleaner Production, 18, 1366-1375.

Food and Agriculture Organization (FAO), 2011. FAOSTAT: melon production. FAO, Rome. Available at: <http://faostat.fao.org/site/567/default.aspx#ancor> (accessed 26.09.2011).

Frischknecht, R.; Jungbluth, N. 2007. Ecoinvent – overview and methodology. Swiss Centre for Life Cycle Inventories, Dubendorf.

Goedkoop, M., Schryver, A., Oele, M., 2008. Simapro 7: introduction into LCA. PRé Consultants, Netherlands.

Instituto Brasileiro de Geografia e Estatística (IBGE) [Brazilian Institute for Geography and Statistics], 2009. Produção Agrícola Municipal [National Agriculture Production]. IBGE, Brasília. Available at: <www.ibge.gov.br> (accessed 20.09.2011).

International Organization for Standardization (ISO), 2006a. ISO 14040:2006- Environmental management, Life cycle assessment, Principles and framework. ISO, Geneva.

International Organization for Standardization (ISO), 2006b. ISO 14044:2006- Environmental management - Life cycle assessment - Requirements and guidelines. ISO, Geneva.

International Panel on Climate Change (IPCC), 2007. IPCC Fourth Assessment Report: Climate Change 2007. IPCC, Geneva. Available at: < http://www.ipcc.ch/publications_and_data/ar4/wg1/en/tssts-2-5.html>. (accessed 22.09.2010).

International Panel on Climate Change (IPCC), 2006. Guidelines for National Greenhouse Gas Inventories. IPCC, Geneva. Available at: <http://www.ipcc-nggip.iges.or.jp>. (accessed 22.09.2010).

Ministério da Ciência e Tecnologia (MCT) [Brazilian Science and Technology Ministry], 2010. Inventário Brasileiro de Emissões Antrópicas por Fontes e Remoções por Sumidouros de Gases de Efeito Estufa não Controlados pelo Protocolo de Montreal [Brazilian inventory of emissions and removal of GHG not controlled by the Montreal Protocol]. MCT, Brasília. Available at: <http://www.mct.gov.br > (accessed 25.02.11).

Ministério do Desenvolvimento, Indústria e Comércio (MDIC) [Brazilian Ministry of Development, Industry and Commerce], 2011. Sistema de Análise das Informações de Comércio Exterior [System for the analysis of foreign commerce]. MDIC, Brasília. Available at: <http://aliceweb2.mdic.gov.br/> (accessed 29.09.2011).

Miranda, F. R., Gondim, R. S., Freitas, J. A. D., Pinheiro Neto, L. G., 2008. Irrigação do meloeiro [Melon irrigation], in: Sobrinho, R. B., Guimarães, J. A., Freitas, J. A. D., Terao, D. (Eds.), Produção integrada de melão [Melon integrated production]. Embrapa Agroindústria Tropical, Banco do Nordeste do Brasil, Fortaleza.

Pandey. D., Agrawal, M., Pandey, J. S., 2011. Carbon footprint: current methods of estimation. Environ Monitoring Assessment, 178, 135–160.

Silva, H. R., Costa, N., 2003. Melão: produção aspectos técnicos [Melon: technical aspects of production]. Embrapa, Brasília.

World Resource Institute (WRI), World Business Council for Sustainable Development (WBCSD), 2011. Product life cycle reporting and standard. WRI, Washington. Available at: <http://www.ghgprotocol.org > (accessed 10.01.2012).

Nutrient based functional unit for meals

Thomas Kägi*, Mischa Zschokke, Fredy Dinkel

[1] Carbotech AG, environmental consulting, 4002 Basel, Switzerland
* Corresponding author. E-mail: *t.kaegi@carbotech.ch*

ABSTRACT
The goal of this study was to compare the environmental impact of meals and their ingredients and to analyse the influence of the functional unit on the results. For this study all relevant life cycle phases were considered. The functional units we compared: 1 meal (about 450g); adjusted by the nutrient density score (NDS); adjusted by the nutrient rich food index (NRF9.3). A comparison of the different meals per plate or adjusted by the NRF9.3 method shows that the most relevant impact comes from meat. If the results are weighted by the NDS method, beef still shows a high impact but is not as dominant due to its high nutrient density. Using this method whether the ingredients are regionally or seasonally produced becomes much more relevant. Taking the uncertainty into account a meal with vegetables out of season can but must not have a significant lower environmental impact than a meat meal. The different outcomes due to the three different functional units used show the importance to take into account the adequate circumstances when defining the functional unit. What is adequate to compare - the amount of food, the nutritional value of food or the nutritional health of food? Otherwise, the functional unit and the results may not answer the questions.

Keywords: LCA, meal, nutritional value, functional unit

1. Introduction

In the European Union more than a quarter of the environmental impact is estimated to come from the food chain. There is an on-going discussion on how to reduce the environmental impact along the food production and supply chain. One of the most widely spread proposition is to alter food consumption patterns by replacing animal foods with plant-based foods. And of course there is a big difference between the environmental impacts of the production of 1kg vegetables and 1 kg meat. However, nutritional values or nutritional health of food are seldom considered in such comparisons. It is important to use a functional unit that is relevant from a nutritional perspective.

2. Methods

2.1. Goal and scope

The goal of this study was to compare the environmental impact of real restaurant meals and their ingredients and to show the influence of the functional unit on the results. The following meal combinations were compared (among others): proteins (beef vs. poultry vs. mushrooms) with potatoes and green beans (fresh beans Switzerland vs. fresh beans Egypt vs. fresh beans greenhouse grown).

For this study all relevant life cycle phases were considered including production, further processing, transportation, packaging and cooking of the meals.

The functional units we compared:

- 1 meal (about 450g).
- 1 meal adjusted by the nutrient density score (NDS).
- 1 meal adjusted by the nutrient rich food index (NRF9.3)

2.2. Nutrient density score (NDS) and nutrient rich food index (NRF9.3)

In this study, the NDS was calculated according to Drewnowski (2005). The sum of proteins, carbohydrates, fats, 10 vitamins and 8 trace elements were considered and each weighted according to the recommended daily intake meaning the higher the nutrient density the more valuable the food was. Nutrient contents were taken from the USDA National Nutrient Database for Standard Reference (2011). The nutrient density of an ingredient was calculated as follows:

Nutrient density (NDS) = $\sum$(PNR x PropN)

- PNR: Percentage of nutrient recommendation
  = 100 x (Content of nutrient i in 100g of edible portion divided by the recommended daily value of nutrient i)

- PropN: Proportion number of nutrients >5% of recommended daily value
  = Number of nutrients in 100g of edible portion >5% of recommended daily value divided by 21 (amount of considered nutrients).

The nutrient rich food index (NRF) was calculated according to Drewnowski (2009). The NRF9.3 was chosen because it best correlates to a healthy diet (Drewnowski 2009). The NRF9.3 contains nine nutrients to encourage (protein, fibre, Vitamins A, C and E, Minerals Ca, Fe, Mg, K) and 3 nutrients to limit (saturated fat, added sugar, Na). Nutrient contents were taken from the USDA National Nutrient Database for Standard Reference (2011). The nutrient rich food index was calculated as follows:

Nutrient rich food index (NRF9.3) = $\sum NE - \sum LIM$

- NE: Nutrients to encourage
  = 100 x (content of nutrient i in 100g of edible portion divided by the recommended daily value of nutrient i)
- LIM: Nutrients to limit
  = 100 x (content of limiting nutrient i in 100g of edible portion divided by the maximum recommended daily value of nutrient i)

2.3. Inventory data

Production data for beef and poultry (integrated production, Switzerland) were based on Jungbluth (2000) updated by the Wirz handbook (LBL 2005) and own data inventories. Potato production (integrated production, Switzerland) was taken from the ecoinvent report No. 15 (Nemecek & Kägi 2007). Data for bean production (good agricultural practice, Germany) was based on Lattauschke (2002) and data for mushroom production (conventional, Poland) was taken from Hessische Landesfachgruppe (2002). Data for the further processing of the ingredients and preparing of meals was based on Dinkel et al., (2006). Data for meals was based on real restaurant dishes. The ecoinvent inventory V2.2 database (ecoinvent 2010) was used for other secondary data (fertiliser production, transportation and other) and emission factors.

2.4. Impact assessment

The LCA was performed using the software EMIS 5.7 (Environmental Management and Information System) developed by Carbotech (Dinkel 2011).

For valuation of the environmental footprint the ecological scarcity method (Frischknecht et al., 2009) was used. The method represents the environmental policy of Switzerland and evaluates the emissions and their environmental impacts according to a “distance-to-target”-approach. For evaluation of processes of which the emissions are generated outside of Switzerland it is assumed, that the relative importance of the emission factors is similar. The Eco-indicator 99 method (Goedkoop and Spriensma 2001) was used as a second method for the evaluation of the environmental footprint.

2.5. Uncertainty considerations

To describe the uncertainty of data and model calculations distribution functions like normal or lognormal distribution are used. Especially for emissions where the distributions typically are not symmetric the lognormal distribution is a better approximation than the normal distribution. But the advantage of normal distributions is that there are analytic functions to calculate the uncertainty propagation over the process chain if it is assumed that the uncertainties are independent of each other which is of course not always the case. Due to this shortcoming an overestimation of the uncertainties may be obtained. By using normal distributions the results can be calculated within seconds instead of hours .This is one of the main reasons why in the EMIS software a simplified uncertainty calculation using normal distribution function is used. This means that the uncertainty propagation will be calculated and the user always gets an estimation of the confidence intervals of the LCA results. Even today there are few LCA studies giving the uncertainties of the results even if there are leading software tools giving the opportunity to do an uncertainty calculation with Monte Carlo simulation.

The methodological uncertainties were assumed to be around 20%. Considering the appropriateness of the data being used, uncertainties of the in– and output processes were taken into account. These uncertainties were defined according to the pedigree matrix used in ecoinvent (2010). The uncertainty intervals are presented on the 68% level (standard deviation).

3. Results

The comparison of the environmental impacts of the meals depends on the functional unit we choose.

A comparison of the environmental impact in reference to one meal shows as expected that the most relevant impact comes from meat (Fig. 1). Meals with beef have a significantly higher environmental impact than meals with poultry whilst vegetarian meals show the lowest environmental impact. All other inputs are of secondary relevance compared to meat.

If the results are weighted by the NDS in order to compare the environmental impact of the meals in reference to their nutritional value, meat (especially beef) still shows a high impact but is not as dominant due to its high nutrient density (Fig. 2). Other ingredients such as potatoes and beans also contribute significantly especially if the ecological scarcity method is considered. This is because the nutrient density of meat is much higher than the nutrient density of vegetables. Using this NDS adjusted functional unit whether the ingredients are regionally or seasonally produced becomes much more relevant. Taking the uncertainty into account a meal with vegetables out of season can but must not have a significant lower environmental impact than a meat meal.

If the results are wighted by the NRF9.3 in order to compare the environmental impact of the meals in reference to their nutritional health, meat (especially beef) shows an even higher impact than in the comparison per meal. All other inputs are of secondary relevance compared to meat.

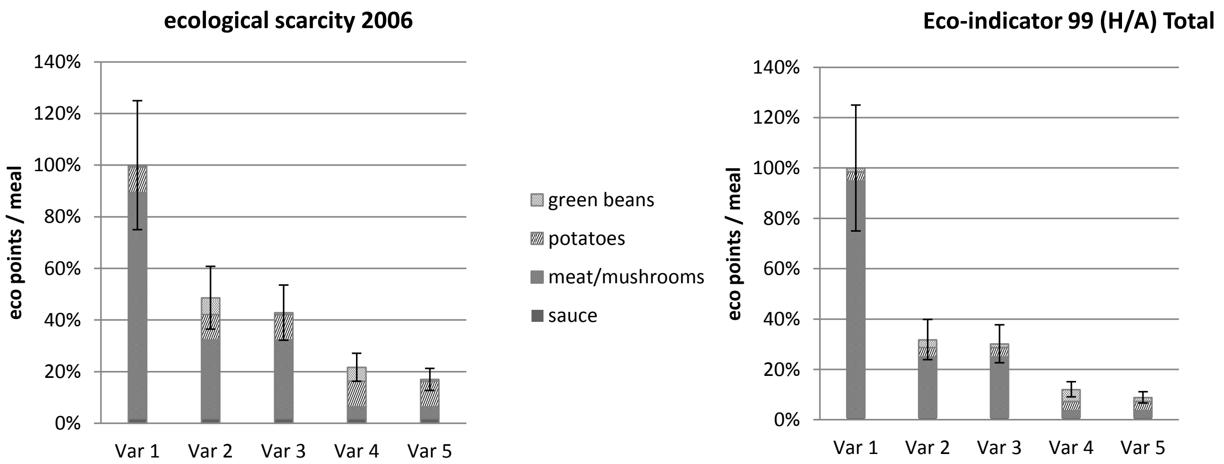


Figure 1. Environmental impact per meal, normalised to the maximum. Var 1: beef+potatoes+green beans CH, Var 2: poultry+potatoes+green beans ES, Var 3: poultry+potatoes+green beans CH, Var 4: mushrooms+potatoes+green beans greenhouse, Var 5: mushrooms+potatoes+green beans CH.

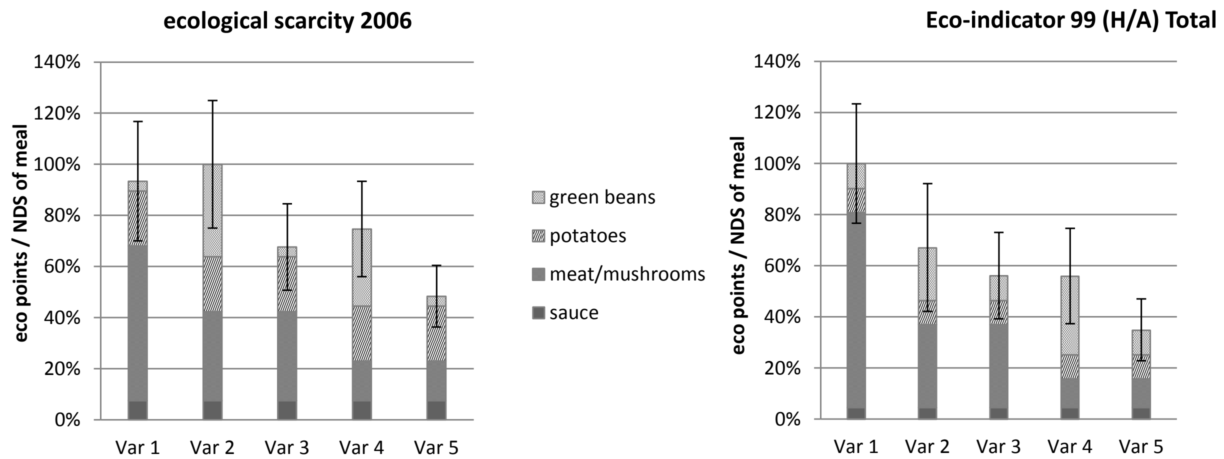
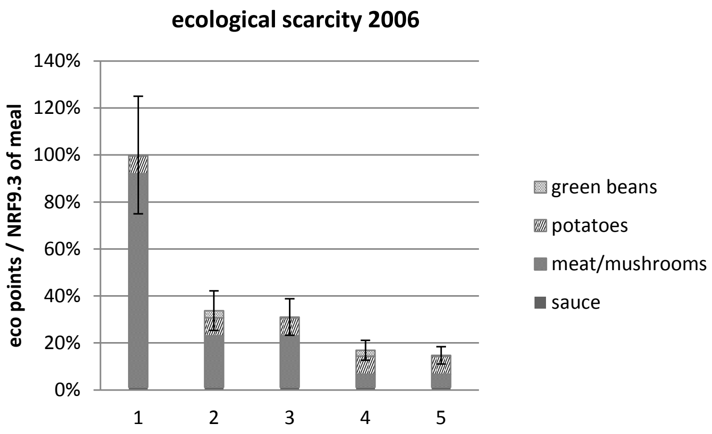


Figure 2. Environmental impact per nutritional value (NDS), normalised to the maximum. Var 1: beef+potatoes+green beans CH, Var 2: poultry+potatoes+green beans ES, Var 3: poultry+potatoes+green beans CH, Var 4: mushrooms+potatoes+green beans greenhouse, Var 5: mushrooms+potatoes+green beans CH.



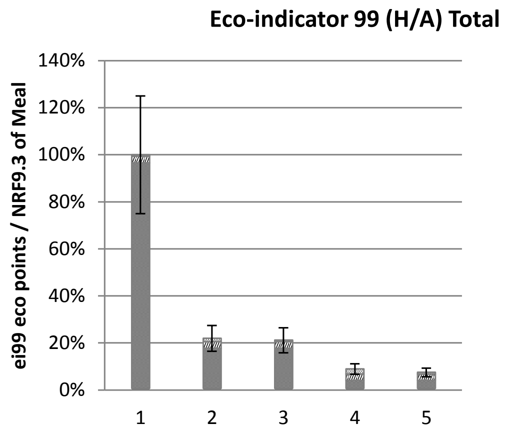


Figure 3. Environmental impact per nutritional health (NRF9.3), normalised to the maximum. Var 1: beef+potatoes+green beans CH, Var 2: poultry+potatoes+green beans ES, Var 3: poultry+potatoes+green beans CH, Var 4: mushrooms+potatoes+green beans greenhouse, Var 5: mushrooms+potatoes+green beans CH.

4. Discussion

With increasing frequency, people and institutions draw conclusions on dietary recommendations from an environmental perspective without a comprehensive analysis of nutritional relevance. Using a functional unit involving only comparable menus in an average restaurant may lead to the conclusion that vegetable alternatives are always better than those of animal origin. There are few studies such as Kurppa et al., (2009) that already included nutritional aspects in environmental impact assessment of food. However, according to our knowledge, most previous studies did not take nutrient aspects into account when discussing the environmental impact of food choices and according to Swedman et al., (2010) environmental emissions have not been explicitly studied when making nutrition recommendations. It is thus important to use both knowledge in nutrition and environmental footprint to avoid simplistic and erroneous conclusions for food recommendations to mitigate the environmental footprint.

The question rises whether it is appropriate to consider nutrient aspects if comparing food choices as most persons choose their meal based on taste, appetite and economic considerations rather than on nutrients. However, there are many labels (e.g. weight watchers) that guide through the food offer on basis of nutrient considerations. On the one hand, although people are driven by appetite and taste, the main goal of eating is not to eat a certain amount of food or savouring good food but remains in providing the body with necessary nutrients. Considering this aspect, it is simply not enough to compare food per weight or per meal without including nutrient considerations. On the other hand, the NDS method does not consider the fact, that too much of nutrient rich food (e.g. meat) is unhealthy and is today a big problem especially in western civilisations. The NRF9.3 considers this aspect by punishing saturated fat, sodium and added sugar. It refers to the nutritional health of food. However, it seems dissatisfying that it does not include all vital nutrients (e.g. unsaturated fatty acids).

Regarding the three different functional units and their different outcomes (Results per NDS differ to the ones per NRF9.3 or per plate) it is of importance to clearly define the circumstances of the study and define the functional unit accordingly: what shall be compared - the amount of food, the nutritional value of food or the nutritional health of food? Otherwise, the choice of the functional will not lead to the adequate results answering the questions of the study. This result also shows that there is not one functional unit to be right for comparing meals but it has to be chosen carefully according to the circumstances. The right functional unit for one country, region or group of people may be the wrong one for other countries, regions or groups of people.

5. Conclusion

This study compared three different functional units for meals of which two considered nutritional aspects of food. Different conclusions may be derived based on the chosen functional unit leading to different food recommendations. The functional unit to be chosen depends not only on the goal of the study but also on the circumstances such as the cultural background or the measure of value of the stakeholder.

We are convinced that the consideration of nutrient density and nutritional health of food is important in the context of the environmental debate. Furthermore, environmental impacts of meals can be directly linked to nutritional considerations of meals.

6. References

Dinkel et al., (2006). Calculations for „Der kluge Einkaufswagen“ Unterrichtseinheit zu den Themen Umwelt, Konsum, Ökobilanzen. Bundesamt für Umwelt (BAFU)

Dinkel F. (2011). EMIS, Environmental Management and Information System, version 5.7, Carbotech AG, Basel, Switzerland.

Drewnowski A. (2005). Concept of a nutrious food: toward a nutrient density score. Am J Clinical Nutr 2005; 82: 721-32

Drewnowski A. (2009). Defining Nutrient Density: Development and Validation of the Nutrient Rich Food Index. Journal of the American College of Nutrition, Vol.28, No. 4, 421S–426S

Ecoinvent (2010). Ecoinvent 2010: Version 2.2, Swiss Center for Life Cycle Inventories. May 2010.

Frischknecht et al., (2009) Methode der ökologischen Knappheit - Ökofaktoren 2006. Bundesamt für Umwelt (BAFU)/ÖBU.

Goedkoop M. and Spriensma R. (2001) The Eco-indicator 99 – a damage oriented method for LCIA, methodology report, , http://www.pre-sustainability.com/content/reports

Hessische Landesfachgruppe, 2002. Fachexkursion nach Polen – Austern- und Champignonanbau, Handel und Substratherstellung. Der Champignon 5/2002.

Kurppa et al., (2009). Environmental impacts of a lunch plate – challenges in interpreting the LCA results. Future of the Consumer Society, May 2009, Tampere

Landwirtschaftliche Beratungszentrale Lindau (LBL), 2005. Wirz Handbuch – Pflanzen und Tiere. Wirz Verlag

Lattauschke, G, 2002. Hinweise zum umweltgerechten Anbau von Freilandgemüse im Freistaat Sachsen – Buschbohnen. Sächsische Landesanstalt für Landwirtschaft. http://www.landwirtschaft.sachsen.de/LFL

Nemecek T. & Kägi T. (2007). Life Cycle Inventories of Swiss and European Agricultural Production Systems. Final report ecoinvent V2.0 No. 15a. Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for Life Cycle Inventories, Zurich and Dubendorf, CH, retrieved from www.ecoinvent.ch

Swedman et al., (2010). Nutrient density of beverages in relation to climate impact. Food & Nutrition Research 2010, 54: 5170

U.S. Department of Agriculture, Agricultural Research Service (2011). USDA National Nutrient Database for Standard Reference. Nutrient Data Laboratory Home Page, http://www.ars.usda.gov/ba/bhnrc/ndl

Carbon footprint of organic vs. conventional food consumption in France

Noëllie Oudet*, Perrine Lavelle, Grégoire Thonier, Maxime Pousse, Eric Labouze

BIO Intelligence Service

* Corresponding author. E-mail: *noellie.oudet@biois.com*

ABSTRACT

Our diet has been proven directly concerned by the struggle against environmental damages. Simultaneously, organic agriculture emerges as a way to mitigate our impact on the environment. Assessing to which extent the rapid development of this rising form of agriculture can have a positive effect on the environment thus appears relevant. The innovative scientific goal of this project is to evaluate the environmental impact of "organic vs. conventional consumption", a wider scale than individual products.

Two steps were followed: characterisation of organic and conventional patterns – considering the global amount of food annually purchased – and application of the LCA methodology to quantify their environmental burden – only global warming potentials were established as no satisfying consumption patterns footprints could be calculated for other impacts.

Organic and conventional consumer carbon footprints respectively reach 1 149 against 1 173 kg CO_2 eq. Improving the definition of the organic pattern could strengthen the conclusions,

Keywords: food consumption pattern, organic, conventional, life cycle assessment, carbon footprint

1. Introduction

European food consumption accounts for 20 to 30% of the environmental impacts generated by consumer products (Tukker, 2006). Our diet is thus directly concerned by the struggle against environmental damages. Simultaneously, organic agriculture, scheme with strict specification, emerges as a way to mitigate our impact on the environment, and organic market has been booming for the past 10 years. It appears relevant to assess the extent to which the rapid development of this rising form of agriculture can have a positive effect on the environment. Indeed, if the environmental benefit of organic farming methods is fairly accepted when results relate to the hectare of land use, impact calculation per unit of production lead to more heterogeneous conclusions. Given foods aim at being eaten, as part of a whole diet, is it relevant to compare environmental impacts of non-organic product and its organic equivalent to estimate the impacts and benefit of the organic farming, as the studies carried out so far seem to suggest? Indeed, what happens if an organic consumer's behaviour differed slightly from the average French consumer's one? Thus, the main scientific and innovative goal of this project is to evaluate the environmental impact of "organic vs. conventional consumption" on a wider scale than individual products.

This project has been carried out in association with the Synabio, the professional union dedicated to French organic food processing companies.

2. Method

Our project was conducted following two steps: first the organic and conventional consumption patterns were defined; second the LCA methodology was applied to quantify their environmental burdens. We focused on the global warming potential, as it remains the only consensual quantitative environmental indicator and has thus been assessed for a greater number of products, comparability of the results being easy to ensure.

2.1. Definition of the organic and conventional consumption patterns

Our definition of the organic consumer is based on the annual quantity of purchased organic items. Kantar Worldpanel, specialist of consumer knowledge based on continuous consumer panels provided us with the number of households purchasing a given number of products per year. For reasons of sample data robustness, we had to define our organic households as purchasing more than 50 organic products per year. The conventional pattern was considered representative of the first two quartile of the population, i.e. purchasing less than 3 organic products per year. On average, annual purchase of organic products in France reaches 20 products.

2.2. Environmental assessment

The following functional unit was chosen: “to product, supply and consume and average food basket per person in France in 2009”.

The carbon footprint of each basket was established looking at each step of the products life cycle. Given the variable accessibility to data according to the life cycle step, different approaches were used to evaluate the associated greenhouse gases emissions.

Cultivation and farming

Literature review was performed to list available and usable carbon footprints. Obviously, the list of food products provided by Kantar Worldpanel was not completely covered by the publications results; simple original products like kiwis, or processed products such as ratatouille have no publicly available environmental information. In addition, organic products have been far less studied than conventional ones, and often lack environmental impact data.

When missing, environmental information for conventional items were estimated conducted simplified LCA, or by analogy with closest products (e.g. grapefruit environmental burdens were considered equal to oranges ones). We acknowledge the uncertainty related to these approximations. Regarding organic items, missing environmental data was replaced with conventional products ones (e.g. organic goat milk environmental impacts were replaced with conventional goat milk burdens).

From raw material transportation to consumer use

Several scenarios were established for each of the following steps of the products life cycles. Number and parameters of the scenarios were defined according to their representativeness and their influence on the final results.

- Raw material freight: 2 scenarios based on transportation modes and distances differences;
- Industrial processing: 19 scenarios based on the energy consumption and the refrigerant emissions;
- Packaging: only primary packaging were considered and 14 scenarios were modelled based on material type and weight, and industrial processes;
- Distribution and retail: 3 scenarios according to storage conditions;
- Consumer use: 6 scenarios depending on the storage and cooking practices.

GHG emissions were calculated for each of these scenarios.

End-of-life

A survey performed on behalf of the ADEME (ADEME, 2007) mentions that a French consumer wastes about 79 kg or food per year on average, meaning that 16% of the food constituting our basket is thrown away. This figure is in agreement with values provided by WRAP (WRAP, 2008) which indicates that 68 kg of food is wasted per person and per year in the UK. However, different values have been found in other studies. For instance, Munoz (Munoz et al., 2010) indicates that approximately 200 kg of food waste are produced per person and per year in Spain. The 79 kg value was used in the study but this value remains quite uncertain and a more precise assessment of food waste is needed.

Repartition of waste according to their destination (landfill, incineration, recovery) was made thanks to data from the ADEME (2007) and the ADEME AFNOR platform for packagings (2011).

3. Results

3.1. Organic and conventional consumption patterns

Kantar Worldpanel provided us with a list of 215 food items which were distributed into 10 categories: Meat, fish and eggs; Dairies; Ready-made meals; Fruits and vegetables; Starchy foods; Bakery wares; Condiments; Coffee, tea and chocolate drinks; Beverages and Baby food.

Among the 215 items, 149 were distinguished between organic and conventional, 66 remained undetermined. Conventional and organic consumers’ purchases amount to 524 and 496 kg per year, with respective mass shares of organic products of 0.3 and 11.6%, beverages excluded.

Figure 2 shows the composition of the five food baskets. We can notice that organic consumer’s purchases exceed conventional ones by near 30 kg. This can be explained by the higher incomes of this population, and by the need of consuming more given the less energy dense products purchased.

Regarding the composition, some differences can also be observed. One notable trend appears: fruits and vegetables consumption is closely correlated with organic product purchases. As for Dairies, Meat, fish and eggs, and Ready-made meals purchases, they slightly decrease while the organic product purchases increases.

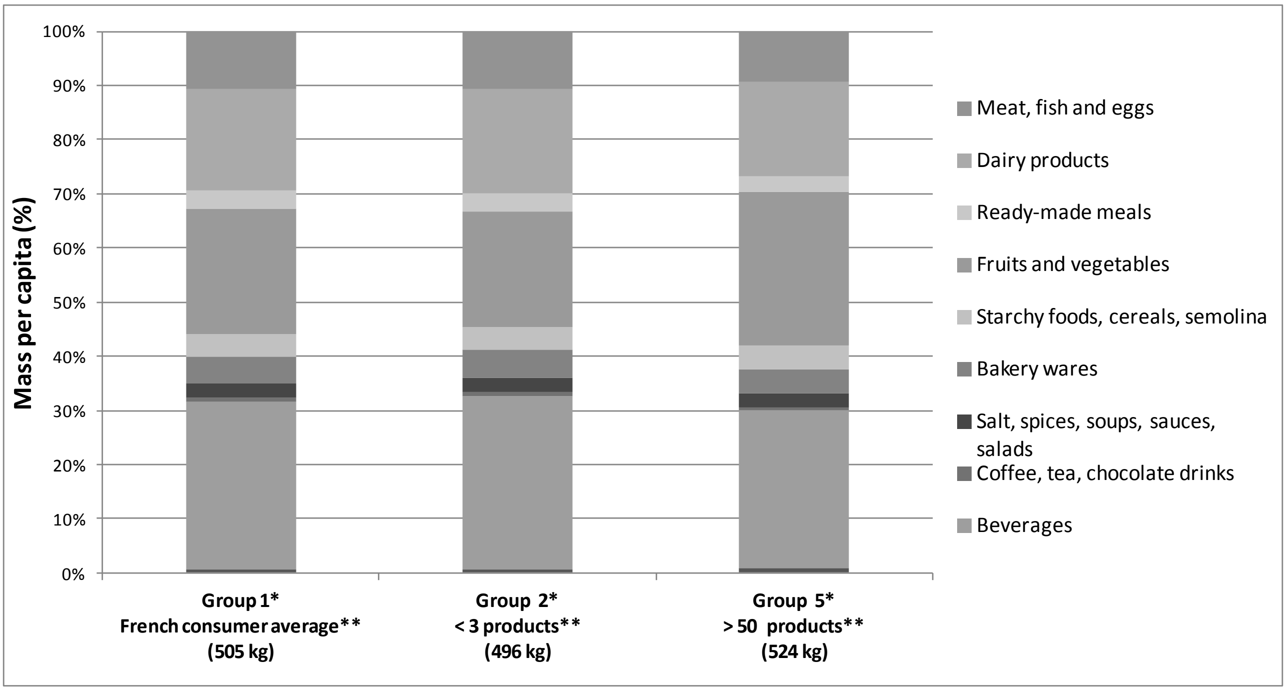


Figure 2. Composition of the food baskets

*: group 1: French average consumer, group 2: conventional consumer, group 5: organic consumer

**: number of organic products purchased per year and per household

3.2. Emission of GHG per kg of product overview

Emissions of GHG per kg of product (carbon footprints) used in this study come from publications or LCA performed during this project, reconstruction via simplified LCA or approximations as explained before. Table 12 details the origins of the selected values, and their respective shares of basket mass and impact for French average consumers. If LCAs (exhaustive or simplified) cover the half of the products, they encompass more than 2/3rd of the global footprint either because of their significant amounts or because of their high environmental burdens, are covered.

Table 12. Number of carbon footprints selected according to the different sources – Distinction provided per number of product, mass and impact shares and per production modes for the French average consumption basket (line totals may differ from 100% due to rounding).

French average consumer	Emissions factors from published LCA	Emissions factors re-calculated from published LCA	Other
Number of product	**104 (28%)**	**85 (23%)**	**176 (48%)**
Organic	26 (17%)	34 (23%)	89 (60%)
Conventional	51 (34%)	42 (28%)	56 (38%)
Undetermined	27 (40%)	9 (13%)	31 (46%)
Mass distribution	**52%**	**11%**	**37%**
Organic	54%	5%	41%
Conventional	46%	19%	35%
Undetermined	59%	3%	38%
Impact distribution	**51%**	**18%**	**31%**
Organic	54%	6%	41%
Conventional	38%	24%	39%
Undetermined	70%	11%	19%

3.3. Environmental impact assessment

Table 13 displays the global annual GHG emissions of each kind of consumer. At first glance, one can see that despite organic consumers purchase more products (in weight), their carbon footprint is very close, but slightly inferior, to the conventional consumers' one. Given the uncertainty of both activity data and emissions factors, the lower carbon footprint of the organic consumer cannot be considered significant. However, it can be propounded that organic consumption basket composition counteracts the higher mass purchased. If the mass were the same as for the conventional consumer, the carbon footprint would be 5.5% lower.

Table 13. Annual purchases and GHG emissions per capita

Consumers groups (nb of organic products purchased)	**Food basket weight (kg/capita/year)**	**Global Warming Potential (kg. CO_2 eq.)**
Group 1 - French consumer average	505	1179
Group 2 - Conventional consumer	496	1173
Group 5 - Organic consumer	524	1149

When considering the impact repartition according to food products categories, we can notice that for most categories, and especially Meat, fish and eggs, Ready-made meals, Fruits and vegetables and Beverages, trends are very correlated to the ones observed on the basket composition (i.e. less Meat, fish and eggs in group 5 basket results in a lower impact). Indeed, mass variation has more effect than the difference between organic and conventional carbon footprints, the latter being often little and not always favouring one or the other farming practices type.

Regarding the organic consumer, the higher carbon footprints of organic products are counterbalanced by the composition of the food basket (i.e. more fruits and vegetables, less beverages, meat and dairy products in comparison to the average consumer). Hence, the latter emerges as a major lever to reduce the carbon footprint of the diet.

The comparison of the mass and impact profiles of each consumer highlights that these two aspects are not correlated, as illustrated in Figure 3. Indeed, some categories are highly consumed but lead to minor contributions to global GHG emissions (e.g. beverages, fruits and vegetables), others carry significant part of the impact even if they are consumed in lower quantities (e.g. Meat, fish and eggs, Dairy products). This comes from the fact that carbon footprints vary greatly from one food category to another.

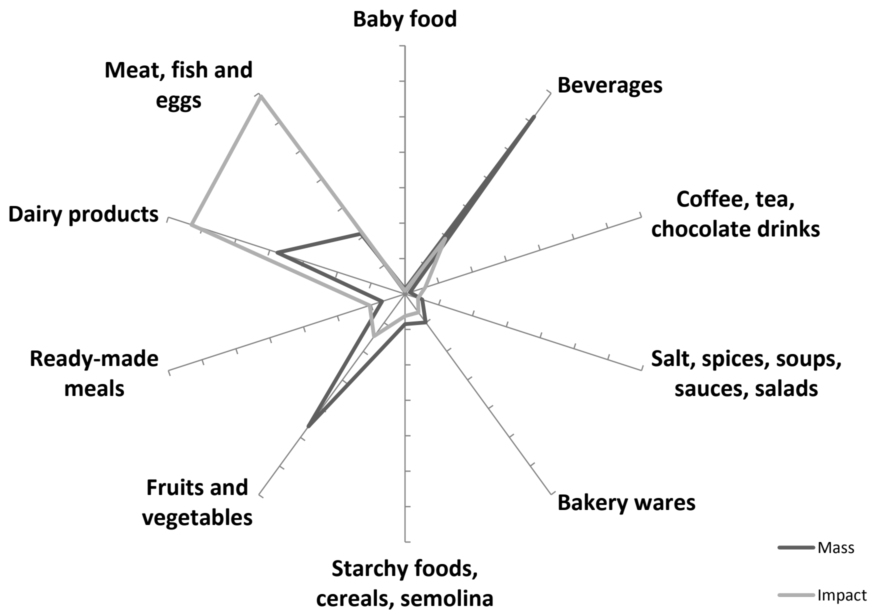


Figure 3. Mass and impact repartition for the French average consumer (1 graduation – 5%)

Focusing on specific categories, we have noticed that although carbon footprints are higher for organic meat and dairy individual products, the average footprints of the respective categories are lower for this scheme of production. This results from the kinds of meat and dairies purchased, since less impacting meats (chicken, eggs) are most commonly consumed when considering organic products while pork and beef consumptions, two strong emitting products, are significant among conventional (and undetermined) purchased meat products.

Finally, provided consumption levels and environmental impacts of these categories, Meat, fish and eggs and Dairy products appear as the main leverages to mitigate the emissions of the consumption basket.

4. Discussion

The comparison of our results to those from previous studies shows that orders of magnitude are on the whole similar, the two Spanish studies (Munoz *et al*., 2010; Santacana et al., 2008) outcomes being superior. Our results are thus in line with the literature, providing the uncertainty in these kinds of studies, and the various followed methodologies.

Moreover, our conclusions about the high contributors to the global emissions are concordant. For example, the share of “raw material production” is dominating the general result. This phase is actually for all groups the most contributing to the general food basket impact. It represents about 75% of the total impact for each group. This conclusion is in adequacy with Munoz’ study on the impact of the average Spanish diet, food production accounting for circa 2/3rd of the global GHG emissions (Munoz *et al*., 2010). The most impacting products categories are Meats, fish and eggs as well as Dairy products, with respectively 406 and 372 kg CO_2eq in the average consumer’s basket. The predominance of these two categories of products in the impact of raw material production is also coherent with Munoz’ results: meat and dairy products contribute to 54% of the impact of raw material production, while their contribution reaches 62% of raw material production in the present study.

We acknowledge our model faces several limitations; first regarding the consumption profiles definition, the lack of discrimination between organic and conventional products and the share of approximated carbon footprints in the model.

When studying the distribution of organic, conventional and undetermined products, organic consumption represents 8% of the total weight of organic food basket. This result raises questions about the “organic” consumer and the little differences among the organic and conventional consumption baskets. However, the Kantar Worldpanel data did not allow defining a more detailed and representative sample, based on consumers buying more than 100 or 200 organic products a year for instance. In addition, in the panel, the share of undetermined products remain high (48% and 25% considering Beverages or not). In particular, fresh (i.e. unprocessed) products such as meats or fruits and vegetable showed very little distinction of the agricultural production scheme, while they account for respectively 13 and 17% of the global organic products turnover in 2010, behind grocery products and beverages (24%) and dairy products (22%) (Agence Bio, 2011); and their contributions to the global emissions of the baskets are predominant.

Therefore, further research is needed to establish the organic consumption pattern; as both the number of products in the basket – which may result in a new basket composition – and the precision of the agricultural production mode of key products have to be improved. This new and more accurate organic profile could be built thanks to consumers interviews for instance, the consumers being selected among recurrent buyers of organic food stores.

Finally, still, one third of the global impact results from product carbon footprints that have been approximated, either by considering the GHG emissions per kg of product of the closest product or by taking the conventional item carbon footprint when the organic one was missing or the production scheme undetermined. For instance, 10 emissions factors were modelled from the emissions factors of organic wheat, and as oranges production mode was not available; the emissions factor of conventional oranges was used. Thus, not only is the emissions factors accessibility a limit to our assessment, but also the availability of distinct information between organic and conventional product. This result tends to confirm the uneasiness to conclude about the potential benefits or drawbacks of consuming organic food compared with conventional food, providing the low availability of emissions factors for organic food. This also addresses the need to conduct LCAs comparing the environmental impacts of organic and conventional products. Ideally, the LCAs conducted should compare the products, the production modes (organic, conventional, integrated) and the geographical areas (region of a country).

5. Conclusion

Although limitations remain about our definition of the organic pattern, distinctions in the consumption have been established, resulting in environmental impacts variations. Given the uncertainties of the results no conclusion can be drawn about the effective benefit of the organic pattern, but the fact that this specific basket composition counterbalances the higher purchased food amount has been proven. In the context of research to identify new consumption modes, our results are a first step toward the definition of a "more environmentally-friendly" diet.

For a definition of a sustainable diet (in terms of environmental impacts), it could be interesting to investigate other environmental impacts, such as water consumption or eutrophication, relevant issues for food products (ADEME AFNOR, 2011). However, these indicators are still under development, as no consensual calculation method has been established, and available results remain hardly comparable.

A further step addressing the nutritional value of the various consumption baskets would help defining better consumption patterns for both human and our planet's health.

6. References

ADEME, 2007, Campagne nationale de caractérisation des ordures ménagères (MODECOM), résultats 2007. http://www2.ademe.fr/servlet/KBaseShow?sort=-1&cid=96&m=3&catid=15163

Agence Bio, 2011, L'agriculture biologique, chiffres clés - Edition 2011. www.agencebio.org

Muñoz, I., Mila I Canals, L., Fernandez-Alba, A., 2010. Life cycle assessment of the average Spanish diet 2010 including human excretion. Int J of Life Cycle Assess. 15:794-805

Santacana M, Pon J, Pon D, Arto I, Casanovas S (2008) Greenhouse gas emissions from a consumption perspective in a global economy — opportunities for the Mediterranean region. Sustainable consumption and production in the mediterranean, regional activity centre for cleaner production (CP/RAC). Annual Technical Publication 7:101 – 112

Tukker, A., Huppes, G., Guinée, J., Heijungs, R., de Koning, A., van Oers, L., Suh, S., Geerken, T., Van Holderbeke, M., Jansen, B., Nielsen, P., European Commission - Joint Research Center - Institute for Prospective Technological Studies, European Science and Technology Observatory, TNO, 2006, Environmental Impact of Products (EIPRO), Analysis of the life cycle, environmental impacts related to the final consumption of the EU-25, EUR 22284 EN.

WRAP, 2008, Food waste report v2, The food we waste, ISBN: 1-84405-383-0 (version 2), Written by Lorrayne Ventour, July 2008.

Nutrition in LCA: Are nutrition indexes worth using?

Merja Saarinen

MTT Agrifood Research Finland, E-mail: *merja.saarinenr@mtt.fi*

ABSTRACT

Environmental impact of food is mostly considered on a mass basis, although nutrition undoubtedly is one of the main functions of food. The paper develops a methodology to combine nutritional and environmental aspects in food LCA. The usability of nutrition indexes is tested and discussed. The nutrition indexes tested were Nutrient Rich Food, NRF9, Naturally Nutrient Rich, NNR15, and Nutrient Adequacy Ratio, NAR15. They are intended for application to all types of food. The test calculation included several foods from several product groups based on general nutrient composition data and environmental impact data. Preliminary results show that environmental impact seems to dominate and strengthen the recommendation to restrict animal-based foods and favour plant-based foods. However, inclusion of recommended and limited nutrients is a necessity before further application of the method, and a reformulation of the nutrient composition of the index has also to be considered.

Keywords: food, nutrient index, LCA

1. Introduction

According to the ISO 14040, standard functions of the system being studied shall be clearly specified, and comparisons between systems shall be made on the basis of the same function(s), quantified by the same functional unit(s) in the form of their reference flows. Selected FU(s) should go together with a goal and scope of the study.

Currently environmental impact of food is mostly considered on a mass basis in public debate and scientific discussion. There is however a growing interest in considering nutrition and environment simultaneously (e.g. Saarinen et al., 2012; Schau and Fet, 2008; Smedman et al., 2011). This can be seen as an outgrowth of increasing efforts to inform consumers about sustainability of food products instead of using LCA as a tool to improve the production chain. As long as environmental sustainability of food products is considered a field for improving eco-efficiency of producers, nutrition will be neglected. But consumer information on sustainable food choices is valueless without considering nutrition. There are also several other functions associated with eating that are very relevant to consumers, like pleasure, social interaction etc., but nutrition is the basic function of food when comparisons are made in the context of sustainability. As comparison between products is made on the basis of the same functional units, opportunities to use “nutritional functional units” should be carefully considered.

The paper develops a methodology to combine nutritional and environmental aspects in food LCA. Climate change and eutrophication potential were used as examples of the environmental category. The usability of nutrition indexes is tested and discussed.

2. Methods

The nutrition indexes tested were Nutrient Rich Food, NRF9, Naturally Nutrient Rich, NNR, and Nutrient Adequacy Ratio, NAR16 (Drewnowski and Fulgoni, 2007). NAR16 includes 16 nutrients, but the Finnish nutrient database does not provide information on pantothenic acid (B5), so in this study NAR16 included 15 nutrients as NAR15. NNR also includes 15 nutrients, and is thus NNR15. These indexes combine recommended nutrients and quantify an average share of nutrients in 100 g of a product (or any other amount of food) from their daily nutrient recommendation. The indexes, their nutrient composition and formulas, are presented in (Table 1).

Other indexes consisting of nutrients to recommend were also considered, but the lack of nutritional data restricted their use. From the point of view of the science of nutrition, indexes that include nutrients to be recommended and nutrients to be limited provide the most accurate feature of the nutritional value of foods, and therefore basically both types of nutrient should be considered. In this study, indexes consisting of both nutrients to recommend and nutrients to limit were applied to nutrient calculations, but not to environmental assessment because they confer negative values to some foods (not shown in the paper). Negative values are not directly applicable in LCA as they result in a negative environmental impact. Indexes that proportion nutrients to energy content of food (i.e. nutrient density indexes) were also applied to nutrient calculations, but not to environmental assessment. They basically are suitable for nutrient calculation in situations where energy needs to be limited. In such a case energy is regarded as a disadvantage. Accordingly, high energy content reduces the nutrient density value, and low energy content increases the nutrient density value. If these are used as a FU in the LCA, the interpretation would be reversed so that energy would be an advantage; high energy content would result in lower environmental impact than lower energy content. Indexes consisting of only nutrients to be limited were not regarded as suitable for linking to the LCA by means of FU because they do not represent a benefit gained from food, but rather a disadvantage.

The nutrient indexes attempt to represent a general nutritional value for individual foods by reflecting dietetic nutritional values. They are intended for application to all types of food. Indexes are meant to be applied to unfortified foods as they are intended to illustrate the natural nutritional value of food. In this study however, drinks were fortified, like skimmed milk with vitamin D, and oat drink and soya drink with calcium and vitamin D. These products are generally fortified, milk actually even by law (in Finland). They are used as alternative drinks linked to meals.

The test calculation included several foods from several product groups. Nutritional data were based on national nutrition recommendations (Anon., 2005) and national food composition database (Fineli® - Finnish Food Composition Database). LCA results in categories of climate change and eutrophication potential used in the calculations are approximations for final products derived from raw material use (recipes) and LCA data from previous studies (Saarinen et al., 2012; Usva et al., 2012; see also Virtanen et al., 2011 and Saarinen et al., 2011), the Ecoinvent database and Wanhalinna (2010). LCA results for most products represent raw material production of main ingredients on farm, but the LCA results for rye breads include also baking. LCA results for bread are not sensitive to the production method and " the darkness of bread", so they were only applied to industrial rye bread and light bread in environmental calculations (excluding traditional rye bread and whole grain wheat bread, unlike in the nutritional calculation). Manufacture of added calcium and vitamin D for fortified soy drink and oat drink were not included in the LCA values. The LCA values for skimmed milk represent milk from the dairy, i.e. there was no allocation applied that differentiated milk and fat. LCA results for macaroni were not available, nor were information on recipe. Therefore macaroni was not included in the environmental calculation. The eutrophication potential of olive oil, and rye and light bread were not available, so they were not included in the eutrophication calculation.

The nutrient indexes were combined in the LCA by the formula:

E/N index = LCA result / nutrient index **(**equation 1)

Both LCA results and nutrient index measure were calculated per 100 g of a product.

Table 1. Nutrient indexes tested in the study.

Nutrient index	Formula	Nutrients included
Nutrient Rich Food, NRF9	NRF9 = $\Sigma_{1\text{-}9}$((Nutrient/DV) * 100)/9	Protein, fibre, Ca, Fe, Mg, K, Vit A, C and E
Naturally Nutrient Rich, NNR	NNR = Σ_{1-15}((Nutrient/DV) * 100)/15	Protein, fibre, MUFA, Ca, Fe, Zn, K, Vit A, C, D, E, thiamin (B1), riboflavin (B2), B12 and folate
Nutrient Adequacy Ratio, NAR15	NAR15 = Σ_{1-15}((Nutrient/DV) * 100)/15	Protein, fibre, Ca, Fe, Mg, Vit A, C, D, E, thiamin (B1), riboflavin (B2), niacin (B3), pyridoxine (B6), B12, and folate

3. Results

3.1. Nutrient index scores

Nutrient indexes NRF9, NNR15 and NAR15 rank foods quite similarly (Figure 1). In general, vegetables, vegetable oils, fruits and berries and rye breads fare well according to every index. However, indexes seem to treat meat and fish products differently and also vegetable oils.

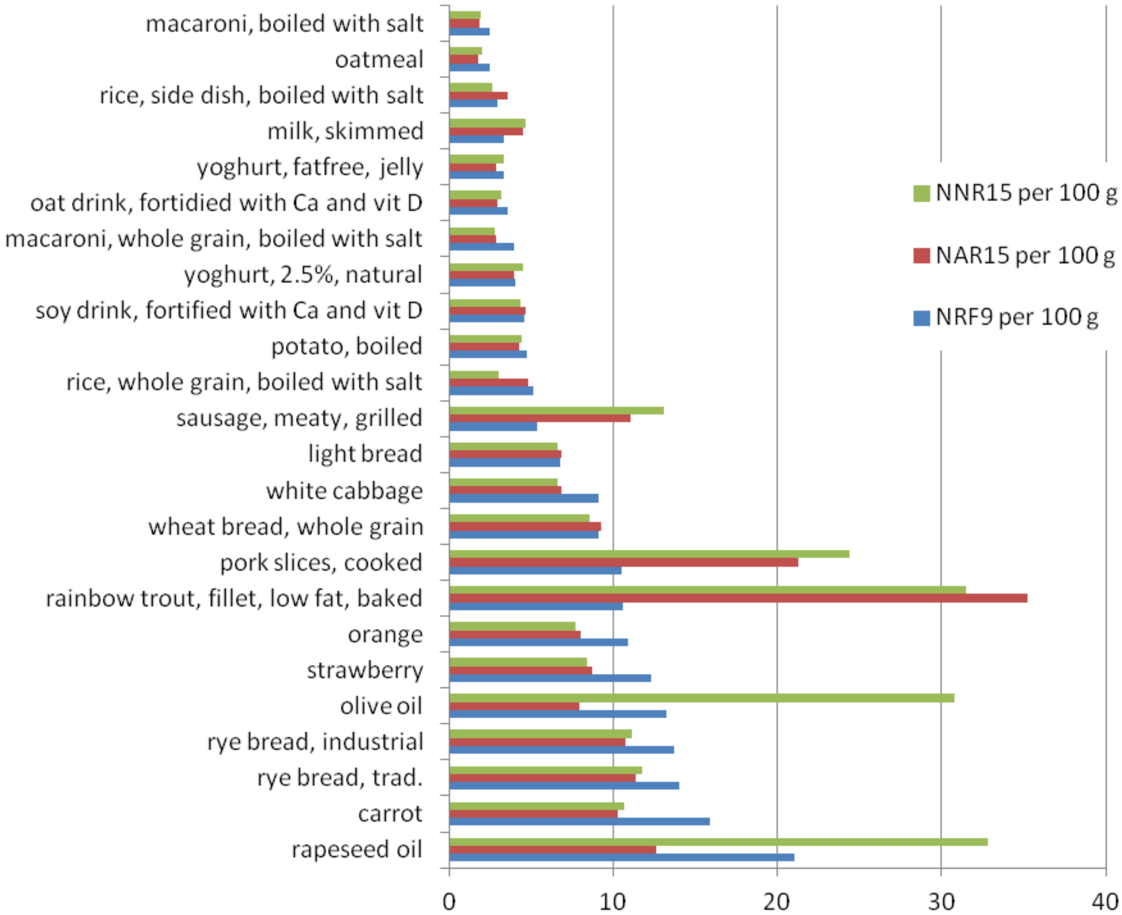


Figure1. Scores for NNR15, NAR15 and NRF9 of foods sorted according to NRF9.

3.2. E/N index

In this study the E/N (environmental/nutrient) index was applied in the categories of climate change and eutrophication potential. In general the E/N index values for the climate impact varied more than the nutrient index values, but not so much for eutrophication. For the climate impact (Figure 2) animal-based foods and rice, especially refined rice, have the highest values, while carrot and olive oil have the lowest values. Values for sausage, pork, rainbow trout, milk, whole grain and refined rice and rapeseed oil vary the most among the different methods.

For the eutrophication potential (Figure 3), rainbow trout have by far the highest values following sausage, pork, milk, fatless and natural yoghurt, rapeseed oil and strawberry, while soy drink and carrot have the lowest values. However, the scale of the E/N index values for eutrophication potential is so large that differences among the lowest foods cannot be seen in the general picture.

The E/N index values are very preliminary as the LCA values are approximations based on LCA results for raw material production of main ingredients: they are not results from the full LCAs. In addition, both environmental and nutritional data used in the study are general data, not data from production in a specific production chain. The data are well suited for methodology development, but not for comparisons between products in general or between products from specific production chains.

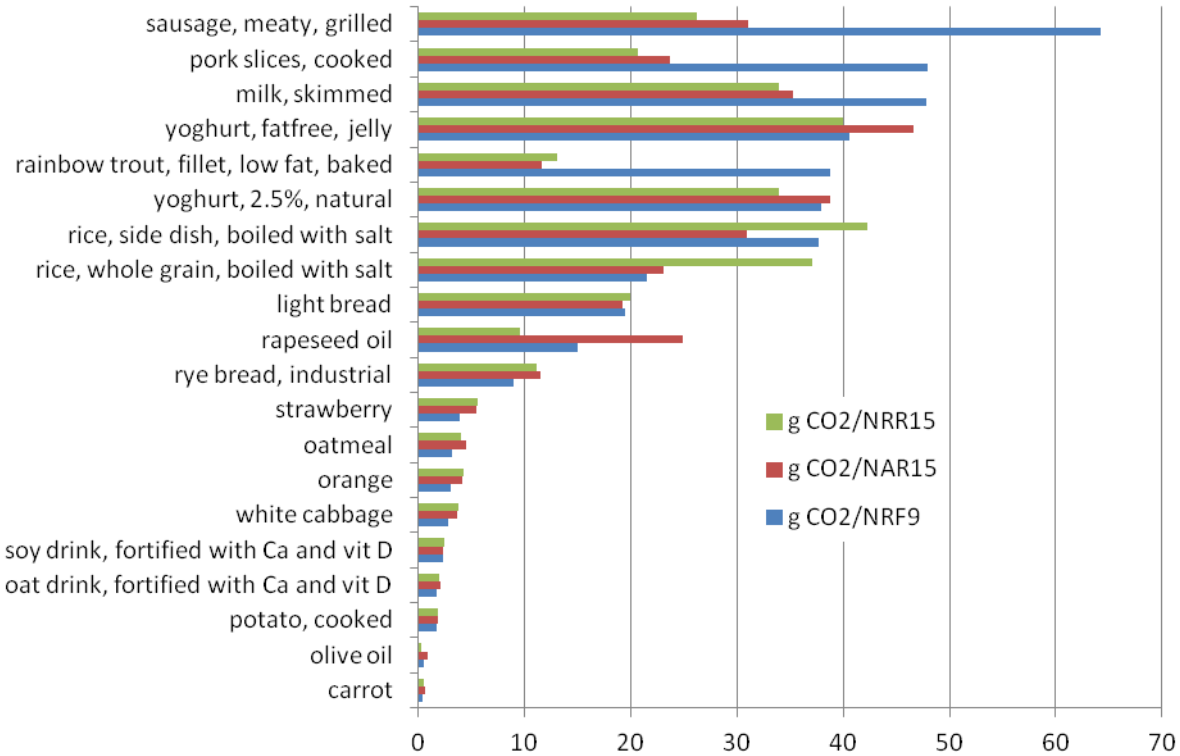


Figure 2. Carbon footprints for the nutritional value unit of foods according to NNR15, NAR16 and NRF9 indexes.

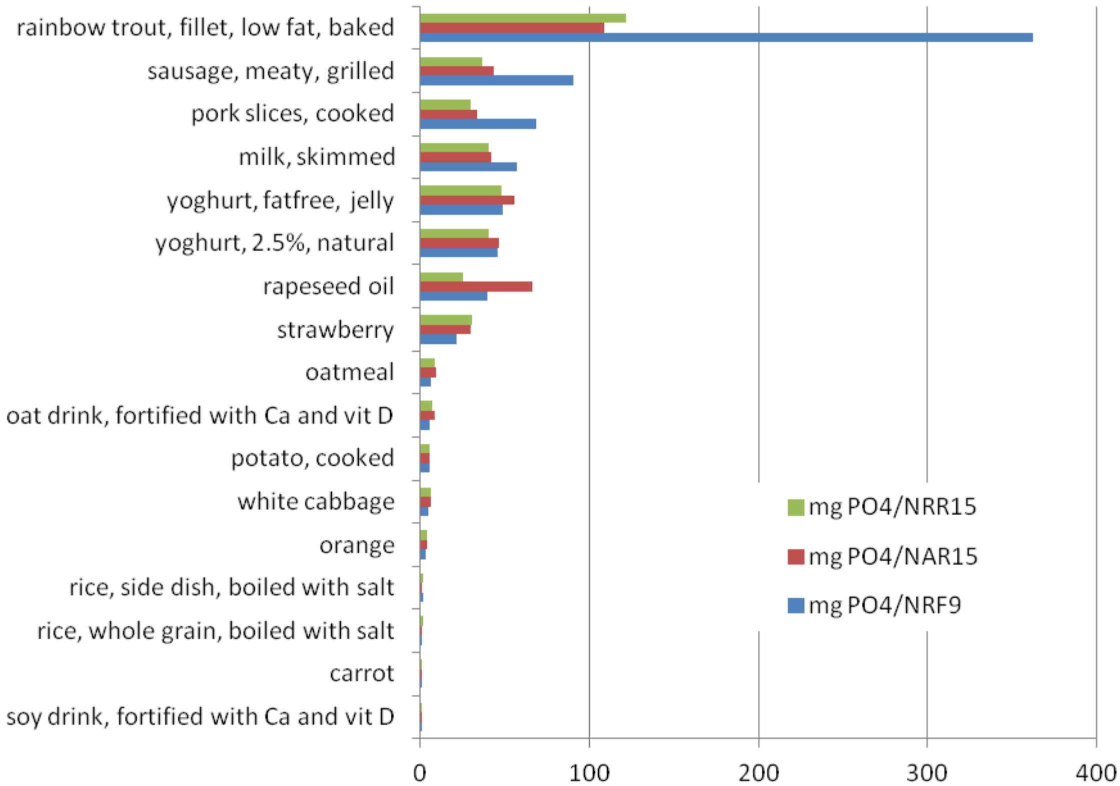


Figure 3. Eutrophication potentials for the nutritional value unit of foods according to NNR15, NAR16 and NRF9 indexes.

4. Discussion

4.1. Nutrient index scores

Nutrient indexes quantify an average share (%) of selected nutrients in 100 g of a product with reference to the daily nutrient recommendation. Differences between results for nutrient indexes (Figure 1) highlight the fact that it is crucial to decide which nutrients to include in the index. Vitamins from group B favour meat products (scores of NNR15 and NAR15) while MUFA (monounsaturated fat acids) favours vegetable oils (scores of NNR15).

It is also notable that the index with the lowest number of nutrients (NRF9) provides the most stable outcome. Fulgoni and colleagues (2009) validated nutrient indexes, which contain different numbers of nutrients, against the healthy eating index (HEI) in the USA. They established that index NRF9.3 correlated best with the HEI. The index consists of nine recommended nutrients, as in this study for index NRF9, and three nutrients to be limited, which are saturated fatty acids, added sugar and Na. In this study, the order of foods would be dramatically different if the nutrients to be limited were included in the indexes. Olive oil, rainbow trout, pork, light bread and sausage would lose ground most, while strawberry, orange, white cabbage, potato, fortified soy drink, whole grain macaroni, fortified oat drink, skimmed milk, oatmeal and macaroni would most clearly improve in their performance. The nutrients to limit have to be built into the nutrient index before further application in the context of LCA.

It is also still questionable as to the most important recommended nutrients included in the indexes. For example, unsaturated fatty acids and antioxidants are currently regarded as very important to our health. Eventually, however, selection of nutrients depends on a selective approach. The approach of the indexes considered in this study is to generate a general nutrient value for food, which is generally applicable to all humans. Alternative approaches might be more culture dependent or even genotype dependent.

4.2. E/N index

The E/N index quantifies environmental impact of the nutrient index unit in a selected impact category. The calculation is based on 100 g of product, but the E/N index value is independent of the amount of food.

The preliminary results should not be interpreted to represent an exact order for the foods considered. General findings regarding the logic of the method can however be made. Environmental impact seems to dominate the E/N index results to a small degree. An order of foods by climate impact does not change the order of foods much by the NRF9 index value compared to the order of foods by the E/N index value (Table 2).

Plant-based products seem to maintain their environmental benefit compared with animal-based products when nutrient content is taken into account in the study approach. However, some shifts occur within the plant-based foods and within the animal-based foods. For example, fat-free yoghurt with jelly loses to natural (2.5%) yoghurt in E/N ranking, but not in the ranking by CO_2 eq. This kind of comparison might be clearer if the index were more sensitive to differences between product croups (and using better defined data). This idea takes us back to the start; would this kind of general nutrient index provide the information that would take account of the nutritional function of food so that it would help consumers choose more sustainable food products? I would say, probably not in the best possible manner, but product-category-specific indexes (and more specific data) may be needed. This is most obvious when vegetable oils are compared with other products and each other. Both olive oil and rapeseed oil act quite unexpectedly in the calculations. There are two main factors behind this: 1) typical portion size differs considerably from other foods studied here and 2) exceptional nutrient composition. These kinds of feature could be taken into account in a product-group-specific index. Also Fulgoni and colleagues (2009) proposed that development of a nutrient index within a product group should be examined. The problem still is how to define the product groups.

5. Conclusion

The study provides an interesting new approach and methodological framework for linking food's nutritional function to food LCA. Nutrient indexes are worth using, but before practical applications are possible, nutrient composition of the nutrient indexes should be evaluated against current nutritional science and nutritional requirements of consumers. There might be a need for a newly composed nutritional index, and its product-group-specific applications. Definition of product groups should be based on a manner to use foods, not on, for example, raw material base of foods. Nutrients to be limited have to be incorporated to the index, but not in the way that they are in the current indexes. And last but not least, more specific data should be used in the practical calculations. The target should be to use production-chain-specific data.

We are addressing these challenges in the SustFoodChoice project, where methodological development and case-study results regarding several food products are pursued in cooperation with the Finnish food industry.

Table 2. The order of foods by NRF9, E/N index and by CO_2 eq.

Product	Order of products by NRF9 index	Order of products by CO_2 eq	Order of products by E/N index
rapeseed oil	1	17	11
carrot	2	1	1
rye bread, industrial	3	12	10
olive oil	4	3	2
strawberry	5	9	9
orange	6	8	7
rainbow trout, filee, low fat, baked	7	19	16
pork slices, cooked	8	20	19
white cabbage	9	7	6
light bread	10	13	12
sausage, meaty, grilled	11	18	20
rice, whole grain, boiled with salt	12	11	13
potato, cooked	13	4	3
soy drink, fortified with Ca and vit D	14	6	5
joghurt, 2,5%, natural	15	15	15
oat drink, fortified with Ca and vit D	16	2	4
joghurt, fatfree, jelly	17	14	17
milk, skimmed	18	16	18
rice, side dish, boiled with salt	19	10	14
oat meal	20	5	8

6. References

Anon., 2005. Suomalaiset ravitsemussuositukset - ravinto ja liikunta tasapainoon 2005. (in Finnish) (Finnish Nutrition Recommendations – nutrition and exercise to the balance 2005.) Valtion ravitsemusneuvottelukunta. 56 p. Edita, Helsinki. ISBN 951-37-4501-5.

Drewnowski, A. and V. Fulgoni III, 2007. Nutrient profiling of foods: creating a nutrient-rich food index. Nutrition Reviews® Vol 66(1) 23-39.

Fineli® - Finnish Food Composition Database. National Institute for Health and Welfare. www.fineli.fi.

Fulgoni, V.L., Keast, D.R. and A. Drewnowski, 2009. Development and Validation of the Nutrient-Rich Foods Index: A Tool to Measure Nutritional Quality of Foods. J Nutrition 139, 1549-1554.

Saarinen, M., Kurppa, S., Virtanen, Y., Usva, K., Mäkelä, J. and A. Nissinen, 2011. LCA approach to the impact of homemade, ready-to-eat and school lunches on climate and eutrophication. J Cleaner Production 28, 177-186.

Saarinen, M., Virtanen, Y. and H. Hyvärinen, 2012. LCAs of large repertoire of Finnish outdoor plant products. Abstract for LCA food 2012 conference.

Schau EM and AM Fet, 2008. LCA Studies of Food Products as Background for Environmental Product Declarations. Int J LCA 13 (3) 255-264.

Smedman, A., Lindmarka-Månsson, H., Drewnowski, A. and A-K.M Edman, 2011. Nutrient density of beverages in relation to climate impact. Food & Nutrition Research, 54, 5170 – DOI: 10,340/fnr,v54i0,5170.

Usva, K., Nousiainen, J., Hyvärinen, H. and Y. Virtanen 2012. LCAs of average animal products pork, beef, milk and eggs in Finland. Abstract for LCA Food 2012 Conference.

Wanhalinna, V., 2010. Leivän hiilijalanjälki (in Finnish) (Carbon footprint of bread). Masters thesis. University of Helsinki. ISSN 0355-1180.

Virtanen Y, Kurppa, S., Saarinen, M., Katajajuuri, J-M., Usva, K., Mäenpää, I., Mäkelä, J., Grönroos, J. and A. Nissinen, 2011. Carbon footprint of food – approaches from national input-output statistics and a LCA of food portion. J Cleaner Production, 19(16), 1841-1856.

Comparing the environmental impact of human diets varying in amount of animal-source food – the impact of accounting for nutritional quality

Heleen R.J. van Kernebeek*, Simon J. Oosting, Imke J.M. de Boer

Animal Production Systems Group, Wageningen University, The Netherlands
* Corresponding author. E-mail: *Heleen.vankernebeek@wur.nl*

ABSTRACT

Studies comparing the environmental impact of human diets that vary in amount of animal-source food generally not account for the nutritional quality of these diets. Because diets may differ in amount of energy and nutrients, the impacts of these diets are not comparable. To make impacts comparable, we quantified the amount of 12 nutrients in diets described in 13 peer reviewed studies. We also computed their composite nutritional quality, expressed as the Nutrient Rich Diet9.3 score (NRD9.3). We expressed the GWP per unit protein and per unit NRD9.3. Diets that had higher levels of animal protein had higher (excessive) levels of total protein and were generally lower in composite nutritional quality. Diets that had lower levels of animal protein had lower GWPs per gram protein and per unit NRD9.3. Accounting for composite nutritional quality gave stronger contrasts in GWP between human diets than original comparisons.

Keywords: GWP, human diets, protein, nutritional quality, NRD9.3

1. Introduction

The production and consumption of animal-source food products (ASFP) has a high environmental impact and leads to high resource use (Cordell et al., 2009; Steinfeld, 2006). Choosing a diet low in ASFP could mitigate environmental impacts and improve resource use efficiency. Several studies assessed the environmental impact of human diets that varied in their ASFP concentration (Carlsson-Kanyama and Gonzalez, 2009; Davis et al., 2010; Saxe et al., 2012). They generally compared impacts of two or more diet scenarios using a life cycle perspective. Life cycle assessment (LCA) is an holistic approach to assess the environmental impact during the entire production chain. These studies support the general conclusion that plant-based diets have a lower environmental impact than animal-based diets.

To compare environmental impacts of different diet scenarios, the environmental impact should be expressed on the basis of a so-called functional unit (FU). A FU depends on the main function of the system, and the primary function of food production is to satisfy the human body's need for nutrients such as kcal, protein, fibre, vitamins and minerals. Studies generally use daily intake or energy, protein or fat content of the diet as functional units. These elements, however, only partly define the nutritional quality of a diet. Smedman et al., (2010), therefore, accounted for the content of 21 nutrients, when comparing emissions of greenhouse gases along the life cycle of various beverages. The FU that Smedman et al., (2010) used, was the nutrient density (NDS). The NDS of a food product is a nutrient over energy ratio representing the composite nutrient score of a product (Hansen, 1973). In the present paper we developed another composite nutrient score at the diet level, and used this score as a FU in the comparison of the environmental impact of human diets.

The main aim of this paper was to evaluate whether the comparison of environmental impacts among diets that varied in their ASFP-concentration, as reported in literature, would be different when nutritional quality was accounted for. We reviewed 13 published studies that assessed the environmental impact of human diets that varied in their ASFP-concentration.

2. Methods

2.1. Selection of published studies

We found 13 studies that met the following five selection criteria: the study contained more than one national diet scenario; diets within studies varied in ASFP-concentration; the weight of each food product included in the diets was given; diets were not designed for specific groups (e.g. infants, people with health problems or a specific gender); the article is published in a peer-reviewed scientific journal.

2.2. Calculation of individual nutrient scores of each diet

We quantified the daily intake of nine qualifying and three disqualifying nutrients in the diets. The nine qualifying nutrients were (the recommended daily value (RDV) is given in brackets): protein (57 g), fibre (25 g), calcium (800 mg), iron (14 mg), magnesium (375 mg), potassium (2000 mg), and vitamins A (800 µg), C (80 mg) and E (12 mg). The three disqualifying nutrients were (maximum recommended value (MRV) given in brackets): sodium (2400 mg), saturated fat (20 g) and total sugar (90 g) (efsa, 2009, 2010, 2012; EuropeanUnion, 2008). To quantify the daily intake of these 12 nutrients, we multiplied the daily intake of each food product in the diet by the nutrient concentration of the food product. The nutrient concentration of food products were taken from the online Dutch Nutrients Database NEVO (RIVM, 2011). Nutrient intake of meals was scaled to daily intake by expressing it relative to a recommended daily energy intake of 2000 kcal (efsa, 2009). Yearly diets were scaled to daily diets by dividing the intake of nutrients by 365.

To evaluate the nutritional quality of each diet in terms of individual nutrients, we compared the individual scores of qualifying and disqualifying nutrients to the RDV and MRV respectively.

2.3. Calculation of the composite nutrient score of each diet

We developed the Nutrient Rich Diet 9.3 (NRD9.3) score to calculate the composite nutrient score of each diet. The NRD9.3 score is an adaptation of the Nutrient Rich Food 9.3 (NRF9.3) algorithm (Drewnowski, 2009), the latter reflecting the nutrient density of a given food product per 100 kcal. In contrast to the NRF9.3 algorithm, the NRD9.3 algorithm is energy independent.

The NRD9.3 score consists of a Total Nutrient Rich9 (TNR9) and a Total Limiting3 (TLIM3) subscore. The TNR9 subscore (Eq. 1) computes the percentages of RDV for the nine qualifying nutrients. Intake levels of these nutrients were capped at 100% of their RDV. The TLIM3 subscore (Eq. 2) computes the percentages of MRV for the three disqualifying nutrients. The NRD9.3 score (Eq. 3) of each diet was computed by subtracting the TLIM3 subscore from the TNR9 subscore.

$$TNR9 = \sum_{i=1}^{i=9} \frac{nutrient_{i,capped}}{RDV_i} \times 100 \qquad \text{Eq. 1}$$

$$TLIM\,3 = \sum_{i=1}^{i=3} \frac{nutrient_i}{MDV_i} \times 100 \qquad \text{Eq. 2}$$

$$NRD9.3 = TNR9 - TLIM\,3 \qquad \text{Eq. 3}$$

where $nutrient_i$ is the amount (in g or mg or µg) of nutrient i in the diet, RDV_i is the Recommended Daily Value of nutrient i, MDV_i is the Maximum Daily Value of nutrient i.

2.4. Comparison of the environmental impact of diets

Subsequently, we computed the Global Warming Potential (GWP) of each diet per FU. The impact of the diets were reported in the reviewed articles. The impact derived from the reviewed articles was expressed as GWP/day. In the present paper we compared this expression with GWP relative to the FUs ‘protein’ and ‘NRD9.3’.

Because overconsumption of qualifying nutrients lessens the environmental impact per unit of nutrient, we explored the effect of capping qualifying nutrient levels to 100% of their RDV. We used this capping also to compute the NRD9.3 score, which is based on capped scores for qualifying nutrients and uncapped scores for disqualifying nutrients. The NRD9.3 score thus does not give credits to overconsumption of qualifying nutrients, whereas it does account for overconsumption of disqualifying nutrients.

To test whether the environmental impacts of the FUs ‘protein uncapped’, ‘protein capped’ and ‘NRD9.3’ differed from the FU ‘day’, which was reported in the reviewed articles, we calculated the indexed GWP/FU for each study and for each FU by setting GWP/FU to 100 for a diet without ASFPs. We regressed the indexed GWP/FU to the fractional content of ASFP in the diet (assessed by animal source protein/total dietary protein). By calculating the difference between the regression coefficient for ‘GWP/day’ and GWP per FUs ‘protein uncapped’, ‘protein capped’ and ‘NRD9.3’ and testing (by t-test) whether this difference differed significantly from 0, we could conclude whether our functional units gave a different contrast between dietary scenarios than the FU 'day'. Differences with a *p*-value less than 0.05 were considered to be statistically significant.

3. Results

3.1. Characteristics of diets in selected studies

We found 13 studies that met our selection criteria (Carlsson-Kanyama, 1998; Carlsson-Kanyama et al., 2003; Carlsson-Kanyama and Gonzalez, 2009; Collins and Fairchild, 2007; Davis and Sonesson, 2008; Davis et al., 2010; Gerbens-Leenes and Nonhebel, 2002; Pathak et al., 2010; Peters et al., 2007; Risku-Norja et al., 2008; Risku-Norja et al., 2009; Saxe et al., 2012; Thibert and Badami, 2011).

The size of the diets described in these studies varied from single meals to yearly per capita diets. Meals contained a median of five food products, whereas daily diets contained a median of 23 products. Some of the diets are representative for a country or region, such as the average Finnish diet (Risku-Norja et al., 2008). Other diets are self-defined alternatives, such as the pork and poultry-free diet (Risku-Norja et al., 2008). The contribution of animal protein in the diets differed from 0% in the various vegan scenarios to 90%.

3.2. Nutritional quality of meals and diets

Meals are not representative for a daily diet. In case we would scale-up a meal to a daily diet, i.e. scaling up to a 2000 kcal diet, we observed vitamin A intake levels of over 1000% of RDV. Because the meals contain a relatively small number of food products, the individual nutrient scores are relatively sensitive to product choice. Because meals cannot be made representative for a daily consumption, further computations focussed exclusively on daily diets and not on meals.

Fig. 1 shows the protein levels of daily diets. The scenarios were clustered per study (indicated by capital letters at the horizontal bar). Within study, the scenarios were ranked by their fractional content of animal-source protein, ranging from 0 to 78%. The horizontal line indicates the RDV of protein of 57 g. Most studies have one blank bar, which represents the average/actual scenario within the studied country of region. Total protein content ranged from 54 g (94% of RDV) in the vegan scenarios by Risku-Norja et al., (2008) to 150 grams (263% of RDV) (Peters et al., 2007). Among the scenarios within study, the total (excess) content of protein generally increased with increasing fractional content of animal protein.

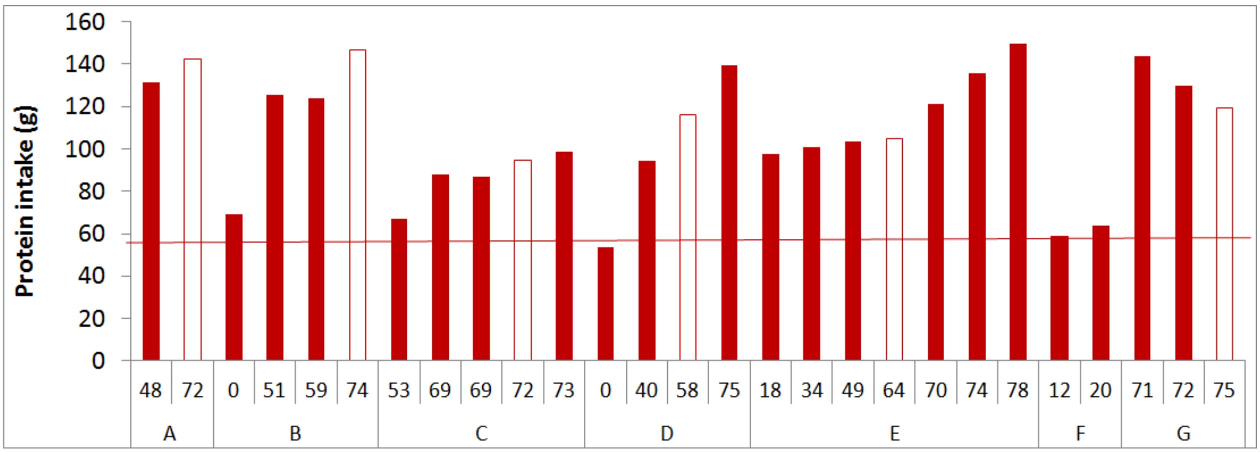


Figure 1. Daily intake of protein. A: Gerbens-Leenes and Nonhebel, (2002); B: Risku-Norja et al., (2009); C: Collins and Fairchild, (2007); D:Risku-Norja et al., (2008); E: Peters et al., (2007); F: Pathak et al., (2010); G: Saxe et al., (2012).

The NRD9.3 score was generally lower for diets that had a higher fractional content of animal protein (Fig. 2). This decrease is due especially to higher levels of sodium, saturated fat and total sugar in diets that had higher fractional contents of animal protein. Although diets which contain a lower fractional content of animal protein have higher excess levels of fibre, diets were not rewarded for this due to the applied capping.

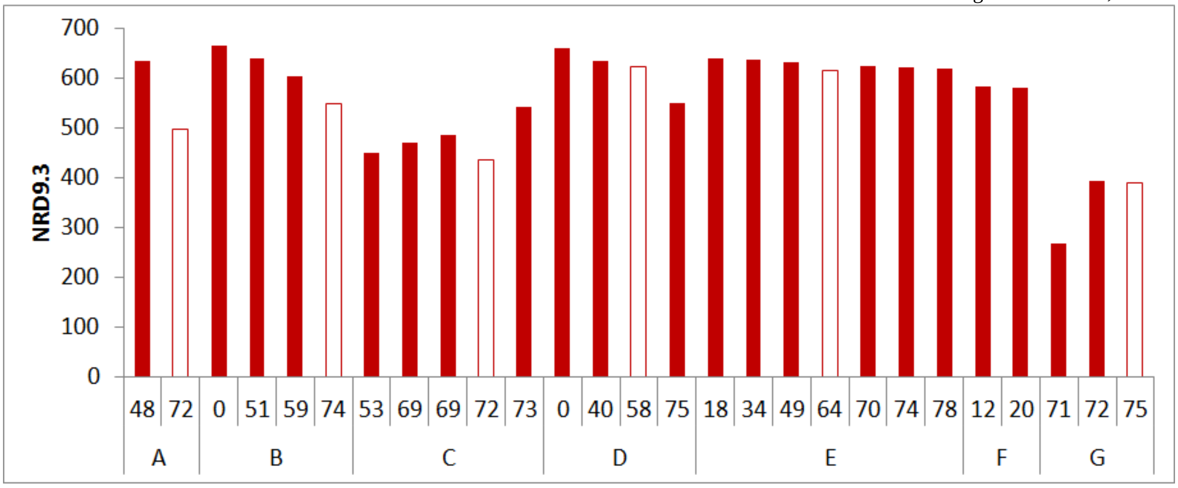


Figure 2. NRD9.3 scores of daily diets. Figure 2 has the same format as Fig. 1. A: Gerbens-Leenes and Non-hebel, (2002); B: Risku-Norja et al., (2009); C: Collins and Fairchild, (2007); D:Risku-Norja et al., (2008); E: Peters et al., (2007); F: Pathak et al., (2010); G: Saxe et al., (2012)

3.3. Global warming potential of diets

Four studies evaluated the GWP of daily diets (Pathak et al., 2010; Risku-Norja et al., 2008; Risku-Norja et al., 2009; Saxe et al., 2012). Within Risku-Norja et al., 2009 and 2008 we distinguish between diets containing food products from conventional (A) and organic (B) agriculture. In Figure 3 we show the indexed GWP per FU. In the original studies, the GWP per day was lower for diets that had lower contributions of animal protein. The same is true for the indexed GWP per 100 g of 'protein capped' and per unit of 'NRD9.3'. The GWP per 100 grams of 'protein uncapped' was generally higher for diets that had lower contributions of animal protein. This is due to the higher (excessive) amount of protein within diets that contain more animal protein (Fig.1). Overall, the regression coefficient for the FU 'day' was significantly higher compared to the regression coefficient for the FU 'protein uncapped' ($p<0.05$), and was significantly lower compared to the regression coefficient for the FU 'NRD9.3' ($p<0.05$).

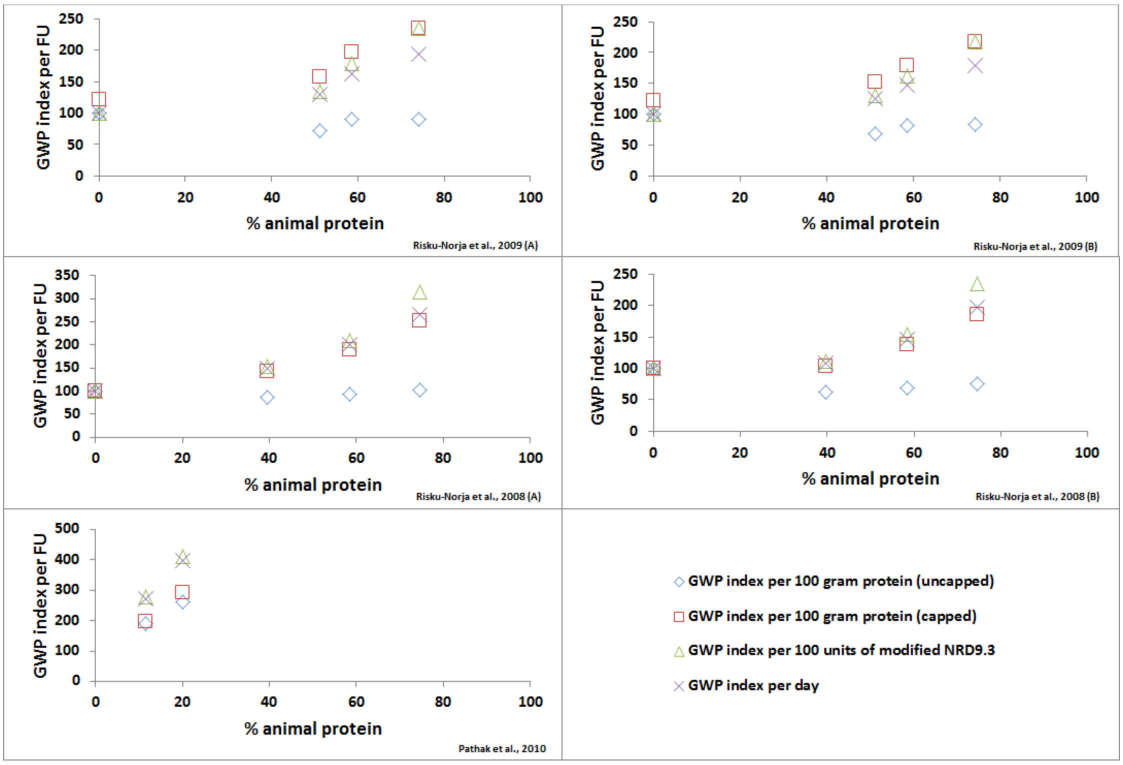


Figure 3. GWP index per functional unit (FU). Index 100 = 0% animal protein. A: conventional food products; B: organic food products

4. Discussion

The number of studies included in our environmental analysis was limited. Besides, two of the three studies were from the same author. We found, however, that meals are not a good basis for comparison of the environmental impact of human diets, as meals are not representative for daily consumption. Because meals contain a limited number of food products, the amount of quantified nutrients in meals is also more sensitive to product choice compared to the quantified amount of nutrients in daily or yearly diets.

The protein quality of a food product can be evaluated using the Protein digestibility-corrected amino acid score (PDCAAS) (Hughes et al., 2011). Because questions were raised concerning the usability of the PDCAAS method for assessing protein quality at a diet level (FAO/WHO, 1991; Schaafsma, 2000), we did not account for differences in protein quality between plant and animal based products.

5. Conclusion

Current comparisons of the global warming potential (GWP) of human diets show that diets that have higher levels of animal-source food products have higher GWP compared to diets that have lower levels of animal-source food products. However, these comparisons do not account for differences in the nutritional quality of these diets. The environmental impact of these diets is expressed relative to the functional unit (FU) ‘day’. In this review, we computed the nutritional quality of diets and evaluated the effect of accounting for nutritional quality when comparing the environmental impacts of diets that vary in their amount of animal-source food products.

We concluded that diets with higher levels of animal-source food products have higher (excess) contents of protein and generally lower composite nutritional quality compared to diets that are lower in animal-source food products.

We evaluated the effect of using the FUs ‘protein intake’ and ‘composite nutritional quality’. When we expressed GWP with respect to grams of protein, we found that GWP was lower for diets that had higher levels of animal-source food products. This lower GWP per gram of protein in diets with higher levels of animal-source food products was due to the higher (excess) intake levels of total protein in these diets. To avoid credit for overconsumption, we capped the protein intake levels at 100% of the Daily Recommended Value and found that GWP per unit ‘protein capped’ was in fact higher for diets that had higher levels of animal-source food products. When we expressed GWP in terms of the composite nutritional quality, i.e. using ‘NRD9.3’ as FU, we found that GWP per unit NRD9.3 was higher for diets that had higher levels of animal-source food products.

Overall, the regression coefficient for the FU ‘day’ was significantly higher compared to the regression coefficient for the FU ‘protein uncapped’, and was significantly lower compared to the regression coefficient for the FU ‘NRD9.3’. Not crediting for overconsumption of protein and accounting for overall nutritional quality thus gives a stronger contrast in the GWP between diets that vary in their amount of animal-source food products.

6. References

Carlsson-Kanyama, A., 1998. Climate change and dietary choices - how can emissions of greenhouse gases from food consumption be reduced? Food Policy 23, 277-293.

Carlsson-Kanyama, A., Ekstrom, M.P., Shanahan, H., 2003. Food and life cycle energy inputs: consequences of diet and ways to increase efficiency. Ecological Economics 44, 293-307.

Carlsson-Kanyama, A., Gonzalez, A.D., 2009. Potential contributions of food consumption patterns to climate change. American Journal of Clinical Nutrition 89, S1704-S1709.

Collins, A., Fairchild, R., 2007. Sustainable food consumption at a sub-national level: An ecological footprint, nutritional and economic analysis. Journal of Environmental Policy and Planning 9, 5-30.

Cordell, D., Drangert, J.O., White, S., 2009. The story of phosphorus: Global food security and food for thought. Global Environmental Change-Human and Policy Dimensions 19, 292-305.

Davis, J., Sonesson, U., 2008. Life cycle assessment of integrated food chains - A Swedish case study of two chicken meals. Int J Life Cycle Assess 13, 574-584.

Davis, J., Sonesson, U., Baumgartner, D.U., Nemecek, T., 2010. Environmental impact of four meals with different protein sources: Case studies in Spain and Sweden. Food Research International 43, 1874-1884.

Drewnowski, A., 2009. Defining Nutrient Density: Development and Validation of the Nutrient Rich Foods Index. Journal of the American College of Nutrition 28, 421S-426S.

efsa, 2009. Scientific Opinion of the Panel on Dietetic products, Nutrition and Allergies on a request from European Commission on the review of labelling reference intake values for selected nutritional elements. The EFSA, 1-14.

efsa, 2010. EFSA Panel on Dietetic Products, Nutrition, and Allergies (NDA); Scientific Opinion on Dietary Reference Values for carbohydrates and dietary fibre. EFSA Journal 3, 1462 [77 pp.].

efsa, 2012. EFSA Panel on Dietetic Products, Nutrition and Allergies (NDA); Scientific Opinion on Dietary Reference Values for protein. EFSA Journal 2, 2557 [66 pp.].

EuropeanUnion, 2008. COMMISSION DIRECTIVE 2008/100/EC of 28 October 2008 amending Council Directive 90/496/EEC on nutrition labelling for foodstuffs as regards
recommended daily allowances, energy conversion factors and definitions, Official Journal of the European Union. pp. 9-12.
FAO/WHO, 1991. Protein quality evaluation. Report of joint FAO/WHO Expert Consultation Bethesda, Md., USA 4-8 December 1989. , FAO Food and Nutrition Paper 51. . FAO,, Rome.
Gerbens-Leenes, P.W., Nonhebel, S., 2002. Consumption patterns and their effects on land required for food. Ecological Economics 42, 185-199.
Hansen, R.G., 1973. An Index of Food Quality. Nutrition Reviews 31, 1-7.
Hughes, G.J., Ryan, D.J., Mukherjea, R., Schasteen, C.S., 2011. Protein Digestibility-Corrected Amino Acid Scores (PDCAAS) for Soy Protein Isolates and Concentrate: Criteria for Evaluation. Journal of Agricultural and Food Chemistry 59, 12707-12712.
Pathak, H., Jain, N., Bhatia, A., Patel, J., Aggarwal, P.K., 2010. Carbon footprints of Indian food items. Agriculture Ecosystems & Environment 139, 66-73.
Peters, C.J., Wilkins, J.L., Fick, G.W., 2007. Testing a complete-diet model for estimating the land resource requirements of food consumption and agricultural carrying capacity: The New York State example. Renewable Agriculture and Food Systems 22, 145-153.
Risku-Norja, H., Hietala, R., Virtanen, H., Ketomaki, H., Helenius, J., 2008. Localisation of primary food production in Finland: production potential and environmental impacts of food consumption patterns. Agricultural and Food Science 17, 127-145.
Risku-Norja, H., Kurppa, S., Helenius, J., 2009. Dietary choices and greenhouse gas emissions - Assessment of impact of vegetarian and organic options at national scale. Prog. Ind. Ecol. 6, 340-354.
RIVM, 2011. NEVO table 2011. National Institute for Public Health and the Environment,, the Netherlands.
Saxe, H., Larsen, T.M., Mogensen, L., 2012. The global warming potential of two healthy Nordic diets compared with the average Danish diet. Clim. Change, 1-14.
Schaafsma, G., 2000. The protein digestibility-corrected amino acid score. Journal of Nutrition 130, 1865S-1867S.
Smedman, A., Lindmark-Månsson, H., Drewnowski, A., Edman, A.K.M., 2010. Nutrient density of beverages in relation to climate impact. Food Nutri. Res. 54.
Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., Haan de, C., 2006. Livestock's long shadow. Environmental issues and options. FAO, Rome.
Thibert, J., Badami, M.G., 2011. Estimating and communicating food system impacts: A case study in Montreal, Quebec. Ecological Economics 70, 1814-1821.

A novel nutrition-based functional equivalency metric for comparative life cycle assessment of food

Martin C. Heller*, Gregory A. Keoleian

Center for Sustainable Systems, University of Michigan, Ann Arbor, MI, USA
* Corresponding author. E-mail: *mcheller@umich.edu*

ABSTRACT

Establishing functional equivalency between disparate food types remains a methodological challenge for comparative environmental impact studies. In this paper, we demonstrate the influence of functional unit choice on the life cycle energy and greenhouse gas emissions for a variety of food types. We introduce a novel functional unit approach utilizing the NuVal nutritional quality scoring system. NuVal combines over 30 micro- and macro-nutrient properties of foods into a single score weighted on the basis of the effects that nutrients have on health. "Nutritional weighting" of environmental impact scores expressed per unit of food energy clearly influences the ranking of foods based on their environmental impact. The nutritionally weighted environmental impact indicator provides a useful means to holistically account for nutritional quality in comparing and differentiating foods using life cycle assessment.

Keywords: functional unit, nutrition, health, ONQI, NuVal, nutritional quality index

1. Introduction

Establishing functional equivalency has always been a methodological challenge in life cycle assessment studies of foods. Food LCAs often use the system reference flow (mass or volume) as the functional unit (Schau and Fet, 2008). Functional units are chosen to serve the goal and scope of the study (ISO, 2006), and often a mass- or volume-based functional unit is sufficient for a stand-alone system assessment. However, this reference flow basis does not fully capture the primary function of foods – delivering nutrition – making comparisons between different food types in consumer-oriented analyses difficult. Water content of foods can also play an important role with mass- or volume-based functional units as water adds mass or volume without effecting quality such as nutrient or energy content. Numerous approaches have been put forth to address this food functional equivalency challenge. Quality-based correction factors such as "energy corrected milk" (e.g., Cederberg & Mattsson, 2000) or "protein corrected wheat" (e.g., Audsley, 2003) assist in normalizing natural variations in qualities that influence value and processing. Martinez-Blanco, et al., (2011) use antioxidant compound content as a functional unit when comparing cultivation and fertiliser options in cauliflower production. In a recent literature review, de Vries and de Boer (2010) compare impacts of livestock products on the functional unit basis of "protein delivered" and "average daily intake," allowing comparison across disparate food types. Such single nutrient based functional units have merit for particular study goals, but the fact remains that healthy nutrition is complex and multi-dimensional. Schau and Fet (2008) propose the need for a quality corrected functional unit that incorporates fat, protein and carbohydrate content, as well as potentially other quality functions, as deemed necessary.

The Overall Nutritional Quality Index (ONQI) is a tool designed to aid consumers in making well-informed dietary choices (Katz et al., 2010). Developed by nutrition and public health experts, the ONQI algorithm combines into a single score over 30 entries representing both micro- and macronutrient properties of foods, weighted on the basis of the effects (both promotional and detrimental) that nutrients have on health. The index has been validated through expert panel rankings and statistical diet comparisons, and has been implemented in over 500 supermarkets across the US as NuVal.

In this paper, we explore the utility of ONQI as a nutrition-based functional equivalency metric in environmental impact assessments of food. Using the NuVal score (ONQI adjusted to values from 1-100) as a nutritional weighting factor allows comparisons of environmental impacts of foods on the basis of their relative contribution to a healthy diet. These results are contrasted with results based on single-dimension functional units (e.g., weight, serving, kg protein, and energy content).

2. Methods

2.1. Food nutritional information

Nutritional and serving size information for the variety of foods found in Table 1 were from the USDA Food and Nutrient Database for Dietary Studies, v 5.0 (Ahuja et al., 2012), accessed via the "What's In The Foods You Eat" Search Tool v. 5.0 (USDA, 2012). Nutritional data are for foods as listed in Table 1. These were matched as closely as possible to LCA and NuVal data.

2.2. Collection of life cycle based environmental impacts of foods

Estimates of greenhouse gas emissions (GHG) (kg CO_2e/ kg food) and energy use (MJ/kg food) were taken from the life cycle inventories aggregated and reported by Gonzalez, et al., 2011. Gonzalez, et al., present LC results on the basis of 1 kg of food product delivered to the entry port of Gothenburg, Sweden; meat products are on a bone-free carcass basis, while cereals and beans are reported as dry grain at the port. Arithmetic means were taken when multiple entries (representing different countries of origin) were reported for the same food type. Values for vegetables reflect impacts of open-field production rather than production in heated greenhouses. Values for salmon represent farmed salmon.

A recognized discrepancy in the water content of rice and dry beans between the cooked state as provided by the USDA Food and Nutrient database and the dry state "as delivered to the entry port" used by Gonzalez et al., was corrected by assuming that dry rice and beans contain 10% moisture (USDA, 2011). All other foods are assumed to have moisture content corresponding to that reported by the Food and Nutrient database (Table 1). While there is likely a discrepancy in water content for meats between the carcass basis and cooked basis, it is assumed that this is relatively small (<5%) and has not been accounted for in this demonstration study.

2.3. Overall Nutritional Quality Index and NuVal scores

Development of the ONQI algorithm is described in detail in Katz et al., 2009. In brief, nutrients with generally favorable effects on health were placed in the numerator of the ONQI algorithm; these include: fiber, folate, vitamin A, vitamin C, vitamin D, vitamin E, vitamin B12, vitamin B6, potassium, calcium, zinc, omega-3 fatty acids, total bioflavonoids, total carotenoids, magnesium, and iron. Nutrients with generally unfavorable effects were placed in the denominator, and include: saturated fat, trans fat, sodium, total added sugar, and cholesterol. In addition, macronutrient factors accounted for in the algorithm include fat quality and protein quality (applied to the numerator) and energy density and glycemic load (applied to the denominator). To account for an individual food's nutrient content relative to overall dietary needs, the ONQI algorithm incorporates a "trajectory score" (TS), which indicates how intake of a given nutrient moves the total dietary intake toward, or away from, a recommended threshold. TS is expressed relative to the caloric content of the food, i.e., TS = [(nutrient dose$_i$/kcal in food item) / (recommended nutrient dose$_i$/kcal in diet)]. In statistical testing, ONQI has correlated well with expert panel rankings as well as the Healthy Eating Index when applied to the National Health and Nutrition Examination Survey (NHANES) 2003-2006 populations (Katz et al., 2010).

NuVal scores (ONQI normalised to a 1-100 scale) were collected from grocery store displays or provided as averages with associated ranges by NuVal, LLC (NuVal, 2012) for the variety of foods listed in Table 1. While the ONQI score is unbundled from serving size, the inclusion of TS in essence makes the score relative to caloric content; i.e., a food with a higher NuVal score has a greater nutrient value *relative to its food energy content*. A "nutritionally weighted" environmental indicator was therefore calculated by dividing the NuVal score (normalised to values from 0-1) into environmental indicators *per kcal of food energy*, as in Eq. 1 (an analogous expression to Eq. 1 is used for "nutritionally weighted" GHG emissions).

$$\frac{\text{MJ}}{\text{nutritional kcal}*} = \frac{\dfrac{\text{MJ}}{\text{kcal food energy}}}{\text{NuVal}/100} \qquad \text{Eq. 1}$$

3. Results

Table 1 summarises the nutritional and environmental impact data used for the foods considered in this study. Tables 2 and 3 present the life cycle energy use and GHG emissions, respectively, on a number of different functional unit bases including dry weight basis, typical serving basis, delivered protein basis, food energy basis, and the newly introduced NuVal "nutritional weighting" applied to a food energy basis. Tables 2 and 3 allow quick comparisons of the effect on ranking of changes in functional unit (impact rankings from high to low are shown in parentheses in Tables 2 and 3). For example, Table 2 shows farmed salmon to have the highest life cycle energy use per dry weight and per unit of food energy, but because of its high NuVal score (87), salmon drops to a ranking of 5 on a "nutritionally weighted" basis. Table 3 shows beef to have the highest GHG emissions regardless of functional unit (presumably due to an enteric fermentation contribution), whereas cheese rises from a ranking of 7th on a food energy basis to a ranking of 3rd on a "nutritionally weighted" basis. Fig. 1 shows the influence of the NuVal "nutritional weighting" on life cycle energy and GHG emissions. Fruits and vegetables with NuVal scores approaching 100 are not affected by the weighting whereas most animal-based foods show a marked increase due to the nutritional weighting. Brown rice and white rice, which are assumed to have the same environmental impact per kg, are differentiated by the nutritional weighting, while skim and whole milk, also assumed to have the same environmental impact per kg (as consumed), show a difference per kcal food energy (due to the caloric content of the fat in whole milk) that is largely negated by the nutritional weighting (again, presumably due to negative effect of whole milk's fat content on its NuVal score.)

Table 1. Nutritional and environmental impact data

Abbreviation used in paper	Food description per USDA nutrition data[a, d]	USDA nutrition data (per serving)[a]				Life Cycle environmental impact[b]		NuVal as displayed in grocer
		g/ serving	g water/ serving	kcal/ serving	g protein/ serving	MJ/kg	kg CO_2e/ kg	
beef	ground beef, 85%-89% lean, cooked	87	52.3	200	21.25	44.63	29.29	32 (23-37)[e]
lamb	lamb, ground, cooked	77	42	216	18.9	46.00	25.67	27 (24-28)[e]
pork	pork, ground or patty, cooked	85	44.46	251	21.66	28.00	8.20	37 (16-39)[e]
chicken	chicken, ground	85	50.12	201	23.01	26.67	4.75	37 (16-39)[e]
tuna	tuna, canned, water pack	85	63.3	99	21.68	26.00	2.60	58
salmon	salmon, raw	57	41.42	83	12.32	45.00	3.27	87
cheese	cheese, natural, cheddar or American type	28	10.29	113	6.97	41.80	8.60	23
skim milk	milk, Ca fortified, cow's, fluid, skim or nonfat	247	224.28	86	8.4	3.05	1.10	81
whole milk	milk, cow's, fluid, whole	244	215.04	149	7.69	3.05	1.10	52
egg	egg, whole, raw	50	38.08	72	6.28	14.33	3.00	33
brown rice	rice, brown, cooked, regular, fat not added in cooking	131	95[c]	144	3.35	7.87[c]	1.20[c]	82
white rice	rice, white, cooked, regular, fat not added in cooking	105	71.19[c]	135	2.79	7.87[c]	1.20[c]	48
dry beans	beans, dry, cooked, NS as to type, fat not added in cooking	88	58.2[c]	111	7.65	2.90[c]	1.00[c]	93
apple	apple, raw	138	118.07	72	0.36	3.56	0.28	96
orange	orange, raw	131	113.64	62	1.23	3.75	0.33	100
strawberries	strawberries, raw	72	65.48	23	0.48	4.10	0.38	100
tomatoes	tomatoes, raw	123	116.26	22	1.08	3.35	0.33	96
potato	white potato, baked, peel eaten, fat not added in cooking	173	128.68	159	4.29	1.98	0.20	93
broccoli	broccoli, raw	44	39.29	15	1.24	3.60	0.37	100
lettuce	lettuce, raw	55	52.6	8	0.5	2.20	0.20	99 (82-100)[e]
winter squash	squash, winter type, baked, no fat or sugar added in cooking	103	91.46	38	0.92	0.96	0.09	94 (91-100)[e]
beets	beets, cooked, from fresh, NS as to fat added in cooking	88	75	50	1.43	1.10	0.11	99
cucumber	cucumber, raw	60	58.04	7	0.35	0.84	0.08	93
cabbage	cabbage, green, raw	45	41.48	11	0.58	1.10	0.12	99 (96-100)[e]
carrots	carrots, raw	28	24.72	11	0.26	1.34	0.12	99
onions	onions, mature, raw	14	12.48	6	0.15	1.00	0.10	93 (91-100)[e]

[a] USDA, 2012
[b] Gonzalez, et al., 2011
[c] corrected discrepancy between cooked moisture content and "as delivered to port" moisture content used by Gonzalez, et al., (assumed to be 10%). See Methods for correction details.
[d] "NS" in this table stands for "not specified" and refers to a lack of specificity in describing the food analysed for nutritional content.
[e] these scores are averages of a collection of food products, as provided directly by NuVal LLC. Values within parentheses represent minimum and maximum NuVal scores for the given collection of food products.

4. Discussion

Functional unit choice clearly influences the outcome in a comparative study of the environmental impact of different foods, as is shown in Tables 2 and 3. This influence has been demonstrated previously: Gonzalez et al., 2011, explore the protein delivery efficiency of various foods (the inverse of environmental

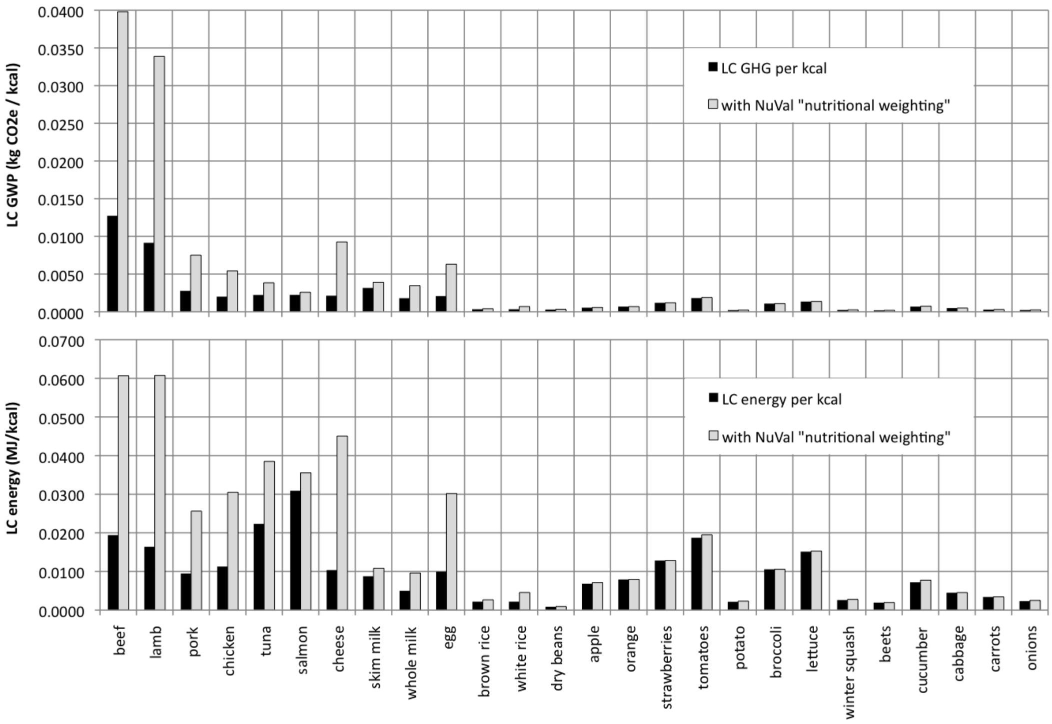


Figure 1. Influence of a NuVal "nutritional weighting" on the life cycle (LC) greenhouse gas emissions and energy use per kcal of food energy for a variety of foods.

Table 2. Life Cycle Energy Use for a variety of foods, expressed on different functional unit bases. Values in parentheses are the ranking (high to low impact) of foods in each column.

	per dry weight	per serving	per g protein	per kcal food energy	per kcal w/ NuVal "nutritional weighting"
	MJ/ kg DW	MJ/ serving	MJ/ g protein	MJ/ kcal food energy	MJ/ nutritional kcal*
beef	111.88 (2)	3.88 (1)	0.18 (9)	0.019 (3)	0.061 (2)
lamb	101.20 (4)	3.54 (2)	0.19 (7)	0.016 (5)	0.061 (1)
pork	58.71 (9)	2.38 (4)	0.11 (14)	0.009 (12)	0.026 (8)
chicken	64.98 (6)	2.27 (5)	0.10 (18)	0.011 (8)	0.030 (6)
tuna	101.84 (3)	2.21 (6)	0.102 (17)	0.022 (2)	0.038 (4)
salmon	164.63 (1)	2.57 (3)	0.21 (6)	0.031 (1)	0.036 (5)
cheese	66.09 (5)	1.17 (7)	0.17 (9)	0.010 (10)	0.045 (3)
skim milk	33.16 (13)	0.75 (8)	0.090 (22)	0.009 (13)	0.011 (12)
whole milk	25.70 (16)	0.74 (9)	0.10 (19)	0.005 (17)	0.010 (14)
egg	60.12 (8)	0.72 (10)	0.11 (13)	0.010 (11)	0.030 (7)
brown rice	8.74 (21)	0.31 (15)	0.094 (20)	0.002 (23)	0.003 (22)
white rice	8.74 (22)	0.30 (16)	0.11 (16)	0.002 (22)	0.005 (18)
dry beans	3.22 (26)	0.10 (22)	0.013 (26)	0.001 (26)	0.001 (26)
apple	24.66 (17)	0.49 (11)	1.37 (1)	0.007 (16)	0.007 (17)
orange	28.30 (14)	0.49 (12)	0.40 (3)	0.008 (14)	0.008 (15)
strawberries	45.28 (11)	0.30 (17)	0.62 (2)	0.013 (7)	0.013 (11)
tomatoes	61.14 (7)	0.41 (13)	0.38 (4)	0.019 (4)	0.020 (9)
potato	7.73 (24)	0.34 (14)	0.080 (24)	0.002 (24)	0.002 (24)
broccoli	33.63 (12)	0.16 (18)	0.13 (12)	0.011 (9)	0.011 (13)
lettuce	50.42 (10)	0.12 (19)	0.24 (5)	0.015 (6)	0.015 (10)
winter squash	8.57 (23)	0.10 (20)	0.11 (15)	0.003 (20)	0.003 (21)
beets	7.45 (25)	0.10 (21)	0.068 (25)	0.002 (25)	0.002 (25)
cucumber	25.71 (15)	0.050 (23)	0.14 (10)	0.007 (15)	0.008 (16)
cabbage	14.06 (18)	0.050 (24)	0.085 (23)	0.005 (18)	0.005 (19)
carrots	11.40 (19)	0.037 (25)	0.14 (11)	0.003 (19)	0.003 (20)
onions	9.21 (20)	0.014 (26)	0.093 (21)	0.002 (21)	0.003 (23)

impact per g protein); deVries and de Boer, 2010 consider the environmental impact of animal-based foods on a per kg protein and a per average daily intake basis. In this paper, we introduce an attempt to incorporate a more nutritionally holistic functional unit. The NuVal nutrition indicator has been applied as a weighting factor to environmental impacts per food energy content because the NuVal algorithm considers nutrient content relative to food energy. In essence, the "nutritionally weighted" functional unit suggests that not all food calories are created equal: depending on other nutritional components, food calories can contribute more or less toward a healthy diet. As can be seen in Figure 1, the carbon footprint of pork, chicken, tuna, salmon, cheese, milk and eggs *on a per kcal basis* are relatively similar, but applying the NuVal based "nutritional weighting" clear differentiates these foods on a per *nutritional* kcal basis.

The influence of a "nutritionally weighted" functional unit is perhaps clearer in more extreme comparisons. Coca-Cola reports that their product delivered in 2 L plastic bottles has a carbon footprint of 0.25 kg CO2e/ L (Coca-Cola Co., 2010), which, on a caloric content basis, is a factor of 3 smaller than whole milk. Coca-Cola receives a NuVal score of 1, however, so on a nutritionally weighted basis, the carbon footprint of Coca-Cola is 17 times greater than that of whole milk.

5. Conclusion

We propose a functional unit that addresses a primary function of food, which is to deliver health-promoting nutrition. The NuVal score, which evaluates multiple nutritional properties of food, provides a convenient basis for more comprehensive comparisons of life cycle environmental sustainability performance across diverse food types.

Table 3. Life Cycle Greenhouse Gas Emissions for a variety of foods, expressed on different functional unit bases. Values in parentheses are the ranking (high to low impact) of foods in each column.

	per dry weight	per serving	per g protein	per kcal food energy	per kcal w/ NuVal "nutritional weighting"
	kg CO_2 eq/ kg DW	kg CO_2e/ serving	kg CO_2e/ g protein	kg CO_2e/ kcal food energy	kg CO_2 e/ nutritional kcal*
beef	73.43 (1)	2.55 (1)	0.12 (1)	0.01274 (1)	0.03981 (1)
lamb	56.47 (2)	1.98 (2)	0.10 (3)	0.00915 (2)	0.03389 (2)
pork	17.19 (3)	0.70 (3)	0.032 (10)	0.00278 (4)	0.00751 (4)
chicken	11.58 (8)	0.40 (4)	0.018 (13)	0.00201 (9)	0.00543 (6)
tuna	10.18 (9)	0.22 (8)	0.010 (20)	0.00223 (6)	0.00385 (8)
salmon	11.95 (7)	0.19 (9)	0.015 (15)	0.00224 (5)	0.00258 (10)
cheese	13.60 (4)	0.24 (7)	0.035 (8)	0.00213 (7)	0.00927 (3)
skim milk	11.96 (6)	0.27 (5)	0.032 (9)	0.00316 (3)	0.00390 (7)
whole milk	9.27 (10)	0.27 (6)	0.035 (6)	0.00180 (11)	0.00346 (9)
egg	12.58 (5)	0.15 (10)	0.024 (11)	0.00208 (8)	0.00631 (5)
brown rice	1.33 (20)	0.048 (11)	0.014 (16)	0.00033 (20)	0.00041 (20)
white rice	1.33 (19)	0.045 (12)	0.016 (14)	0.00033 (19)	0.00070 (16)
dry beans	1.11 (21)	0.033 (17)	0.004 (26)	0.00030 (21)	0.00032 (21)
apple	1.96 (17)	0.039 (15)	0.11 (2)	0.00054 (17)	0.00057 (18)
orange	2.45 (15)	0.043 (13)	0.035 (7)	0.00069 (15)	0.00069 (17)
strawberries	4.20 (13)	0.027 (18)	0.057 (4)	0.00119 (13)	0.00119 (13)
tomatoes	5.93 (11)	0.040 (14)	0.037 (5)	0.00182 (10)	0.00189 (11)
potato	0.79 (25)	0.035 (16)	0.008 (24)	0.00022 (25)	0.00024 (25)
broccoli	3.46 (14)	0.016 (19)	0.013 (18)	0.00109 (14)	0.00109 (14)
lettuce	4.51 (12)	0.011 (20)	0.022 (12)	0.00135 (12)	0.00137 (12)
winter squash	0.80 (24)	0.009 (22)	0.010 (21)	0.00024 (23)	0.00026 (23)
beets	0.74 (26)	0.010 (21)	0.007 (25)	0.00019 (26)	0.00020 (26)
cucumber	2.45 (16)	0.005 (24)	0.014 (17)	0.00069 (16)	0.00074 (15)
cabbage	1.53 (18)	0.005 (23)	0.009 (23)	0.00049 (18)	0.00050 (19)
carrots	0.98 (22)	0.003 (25)	0.012 (19)	0.00029 (22)	0.00030 (22)
onions	0.92 (23)	0.001 (26)	0.009 (22)	0.00023 (24)	0.00025 (24)

Of course, there are always inherent limitations to scoring systems that attempt to incorporate multiple attributes, and NuVal is no exception. The NuVal score is based on empirically derived weighting factors that currently remain confidential; thus, the nutritional factors that are the major drivers of the NuVal score are not immediately apparent.

This approach to establishing a nutritional equivalency metric for environmental impact studies of food is intended to stimulate and encourage discussion and further exploration of an important topic. Given limitations to the approach presented here, extreme care must be taken in fully communicating, and further considering, the meaning of a "nutritionally weighted kcal." Other similar nutritional indicators, such as the Nutrient Rich Foods Index (Drewnowski, 2010), may warrant consideration. Ultimately, it may be more beneficial to consider nutritional equivalency and environmental impact of foods by aggregating to the whole diet level. Given a diversity of research goals and the complexity of food and agricultural systems, there are likely many 'correct' answers to the question of food functional equivalency; our hope is that the approach presented here provides practitioners with an additional tool to consider.

6. References

Ahuja J.K.A., Montville J.B., Omolewa-Tomobi G., Heendeniya K.Y., Martin C.L., Steinfeldt L.C., Anand J., Adler M.E., LaComb R.P., Moshfegh A.J., 2012. USDA Food and Nutrient Database for Dietary Studies, 5.0. U.S. Department of Agriculture, Agricultural Research Service, Food Surveys Research Group, Beltsville, MD.

Audsley, E., 2003. Harmonisation of environmental life cycle assessment for agriculture: final report. Concerted Action AIR3-CT94-2028, European Commission DG VI Agriculture, Brussels.

Cederberg, C. Mattsson, B., 2000. Life cycle assessment of milk production – a comparison of conventional and organic farming. J. Cleaner Prod. 8, 49-60.

Coca-Cola Co., 2010. Coca-Cola, Great Britian/Environment. Website accessed: 5/25/2012. http://www.coca-cola.co.uk/environment/what-s-the-carbon-footprint-of-a-coca-cola.html.

De Vries, M., de Boer, I.J.M., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livestock Science. 128, 1-11.

Drewnowski, A., 2010. The Nutrient Rich Foods Index helps to identify healthy, affordable foods. Am. J. Clin. Nutr. 91, 1095S-1101S.

Gonzalez, A.D., Frostell, B., Carlsson-Kanyama, A., 2011. Protein efficiency per unit energy and per unit green house gas emissions: Potential contribution of diet choices toc limate change mitigation. Food Policy 36(5). 562-570.

ISO, 2006. Environmental management – life cycle assessment – principles and framework. ISO 14040.

Katz, D.L., Njike, V.Y., Faridi, Z., Rhee, L.Q., Reeves, R.S., Jenkins, D.J.A., Ayoob, K.T., 2009. The Stratification of Foods on the Basis of Overall Nutritional Quality: The Overall Nutritional Quality Index. Am. J. Health Promot. 24, 133-143.

Katz, D.L., Njike, V.Y., Rhee, L.Q., Reingold, A., Ayoob, K.T., 2010. Performance characteristics of NuVal and the Overall Nutritional Quality Index (ONQI). Am. J. Clin. Nutr. 91, 1102S-1108S.

Martinez-Blanco, J., Anton, A., Rieradevall, J., Castellari, M., Munoz, P., 2011. Comparing nutritional value and yield as functional units in the environmental assessment of horticultural production with organic or mineral fertilization. Int. J. LCA. 16(1), 12-26.

NuVal, LLC., 2012. Email communication, 5/22/2012. http://www.nuval.com/.

Schau, E. M., Fet, A. M., 2008. LCA Studies of Food Products as Background for Environmental Product Declarations. Int. J. LCA. 13, 255-264.

USDA, 2011. USDA National Nutrient Database for Standard Reference, Release 24. U.S. Department of Agriculture, Agricultural Research Service, Nutrient Data Laboratory Home Page, http://www.ars.usda.gov/nutrientdata

USDA, 2012. What's In The Foods You Eat Search Tool, v. 5.0. U.S. Department of Agriculture, Agricultural Research Service. http://www.ars.usda.gov/Services/docs.htm?docid=17032.

State of the art of LCA application in the fruit sector

Alessandro K. Cerutti[1], Simona Bosco[2], Gabriele L. Beccaro[1], Cristiana Peano[1], Bruno Notarnicola[3], Giancarlo Bounous[1]

[1] Department of Arboriculture, University of Torino, Via Leonardo da Vinci, 44 – 10095 Grugliasco (TO), Italy
[2] Institute of Life Sciences – Scuola Superiore S. Anna, Piazza Martiri della Libertà, 33 - 56127 Pisa, Italy
[3] II Faculty of Economics, Taranto, University of Bari Aldo Moro, Piazza Umberto I, 1 - 70121 Bari, Italy
[*] Corresponding author. E-mail: *alessandro.cerutti@unito.it*

ABSTRACT

This paper critically reviews LCA studies in fruit production and distribution systems as a part of the results of the work carried out by the Agri-Food working group of the "Rete Italiana LCA" (Italian LCA Network) – fruit products subgroup. Most used features of LCA application in the fruit sector are investigated in order to describe a specific research framework and to suggest standardisation in some specific aspects of the evaluation.

Keywords: fruit products, orchard, fruit cultivars, sustainable production, sustainable distribution, retail systems

1. Introduction

In 2010, world production of fruit was 609,213,512 t, mostly concentrated in Asia (52%) and America (22%) (FAOSTAT, 2012). In Europe, the same year, there were produced 67,254,709 t of fruit (about 11% of the fruit produced in the world), with significant contributions by Italy (25.14% of the fruit produced in Europe), Spain (22.57%) and France (12.93%). The important role played by the Asian markets is even more evident if we analyse the production trends of the last 10 years: while America, Europe, Africa and Oceania have a fairly constant fruit production, in Asia it has increased by about 55%, making China and India the highest producers of fruits in the world with respectively 20.06% and 13.92% of the world production.

Fruit products are generally considered to be some of the less environmental impacting foods in occidental diets. For example, Carlsson-Kanyama et al., (2003), quantified the energy consumption of different diets and evaluated an average of 5 MJ per kg of fruit in season (26 MJ per kg of fruit out of season), 15 MJ per kg of vegetables, 17 MJ per kg of bread and flour products, 33 MJ per kg of dairy products, 37 MJ per kg of meat, and 75 MJ per kg of fish products. Furthermore, in several works that focus on the carbon footprint of different food choices, it is reported that the fruit category is one with the least environmental impact (e.g. Wallén et al., 2004, Berners-Lee et al., 2012). However, these works consider results in environmental assessment of generic fruit productions, which takes no account of specific issues of orchard systems and fruit supply chains. Indeed, different results may arise considering the production protocol (e.g. conventional vs organic), the production site (specific soil and climatic conditions affect yields and agronomic performances) or the retailing system (long cold storage may dramatically influence the environmental performance of the product).

2. Methods

The modern food production has a great heterogeneity associated with high levels of specialisation and complexity. These features inevitably reflect on methodologies in the application of LCA to food products and agro-systems (Notarnicola et al., 2011). It is therefore important to study the works that have been already completed for standardise the application protocols and make appropriate comparison among results.

In order to assess papers that reflect the mainstream ideas about application of LCA in fruit systems, just peer-reviewed papers from international journals and conference proceedings were considered. We preferentially included studies that considered the part of the life cycle until the fruit was produced. Studies that considered the whole production of derivates (e.g. fruit juice production) were only included if they added to the analysis of the plantation stage.

The review covers all main aspects for conducting an LCA in fruit production system; in particular the following characteristics were considered: objectives, system boundaries, product considered, functional unit, data origin, life-cycle-based methodology adopted and environmental impact assessment method. The evaluation on the objectives was conducted considering eight objectives highlighted from the general literature on LCA applications in the food sector (not just fruit production). 1) Profile the environmental burden of a fruit product, in which a specific production is evaluated and results are related to the case study without meaning of generalisations. 2) Identify the environmental hot spots in production systems performance; considering the different field operations and stages of the system. 3) Describe management strategies to increase environmental performance; this focus is usually applied after the objective 2 in order to give practical suggestions after the evaluations. 4) Compare the environmental burden of different food products on a common functional unit, e.g. a specific unit of nutrient content. 5) Compare different agro-techniques, e.g. organic and conventional. 6) Compare different environmental assessment methods, thus compare the results of the application of methods, such as LCA, ecological footprint analysis or water footprint, on the same study case. 7) Profile the environmental burden of a production in a given area; in this studies the LCA evaluation is applied to a statistical database of farms collected in a specific area. 8) Evaluate the environmental properties of a supply chain, with usually focus on the different in environmental impacts on long and short distance between production and consumption sites. 9) Assess a preliminary study for statistical investigations; in this case the LCA results are used with the results of other indicators to elaborate complex indexes.

3. Critical review of LCA studies of fruit

A total of 19 works were identified; 11 articles in ISI journals and 8 papers in proceedings from the LCA congress series (Table 1).

General aspects of the study cases. With the exception of rare pioneer studies, it can be assumed that mainstream research on the LCA applied to fruit production systems began in the second part of the first decade of the century. A number of papers were published in 2010 during the occurrence of the 7th International Conference on LCA in the Agri-Food Sector. Despite the high quantity of fruit produced in Asia, most of the LCA applications published internationally focus on case studies which are located in Europe and South America. Just one work may be found in China (Liu et al., 2010); it is therefore realistic to assume that in the coming years, a lot of researches on this subject will focus on the Asian continent both for case studies and for environmental evaluation of fruit commercialisation.

Objectives. Most of the papers present more than one objective with the exception of the works on the supply chain (ob. 8), which are usually focused on just this aspect (e.g. Blancke and Burdick, 2008) even if they deeply investigate the field phase of the production process (e.g. Knudsen et al., 2011). The description of the environmental burden of the product (ob. 1) is the first result of each investigation, but it is often not the main object of the paper, which may be instead, for example, the comparison of different methodologies (e.g. Cerutti et al., 2010). The suggestion of ways to increase sustainability (ob. 3) is often associated to the evaluation of environmental hot spots (e.g. Cudjoe et al., 2010). The comparison of different assessment methods is not usually applied to fruit production; it can be only found in the comparison of LCA with Ecological Footprint Analysis (Cerutti et al., 2010) and LCA with PAS 2050 (McLaren et al., 2010).

System boundaries. The two most used system boundaries are the cradle-to-gate approach and cradle-to-market approach. In the first category, the environmental impacts are quantified for the production phase including all upstream impacts until the farm gate (8 papers). The cradle-to-market category includes studies in which the distribution and commercialisation phase is included in the assessment (9 papers). Two particular boundaries are the cradle-to-retailer (2 papers) in which also processing and transport to the distribution system is accounted, and the cradle-to-use (1 paper) in which also impacts from the consumer phase are accounted. The nursery, where orchard seedlings are produced, should be considered an upstream process delivering grafted plants to the orchards and the impact during this stage should be included in assessments of fruit production systems. Although many authors stress that it is important to consider the nursery in environmental impact assessments (Milà i Canals and Polo, 2003; Cerutti et al., 2010), the lack of data makes this difficult. Another important aspect that has to be considered, when the assessment is done on the entire life cycle of the orchard and not just on a productive year, is the yield in relation to the age of the plantation (Cerutti et al., 2010). Most of the temperate fruit cultures reach maturity in 2-4 years after installation of the orchard. Before that age, the yield may be significantly lower (or even zero) because the plants are still too young. This may significantly affect the average yield, and has to be considered. Furthermore, the yield variability between years may be very high, e.g. McLaren et al., (2010) reported that the difference between the lowest and highest yields for green kiwifruit over a period of six year, measured as a percentage of the lowest value, is 31%.

Product considered and functional unit. Fruits and fruit products may have different quality, nutrient and economic values, thus it may be difficult to find a significant functional unit. For fruit products, typical functional units are 1 kg of fruit packed and delivered to the customer or 1 tonne of fruit at the farm gate.

Table 1. List of all papers presenting applications of LCA in fruit production systems since January 2012 from ISI Journal and conferences, listed by date of publication. Country category considers the area of the study and not necessarily the location of the research group. For objectives description see reference numbers in the text.

REFERENCE	**COUNTRY**	**PRODUCT**	**MAIN OBJECTIVES**	**FUNCTIONAL UNIT**	**BOUNDARIES**	**DATASET**	**ASSESSMENT METHOD**
Blanke and Burdick, 2005	Germany, New Zealand	Apple	8	Mass based (kg)	Cradle-to-market	Literature and other databases	Characterisation factors from literature
Sanjuán et al., 2005	Spain	Orange	1, 2, 7	Mass based (kg)	Cradle-to-gate	Literature and other databases	CML, WMO, POPC and USES
Milà i Canals et al., 2006	New Zealand	Apple	1, 2, 3	Mass based (t)	Cradle-to-market	Commercial orchards + validation	EDIP1997
Mouron et al., 2006	Swiss	Apple	1, 2, 3, 7	Land based (ha); Currency based ($)	Cradle-to-gate	Commercial orchards	SALCA (2003)
Milà i Canals et al., 2007	UK, New Zealand	Apple	8	Mass based (kg)	Cradle-to-market	Literature and specific databases	Characterisation factors from literature
Sim et al., 2007	Brazil, Chile, Italy, UK	Apple	8	Mass based (t, just grade 1)	Cradle-to-retailer	Literature and specific databases	CML 2 Baseline 2000
Williams et al., 2008	UK, Spain	Strawberry	8	Mass based (t at distribution)	Cradle-to-market	Literature and specific databases	Characterisation factors from literature
Beccali et al., 2009	Italy	Citrus based products	1	Mass based (kg of juices and oil)	Cradle-to-market	Primary data from field and secondary from literature	IPCC 2001 GWP100; CML 2 Baseline 2000
Coltro et al., 2009	Brazil	Orange	1, 7	Mass based (t)	Cradle-to-gate	Commercial orchards	Characterisation factors from literature
Beccali et al., 2010	Italy	Citrus based products	3	Mass based (kg of juices and oil)	Cradle-to-market	Primary data from field and secondary from literature	IPCC 2001 GWP100; CML 2 Baseline 2000
Cerutti et al., 2010	Italy	Peach	1, 2, 6	Mass based (kg)	Cradle-to-gate	Commercial orchards + validation	Eco-Indicator 99
Cudjoe et al., 2010	Ghana	Pineapple	2, 3	Mass based (kg)	Cradle-to-gate	Commercial orchards	Characterisation factors from literature
Ingwersen, 2010	Costa Rica	Pineapple	1, 9	Mass based (serving portion)	Cradle-to-retailer	Commercial orchards	ecoinvent 2.0
Liu et al., 2010	China	Pear	2, 8	Mass based (t)	Cradle-to-market	Commercial orchards	IPCC 2007
Clasadonte et al., 2010 a	Italy	Peach	4	Mass based (kg)	Cradle-to-gate	Commercial orchards	Impact 2002+
Clasadonte et al., 2010 b	Italy	Orange	1, 3	Mass based (kg)	Cradle-to-gate	Commercial orchard	Impact 2002+
McLaren et al., 2010	New Zealand	Apple, Kiwifruit	1, 3, 6	Mass based (kg)	Cradle-to-use	Commercial orchards	PAS 2050
Cerutti et al., 2011	Italy	Apple	8	Mass based (kg)	Cradle-to-market	Retailer and associated orchards	EDIP 1997
Knudsen et al., 2011	Brazil	Orange	5, 8	Mass based (l of juice); Mass based (t of fruit)	Cradle-to-market Cradle-to-gate	Commercial orchards and statistics	EDIP 1997 + IPCC 2007 (GHG); IMPACT2002+ (energy)

There is just one work (Mouron et al., 2006) in which a land based and a currency based functional unit are related. The land based functional unit, such as 1 ha of orchard, is not frequently used in LCA, partly because land use is not directly a service and does not provide a productive function, but it can give interesting results. In general, converting resource consumption or environmental impacts to units of land use allows evaluation of the impacts of cultivating a certain area. This parameter is also called the impact intensity of a farm (Mouron et al., 2006). The land-based functional unit in fruit production is complementary to the mass-based functional unit because they give different results and both should be used. Indeed, when considering just impacts per unit area, low input-output systems will have better ranking for decreased impacts at regional level, but may create a need for additional land use elsewhere, giving rise to additional impacts (van der Werf et al., 2007). Furthermore, as most of fruit are rapidly perishable products, quantification of product loss in the supply chain would be needed in order to evaluate the environmental impact of the product actually consumed (Schau and Fet, 2008).

Data origin. Most studies (11 papers) are based on data collected from commercial orchards, either directly in field surveys or with questionnaires or interviews with farmers. Sometime these approaches are mixed and the data collection method used for the different data in the study is not always clearly described. Four studies investigate commercial orchards and then compare the field dataset obtained with reference values. This approach allows conclusions about specific orchards to be drawn, while the validation allows identification of unusual agricultural practices only of interest for the specific farm (e.g. Milà i Canals et al., 2006). The other method used to obtain statistically robust datasets is to consider a larger number of commercial orchards and look at average values for these farms. Seven studies used literature and available databases in order to obtain data instead of surveying commercial orchards. Appling this methodology it possible to achieve more generic results, but it could be impossible to consider site specific differences among orchards.

Environmental impact assessment method. Using different environmental impact assessment methods may lead to different conclusions. Across reviewed papers, the typical impact categories are the categories which quantifies environmental impacts on ecosystems more than the ones on resource consumption or human toxicity; with particular attention to the potentials of global warming, eutrophication, and acidification. The first one is mainly related to the combustion of fuels, thus it is considered a key indicator in studies involving comparison of systems with different transport distances (e.g. Blanke and Burdick, 2005). Eutrophication and acidification are generally more related to the use of fertilisers and pesticides, thus they depend on the agro-technique used and climatic conditions.

When defining the impact categories for fruit production, it is very important to consider the typical environmental problems that may arise in orchards (Milà i Canals and Polo, 2003). Fruit is usually produced in sunny regions because sun increases yield and improves fruit quality. However, these regions are also prone to water scarcity and resulting losses of nutrients and pesticides to the surrounding environment. These effects can influence all impact categories, but particularly nutrient enrichment potential and acidification potential (Coltro and Mourad, 2009), as well as human toxicity.

4. Conclusions

Despite a general standardisation of phases in orchard management, the high variability in agro-techniques and fruit products leads to different way of applying LCA in such systems. Nevertheless a tentative to standardise research protocols when applying environmental assessment method in fruit production is needed. In particular about the set of indicators and the orchard model that should be considered. Otherwise result may be impossible to compare and results risk remaining isolated to the case study. Being able to compare the results from different studies would be important also in order to identify sustainability threshold, as suggested by several authors (e.g. Van der Werf and Petit, 2002).

Suggestions for standardisation of assessment methods application in fruit production may consider the inclusion of the whole lifetime of the orchard in the system boundaries. As a consequence also impacts from orchard installation, destruction and the nursery phase should be assessed. As orchards are not a single year production system (as can be open field crops), the application of an environmental indicator just to the full production year will probably underestimating the real ecological impact, in a variable percentage (in our studies about 30% depending on fruit considered and assessment method) (Fig.1).

As in other reviews about similar topics (Petti et al., 2010) one of the most frequent problems was the difficulty in finding specific data and characterisation factors for pesticides and fertilisers. The fate and the effects of chemicals in the environment are very different depending on the pedo-climatic condition of the orchard. Therefore it is necessary implement the analysis with a predictive mathematical method which is able to model pesticide dispersion, such as PestLCI developed by Birkveda and Hauschild (2006). Nevertheless, the use of pesticide dispersion models requires specific competences and several pedo-climatic data, such as soil properties, average rain quantities and wind intensities of the investigation site. The alternative method is to considered the environmental impact of the entire amount of pesticides as emitted to the soil.

These approaches lead to different results in the assessment with potentially dramatic effects in comparative studies if one uses different methods of impact assessment. Similar consideration can be made when evaluating impacts from fertilisation. In this aspect two approaches are usually applied: the use of a dispersion model or a nutrient balance in which impacts of distribution are related to the effects in the environment of the surplus nutrient (Milà I Canals et al., 2006). This second approach requires specific agronomic investigations about the nutrient needing of the plants and the nutrient content in the soil of the orchard.

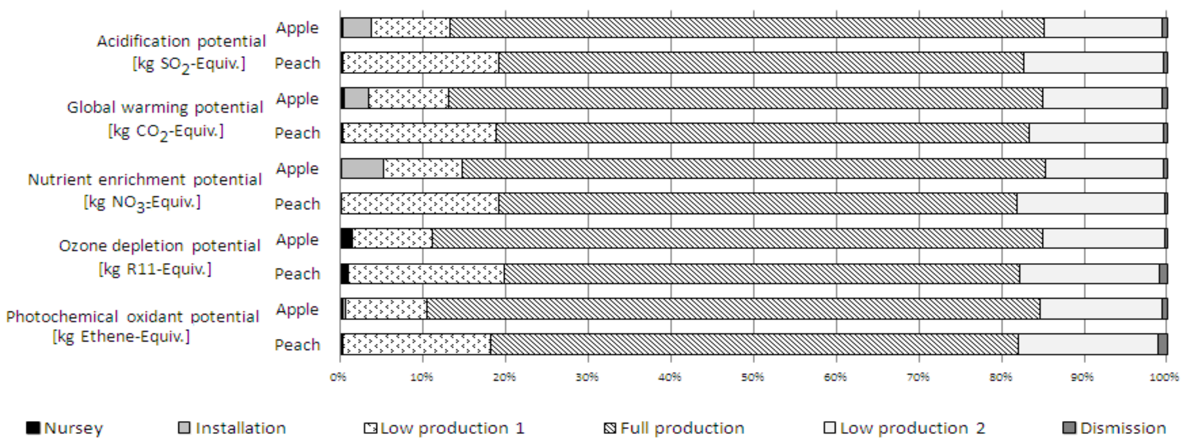


Figure 1. Hotspot analysis of two previous case studies. Modified from Cerutti et al., 2011.

Furthermore, the use of different functional units may result in deviating results. The reviewed literature shows that simple mass based functional units is not always able to capture the complexity of orchard systems. Thus, using combined functional units or other functional units may be necessary. For example, impacts per land units may be used together with mass based functional unit in order to complete results and avoid resource use efficiency overvaluation and delocalisation of environmental impacts.

Started with this critical review, the main objective of the Working Group on Fruit LCA will be the elaboration of a framework methodology and specific guidelines to improve LCA analysis on orchard cultivation and fruit products.

5. References

Beccali, M., Cellura, M., Iudicello, M., Mistretta, M., 2009. Resource consumption and environmental impacts of the agrofood sector: life cycle assessment of italian citrus-based products. Environmental management. Springer New York. 43, 707-24.

Beccali, M., Cellura, M., Iudicello, M., Mistretta, M., 2010. Life cycle assessment of Italian citrus-based products. Sensitivity analysis and improvement scenarios. Journal of environmental management. 91, 1415-28.

Berners-Lee, M., Hoolohan, C., Cammack, H., Hewitt, C.N., 2012. The relative greenhouse gas impacts of realistic dietary choices. Energy Policy. 43, 184-190.

Blanke, M., Burdick, B., 2005. Food (miles) for Thought - Energy Balance for Locally-grown versus Imported Apple Fruit (3 pp). Environmental Science and Pollution Research - International. Springer Berlin / Heidelberg. 12, 125-127.

Birkveda M., Hauschild M.Z., 2006. PestLCI—A model for estimating field emissions of pesticides in agricultural LCA. Ecological Modelling 198:433–451

Carlsson-Kanyama, A., Ekström, M.P., Shanahan, H., 2003. Food and life cycle energy inputs: consequences of diet and ways to increase efficiency. Ecological Economics. 44, 293-307.

Cerutti, A.K., Bagliani, M., Beccaro, G.L., Peano, C., Bounous, G., 2010. Comparison of LCA and EFA for the environmental account of fruit production systems: a case study in Northern Italy. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.), Proceedings of LCA food 2010. Bari, pp. 99-104.

Cerutti, A.K., Galizia, D., Bruun, S., Mellano, G.M., Beccaro, G.L., Bounous, G., 2011. Assessing environmental sustainability of different apple supply chains in northern italy, in: Finkbeiner, M. (Ed.), Towards Life Cycle Sustainability Management. Springer Netherlands, Dordrecht, pp. 341-348.

Clasadonte, M.T., Matarazzo, A., Ingrao, C., 2010. Life Cycle assessment of Sicilian peach sector. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.), Proceedings of LCA food 2010. Bari, pp. 295-300.

Clasadonte, M.T., Lo Giudice, A., Ingrao, C., 2010. Life Cycle Assessment of the Sicilian citrus fruit field. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.), Proceedings of LCA food 2010. Bari, pp. 301-306.

Coltro, L., Mourad, A.L., Kletecke, R.M., Mendonça, T.A., Germer, S.P.M., 2009. Assessing the environmental profile of orange production in Brazil. The International Journal of Life Cycle Assessment. Springer Berlin / Heidelberg. 14, 656-664.

Cudjoe Adebah, E., Langeveld, C., Kermah, M., 2010. Environmental impact of organic pineapple production in Ghana: a comparison of two farms using Life Cycle Assessment (LCA) approach. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.), Proceedings of LCA food 2010. Bari, pp. 225-230.

Ingwersen, W., 2010. Product category range of environmental perfformance for EPDs: example of Costa Rican pineapple. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.), Proceedings of LCA food 2010. Bari, pp. 337-341.

Knudsen, M.T., Fonseca de Almeida, G., Langer, V., Santiago de Abreu, L., Halberg, N., 2011. Environmental assessment of organic juice imported to Denmark: a case study on oranges (Citrus sinensis) from Brazil. Organic Agriculture. Springer Netherlands. 1, 167-185.

Liu, Y., Langer, V., Høgh-Jensen, H., Egelyng, H., 2010. Life Cycle Assessment of fossil energy use and greenhouse gas emissions in Chinese pear production. Journal of Cleaner Production. 18, 1423-1430.

McLaren, S.J., Hume, A., Mitraratne, N., 2010. Carbon management for the primary agricultural sector in New Zealand: case studies for the pipfruit and kiwifruit industries. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.), Proceedings of LCA food 2010, pp. 293-298.

Milà i Canals, L., Burnip, G.M., Cowell, S.J., 2006. Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): Case study in New Zealand. Agriculture, Ecosystems & Environment. 114, 226-238.

Milà i Canals, L., Cowell, S.J., Sim, S., Basson, L., 2007. Comparing domestic versus imported apples: A focus on energy use. Environmental Science and Pollution Research - International. Springer Berlin / Heidelberg. 14, 338-344.

Mouron, P., Nemecek, T., Scholz, R.W., Weber, O., 2006. Management influence on environmental impacts in an apple production system on Swiss fruit farms: Combining life cycle assessment with statistical risk assessment. Agriculture, Ecosystems & Environment. 114, 311-322.

Notarnicola, B., Tassielli G., Settanni E., 2010. Including more technology in the production of a quality wine: the importance of functional unit. In Notarnicola B., Settanni E., Tassielli G., Giungato P. (Eds.), Proceedings of LCA food 2010. Bari, vol. 2, pp. 235-240.

Petti, L., Ardente, F., Bosco, S., De Camillis, C., Masotti, P., Pattara, C., Raggi, A., Tassielli, G., 2010. State of the art of Life Cycle Assessment in the wine industry. In Notarnicola B., Settanni E., Tassielli G., Giungato P. (Eds.), Proceedings of LCA food 2010. Bari, vol. 1, pp. 493-498.

Sanjuan, N., Ubeda, L., Clemente, G., Mulet, A., Girona, F., 2005. LCA of integrated orange production in the Comunidad Valenciana (Spain). International Journal of Agricultural Resources, Governance and Ecology 4 (2), 163-177.

Schau E.M., Fet A.M., 2008. LCA Studies of Food Products as Background for Environmental Product Declarations. International Journal of LCA 13(3):255–264.

Sim, S., Barry, M., Clift, R., Cowell, S.J., 2007. The relative importance of transport in determining an appropriate sustainability strategy for food sourcing. The International Journal of Life Cycle Assessment. Springer Berlin / Heidelberg. 12, 422-431.

van der Werf H.M.G., Tzilivakis J., Lewis K., Basset-Mens C., 2007. Environmental impacts of farm scenarios according to five assessment methods. Agriculture, Ecosystems and Environment, 118:327–338

Wallén, A., Brandt, N., Wennersten, R., 2004. Does the Swedish consumer's choice of food influence greenhouse gas emissions? Environmental Science & Policy. 7, 525-535.

Williams A, Pell E, Webb J, Moorhouse E, Audsley E., 2008. Strawberry and tomato production for the UK compared between the UK and Spain. Proceedings of the 6th International Conference on LCA in the Agri-Food Sector, pp. 254-262

An LCIA-based typology for more representative results and refined data collection of a horticultural cropping system in the Tropics. The case of tomato production in Benin, West Africa

Aurélie Perrin[1,*], Claudine Basset-Mens[1], Wilfried Yehouessi[2], Joel Huat[1], Benoit Gabrielle[3]

[1] CIRAD- UPR Hortsys, TA B-103/PS4 -Boulevard de la Lironde, F-34398 Montpellier Cedex 5, France
[2] Faculté des Sciences Agronomiques (FSA) de l'Université d'Abomey – Calavi (UAC), Benin
[3] AgroParisTech, INRA, UMR 1091 Environnement et Grandes Cultures, F-78850 Grignon, France
[*] Corresponding author. E-mail: *aurelie.perrin@cirad.fr*

ABSTRACT

Application of Life-Cycle Assessment (LCA) to horticultural products raises renewed and specific issues, especially in tropical areas. Given the scarcity of data and expert knowledge in these areas, the first challenge lies in obtaining Life-Cycle Inventory (LCI) representative of the diversity of cropping systems involved in the production function. In this study, we identified the cropping systems' components that are likely to influence the environmental impacts of the production, and their variability. To explore the diversity of cropping systems from an LCA perspective, 12 cropping systems producing dry season tomato production in the coastal area of Benin were selected based on local-expert knowledge and assessed with LCA. A two-step statistical analysis (CPA-AHC) allowed to produce a typology of cropping systems with regard to their environmental impacts and highlights the key components responsible for the hot-spots. This result has direct implications for a better design of data collection in a highly complex system.

1. Introduction

The growing awareness regarding the environmental issues associated with global food supply chains has now reached the fruits and vegetables sector, stressing the need for evaluating products with different origins and the associated technologies (Martínez-Blanco et al., 2011). Although the relevance of Life-Cycle Assessment (LCA) to assess agricultural systems has been validated for a large number of products (Brentrup et al., 2004), its application to horticultural production systems especially in tropical contexts is more recent and comes with renewed and specific issues. Considering the scarcity in data and expert knowledge in tropical areas, the first challenge lies in obtaining Life-Cycle Inventory (LCI) data representative of a given function at a given scale according to the goal and scope defined in a particular LCA. Little attention has been paid to LCI data for horticultural crops in the tropics. Characterising the whole diversity and complexity of these cropping systems is indeed data- and time-consuming. Methods are required to identify the key characteristics of cropping system specificities that determine their environmental impacts. The purpose of the present study was to propose and test an approach that allows the identification of such key drivers relevant to the LCA results for horticultural crops in the Tropics. We set out 1) to classify cropping systems with a typology based on their potential impacts, and 2) to analyse the contribution of cropping system components to the environmental impacts of the systems. Here, we focused on dry land tomato cropping systems in Benin, as a case-study. The proposed approach is generic and should be useful for designing a more efficient data collection protocol, for researchers and industrials who want to properly assess the environmental impact of tropical fruits and vegetables.

2. Material and methods

2.1. Field selection and characteristics

This case-study was based on the 2011 dry season tomato production in the coastal area of Benin. Benin is located in West Africa, and its coastline presents a subequatorial climate with two dry and two rainy seasons every year. The main season for growing tomatoes in West Africa is the longest rainy season as it does not require extra work and infrastructure for irrigation. Nevertheless, out-of-season tomato is grown commonly on the coast, where water is available for irrigation, to provide all-year long fresh tomatoes for traditional dishes. Among the 50 tomato growers identified in the region, 12 were selected based on two criteria, presenting all three modalities: geographical location (in the vicinity of Cotonou, Pahou, and Grand-Popo, which are the largest cities in the area) and irrigation systems (manual, hose and sprinkler). On the basis of local expert knowledge, the field selection aimed to cover the different combinations of these modalities at regional scale. Each agricultural operation occurring in the field was recorded and quantified during the full crop-cycle, from nursery to harvest. The main characteristics of the selected fields are displayed in Table 1.

Table 1. Characteristics of the 12 selected field representative for against the season tomato in Benin

Field codes	Geo Loc	Irr syst	Field area (m²)	Duration (days)	Yield ($kg.ha^{-1}$)	Nitrogen rate ($kg.ha^{-1}$)	Number of pesticide applications	Water supplied ($m^3.ha^{-1}$)	Soil type
P06	Cot	Man	306	118	4902	2684	29	7200	Sand
P07	Cot	Man	126	104	12452	2807	4	6360	Sand
P10	Cot	Sprink.	196	104	0	905	10	2665	Sand
P12	Cot	Man	546	111	7875	755	17	10400	Silt
P28	Pa	Man	922	110	8	1328	7	586	Silt
P17	Pa	Hose	169	102	127	665	5	4515	Sand
P19	Pa	Hose	1915	56	0	1389	12	919	Silt
P38	Pa	Hose	895	88	10	72	11	3759	Sand
P39	Pa	Hose	760	103	5662	507	17	10581	Sand
P33	GPP	Hose	2200	104	4498	2258	10	771	Clay
P37	GPP	Hose	576	113	21163	1595	5	5318	Sand
P40	GPP	Hose	963	113	1703	60	1	7048	Sand

2.2. Life-cycle assessment methodology

An ISO-compliant LCA was performed to compare the environmental impact of the 12 fields selected. The functional unit assigned for all fields was one hectare of tomato production in agreement with the farmers' strategy to give value to an area through the production of a commercial product. The fields were assessed from-cradle-to-field-gate. Input production and infrastructures were included, while transport and end-of-life were not taken into account. Data on the production of agricultural inputs was taken from the Ecoinvent database (v2.2). Field emissions were estimated using the best available methods for such a specific context. Ammonia (NH_3) emissions following the application of mineral fertiliser nitrogen were estimated using emission factors from the ECETOC report (ECETOC 1994). Emission factors of group I (high NH_3 emission potential due to high temperature and pH) were chosen to be representative as much as possible for the tropical context with high temperatures. For volatilisation from poultry manure, a 20% emission factor was selected from the Ecoinvent guidelines (Nemecek and Kägi 2007). The nitrogen content of poultry manure was set to 3% according to the organic fertilisation guide of the Reunion Island (Chabalier et al., 2006) taken as representative for a tropical context. The emissions of nitrous oxide (N_2O) and nitric oxide (NO_x) were estimated following the Tier 1 methodology of the IPCC guidelines (IPCC 2006) including both direct and indirect emissions. Due to the lack of a suitable method for horticultural crops in the Tropics, we estimated nitrate leaching as 30% of total N applied (IPCC, 2006). Emissions of carbon dioxide (CO_2), phosphorus, and pesticides were estimated following methods from Nemecek and Kägi (2007). The SimaPro (v7.2) software was used to analyse the environmental impact through the 13 impact categories defined by ReCiPe (Goedkoop et al., 2009). The names and units of the impact categories are given in Table 2.

2.3. Two-step statistical analysis

To describe the diversity of environmental impacts for the studied cropping systems, we used a two-step statistical treatment. The first step was to transform the impacts categories into non-correlated variables with a Principal Component Analysis (PCA). The number of variables was reduced into less dimensions representative for most of the variability observed in the population. Once individual fields were spread on this multidimensional space, the PCA method allowed identifying one most representative field located closest to the intersection of the dimensions and, fields responsible for the variability located at the edge of the cloud. Fields were then grouped into clusters corresponding to cropping system types using an Agglomerative Hierarchical Clustering (AHC) algorithm, in which the principal components of the PCA were used as input variables. As PCA allows the calculation of one representative field for the whole population, AHC allows identifying one representative field (called paragon) for each cluster (or type). These paragons are located near the centroid of each cluster. The population can therefore be analysed using those typical individuals, avoiding any aggregation or averaging often leading to discrepancies with the reality and the system logic of the cropping system. Such a two-step statistical treatment is also called a goal-oriented typology and has been used in studies from the agronomic discipline to analyse the diversity of cropping systems and farms (Poussin et al., 2008). Our analysis was strengthened through the testing of the distribution of additional qualitative or quantitative variables (geographical, irrigation system, soil types or cropping systems components) using Kuiper's test against dimensions and clusters. These variables are summarized in Table 1.

3. Results

3.1. Main contributors for the environmental impact of out-of-season tomato in Benin

Most of the variability (90%) observed between individual fields was explained by the 3 first dimensions of the PCA. In this space, the individual fields P38 is located near the intersection of the dimension. By definition from the PCA method, this individual field can be considered representative fields for the environmental impacts of the population. For the following analysis, P38 which had the smallest distance to the origin of the PCA map (d=1.98) is used to analyse the environmental impact of one hectare of out-of-season tomato grown in the region. The contribution analysis of P38 (Table 2) identified energy used for irrigation as the main contributor for 7 among 14 impact categories. Then Nitrogen reactive emissions were the main contributor for 2 impact categories (TA and ME) and also contributed to 2 other categories (GWP and POF). Crop protection (insecticides and fungicides) showed high contribution for TET, FET and MD.

Table 2. LCIA results and contribution analysis for P38 and main correlations between environmental impacts and the first 3 dimensions of the PCA

Impact categories	P38 LCIA		P38 contribution	Correlation
Global Warming Potential- GWP	1.17E+04	kg CO_2 eq	Irr. machinery (85%), N emissions (15%)	Dim 1 (0.99)
Ozone Depletion- OD	1.22E-03	kg CFC-11 eq	Irrigation (Irr.) machinery (100%)	Dim 1 (0.96)
Human Toxicity- HT	6.07E+06	kg 1,4-DB eq	Irrigation machinery (100%)	Dim 1 (0.94)
Photo Oxidant Formation - POF	2.11E+01	kg NMVOC	Irr. machinery (95%), N emissions (5%)	Dim 1 (0.97)
Terrestrial Acidity- TA	2.41E+02	kg SO_2 eq	N emissions (92%), Irr. machinery (8%)	Dim 2 (0.81)
Freshwater Eutrophication- FE	9.54E-01	kg P eq	P emissions (59%), Irr. machinery (36%), Insecticides (4%)	Dim 3 (-0.83)
Marine Eutrophication- ME	3.99E+01	kg N eq	N emissions (99%), Irr. machinery (1%)	Dim 2 (0.85)
Terrestrial Eco Toxicity- TET	1.97E+02	kg 1,4-DB eq	Insecticides (98%), Irr. machinery (2%)	Dim 2 (0.90)
Freshwater Eco Toxicity- FET	1.22E+02	kg 1,4-DB eq	P emissions (51%), Insecticides (39%), Irr. machinery (10%)	Dim 2 (0.92)
Marine Eco Toxicity - MET	8.60E+02	kg 1,4-DB eq	Irr. machinery (99%), Insecticides (1%)	Dim 1 (0.95)
Agricultural Land Occupation- ALO	1.57E+03	m^2a	Field Area (99%), Irr. machinery (1%)	-
Water Depletion- WD	3.81E+03	m^3	Irrigation water (100%)	-
Metal Depletion- MD	1.09E+02	kg Fe eq	Irr. machinery (75%), Fungicides (24%), Insecticides (1%)	-
Fossil Depletion- FD	3.49E+03	kg oil eq	Irrigation machinery (100%)	Dim 1 (0.96)

3.2. Correlations between major impact categories and cropping system characteristics

Table 2 presents the correlations between the 3 first dimensions and the environmental categories, and between LCIA results and cropping system components for the field P38. Eleven out of 14 impacts categories were correlated to one of the three dimensions. These categories are of major concern for the variability of the environmental impact for the studied system.

Since GWP, POF, FD, OD, MET and HT (see Table 2 for acronyms) were correlated to dimension 1, this dimension was a good synthetic descriptor for the variability of the dataset (50%). All these impact categories were primarily affected by the energy used for irrigation. GWP and POF were secondarily affected by N emissions, while MET was affected by pesticides emissions. Dimension 2 represented 26% of the overall variability, and was mainly correlated to FET, TET, ME, and TA. ME and TA were mainly impacted by nitrogen emissions, while FET was impacted by both phosphorus emissions and insecticides. Finally TET was mainly impacted by insecticides.

The Kuiper's test showed that the distribution of certain additional variables was consistent with the dimensions of the PCA. Soil type is correlated to dimension one, with sandy soils showing higher values (v-test =2.28) and silty soils showing lower values than the population average (v-test =-1.97). The energy use for irrigation (through the number of pumping hours) was also correlated to dimension 1, with manual systems showing lower values than the population average (v-test =-2.05). Unexpectedly, the volume of water supplied did not correlate with this dimension suggesting that a greater energy use for pumping water did not necessarily result in a greater volume applied due to important discrepancies in the technologies' efficiency. The geographical location (urban or peri-urban), was correlated to dimension 2 with the fields of Cotonou showing greater values (v-test=2.13) than the overall population. The number of pesticides applications at the field stage and the number of fertiliser applications were also correlated to dimension 2. Finally the rate of mineral nitrogen was negatively correlated to dimension 3, highlighting a dichotomy between the impact of phosphorus emission and the use of mineral fertiliser (the main source of phosphorus being poultry manure).

3.3. Variability of impacts and cropping system characteristics as explained by the clustering

The optimal number of clusters minimizing the distance between fields in a group and maximizing the distance between groups was 4. The cluster tree and characteristics of each nod are showed in Figure 1.

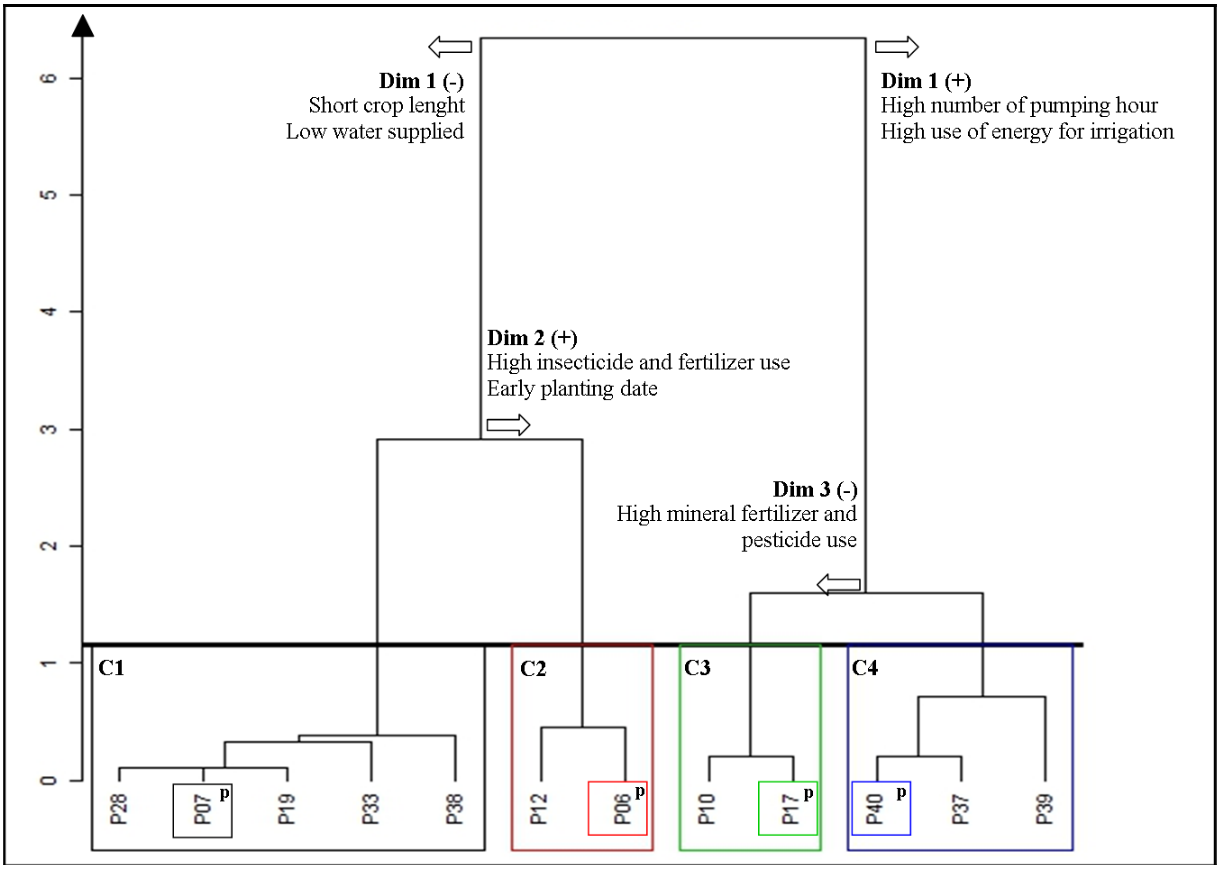


Figure 1. Hierarchical tree presenting the 4 clusters with positive (+) or negative (-) variations for the dimensions of PCA and correlated cropping system characteristics. For each cluster paragons are identified by ‘p’

Cluster one (C1) was negatively correlated to dimension 1. Accordingly, C1 showed significantly less impact for OD, MET, FD, POF and GWP (v-test>2), which were the impact categories correlated to dimension 1. In addition, cluster 1 showed significantly lower values for three additional variables: crop duration, number of irrigation and water supply (v-test>|2|). The LCIA results of the paragon P7 for cluster 1 were less than that of P38 (representative for the whole population) for almost all impact categories except for FE and MD. N,P,K-fertiliser manufacturing was pinpointed as the main contributor for impact categories correlated to dimension 1. Conversely, Cluster 4 (C4) was positively correlated to dimension 1 and showed significantly greater impacts for HT, MET, OD, FD, GWP and POF (v-test>2) than the overall population. In addition, C4 was positively correlated to the number of water pumping hours during both the nursery and the field stages. For categories correlated to dimension 1, the LCIA results for the paragon P40 were two fold greater than that of the reference P38 and the main contributor was clearly energy use for irrigation.
Cluster 2 (C2) was positively correlated with dimension 2 and had significantly greater TET and FET impacts than the overall population. C2 showed significantly greater values for the number of pesticide applications, as a result of a larger number of insecticide applications. The application rates of insecticides were also greater for cluster 2. Similarly, cluster 2 showed significantly greater values for the number of fertiliser applications, correlated to the number of mineral fertiliser applications. Moreover, C2 was negatively correlated to sowing date and planting date, suggesting that, the earlier was the crop cycle, the higher were input levels for those fields. LCIA results of the paragon P6 for cluster 2 further supported these results, highlighting insecticides application as main contributor for both TET and FET. Finally, cluster 3 (C3) was negatively correlated to dimension 3 and showed significantly greater impacts for FE and MD. C3 showed significantly higher values for mineral nitrogen application rates. The LCIA results of P17 showed high contribution of

fertiliser production for FE and MD. It also evidenced a high contribution of energy use for irrigation, similarly to cluster 4. P17 and P40 (the paragons of clusters 3 and 4, respectively) showed similar impacts for most categories, except for TA, FE, ME, TET and FET for which the impacts of P40 were significantly lower. The greater impacts of P17 was due to pesticide applications (TET and FET), N,P,K-fertiliser manufacturing and nitrate emissions (TA, FE and ME).

4. Discussion

4. 1. Characteristics of the types of cropping systems with regard to field selection

The diversity of horticultural cropping systems in the Tropics in general and in our context of study in particular led us to infer that the associated environmental impacts would vary greatly between individual fields. To assess the environmental impacts of out-of-season tomato cropping systems in Benin, we selected a large sample of 12 fields over 50 and assumed it was large enough to represent the whole population. This selection was guided by local expert knowledge and included geographical location (urban/peri-urban) and irrigation system (manual/mechanised). Our principal component analysis showed that individual fields were all different. However, by applying the AHC approach, we could group the population into four clusters defined each by one representative fields (the paragon) showing a typical profile of the whole cluster in terms of environmental impacts.

Looking in more detail, the first driver was energy use for irrigation, as reflected by dimension 1 of the PCA (making up 50% of the variability). Energy use for irrigation was also responsible for the first dimension, grouping C1 and C2 apart from C3 and C4. Soil type contributed to this effect, as the impact of energy use for irrigation was higher on sandy soils. These findings support the relevance of our initial typology based on irrigation systems. It also emphasized the need to include soil type as well as an important criterion to identify representative cropping systems. Geographical location was correlated to the variability explained by dimension 2. Farmers around Cotonou (urban area) have more intensive practices in terms of fertilisation and crop protection than the average population. These findings suggest that practices are more intensive in urban areas, due to scarcity of agricultural land. When assessing the environmental impacts of out-of-season tomato cropping systems in Benin, both urban and peri-urban systems need to be addressed as their environmental impacts are significantly different.

In addition to the expected sources of variability identified a priori, the AHC evidenced other cropping systems specificities. Based on the analysis of clusters presented in table 3, we could dissociate extensive (C1 and C3) from intensive (C2 and C4) cropping system types and identify the source of intensification: irrigation system (C4) or pesticides and fertilisers (C3). We also demonstrated a possible inconsistency between yields and practices intensity, highlighting the actual risk due to pest pressure and the question of rate of return. This observation is quite specific to horticultural crops grown in tropical peri-urban areas and clearly questions the choice of the most relevant functional unit for such systems. Finally, we also highlighted inconsistencies resulting in methodological weaknesses discussed in the following section.

Table 3. Environmental and agronomical analysis of clusters and subsequent outcomes for a better understanding of cropping systems

	Environmental impact variability	Agronomical performances *(*not statistically significant)*	Main outcomes
C1	Lower than P38	- Short crop length - *Yield losses due to biotic & abiotic causes**	**2 strategies** - Low investment strategy, rate of return? - Early crop failure
C3	↑ impact of irrigation system and NPK production ↑ impact of nitrogen emissions	- High mineral fertiliser rates - *Yield losses due to pest pressure**	**1 conclusion** Intensive practices ≠ high yields ➔ Investments = risk
C2	↑ impact of insecticides and nitrogen emissions	- High insecticide &fertiliser application number - Early planting dates	**2 conclusions** - Intensive practices ≠ technical irrigation infrastructures - Early planting = large use of insecticides
C4	↑↑ impact of irrigation	- High use of pump - Large water supplied	**2 questions** - Valorisation of investments in irrigation system? - Effect of water supply on impact of nitrogen emissions?

4.2. Implications of LCA impacts variability in the refining of methods and data collection

Crop protection showed relatively low contributions to the impact variability compared to other cropping system components. This result is not due to a low impact of pesticides, since normalised results (not shown) pointed terrestrial ecotoxicity (TET) as second important impact. The main reason why crop protection was not discriminating is that all pesticides emissions were emitted into the soil as recommended by (Nemecek and Kägi 2007).This method does not consider the behaviour of pesticides in the environment in relation to application methods, and subsequent transports in the main environmental compartments. As a consequence, the main drivers were pesticide application rates and toxicity potentials. Considering these drivers, the use of cypermethrin and organo phosphorus compounds (terbufos and dimethoate) were clearly discriminating the systems' impact, the first one due to its high application rate, the two other ones for their high toxicity. The assessment of crop protection practices requires improved methods for the estimation of pesticide emissions to the different environmental compartments as it should be a hot-spot for tropical crops. Fertilisation practices contributed to differentiating intensive cropping systems from extensive ones, mainly based on fertiliser types and rates. However, intensive fertilisation practices did not correlate with greater yields. This observation led us to question the methods used to estimate field emissions. Crop uptake is the main sink for applied nutrients. If it varies, field emissions should vary accordingly, especially for nitrate leaching which is related to the amount of nitrogen available in soils during drainage events. We could not use a method based on nitrogen balance for nitrate leaching since it requires to estimate drainage events (Brentrup et al., 2000), which depended in our cropping systems on the timing of irrigation inputs. No methods in LCI guidelines allowed including effect of irrigation practices on nitrate emissions. In addition, for cropping system with no drainage events (low water supplied per irrigation event), there should be an accumulation of nitrogen in soil. This might involve higher emissions of nitrous and nitric oxides than the average factor proposed by IPCC (IPCC 2006) and would deserve an improved modelling as well. For tropical horticultural systems where yields can be significantly reduced due to pest pressure, the behaviour of nitrogen in the environment needs to be modelled considering irrigation practices, the timing and method of fertiliser application and the actual yield.

5. Conclusion

The proposed typology allows the identification of representative individual fields to assess the environmental impact of tomato cropping systems in Benin. Our approach identified key data to be collected and estimated accurately in the LCI for horticultural products in the Tropics. In a near future we are planning to include these data in our LCI to assess their effects on LCA results in terms of sensitivity and uncertainty. Identifying cropping system types and specific LCI data makes it possible to reveal site-specific differences in the LCA results which did not appear when using the recommended LCI data set. Although LCA claims to be a method assessing global environmental impacts, we believe that results should account for key local variables.

6. References

Brentrup F, Küsters J, Lammel J, Barraclough P, Kuhlmann H (2004) Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. Eur. J. Agon. (3):265-279.

Brentrup F, Küsters J, Lammel J, Kuhlmann H (2000) Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int J Life Cycle Ass (6):349-357.

Chabalier PF, Van de Kerchove V, Saint Macary H (2006) Guide de la fertilisation organique à la Réunion. CIRAD.

ECETOC (1994) Ammonia emissions to air in Western Europe. Technical Report, vol Vol. 62. Brussels.

Goedkoop M, Heijungs R, Huijbregts M, De Schryver A, Struijs J, van Zelm R (2009) ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition, vol Report I: Characterisation.

IPCC (2006) Chapter 11: N2O emissions from managed soils, and CO2 emissions from lime and urea application. In: Eggleston S, Buendia L, Miwa K, Ngara T, Tanabe K (eds) IPCC guidelines for national greenhouse gas inventories. Institute for Global Environmental Strategies, Hayama, Japan.

Martínez-Blanco J, Muñoz P, Antón A, Rieradevall J (2011) Assessment of tomato Mediterranean production in open-field and standard multi-tunnel greenhouse, with compost or mineral fertilizers, from an agricultural and environmental standpoint. J Clean Prod (9-10):985-997.

Nemecek T, Kägi T (2007) Life Cycle Inventories of Agricultural Production Systems. Final report ecoinvent, vol v2.0 No.15. Swiss Centre for Life Cycle Inventories, Zurich, Switzerland.

Poussin JC, Imache A, Beji R, Le Grusse P, Benmihoub A (2008) Exploring regional irrigation water demand using typologies of farms and production units: An example from Tunisia. Agricultural Water Management (8):973-983.

French agricultural practices traceability system employed for environmental assessment

Pauline Maupu[1,*], Amandine Berthoud[1], Ophélie Negri[1], Edith Martin[2], Seheno Ratsimbazafy[2], Noëllie Oudet[2], Antoine Poupart[1]

[1] InVivo AgroSolutions, Direction Agriculture Durable et Développement, 83 av. de la Grande Armée, 75782 Paris Cedex 16, France
[2] Bio Intelligence Service, 20-22 Villa Deshayes, 75014 Paris, France
[*] Corresponding author. E-mail: *pmaupu@invivo-group.com*

ABSTRACT

InVivo, a French agricultural cooperatives union, has developed and implemented decision-making tools to help farmers manage fertilisation and plant protection. These tools enable pedologic, climatologic and agronomic data collection for each agricultural field on a farm, creating a traceability system allowing for field-scale LCA indicators calculation.
An LCA was conducted on winter wheat crop based on real agricultural practices for crops harvested in 2009, involving six cooperatives, 13941 fields, and covering 95432ha in production basins across France. Several indicators were calculated: water consumption, greenhouse gas emissions, marine and freshwater eutrophication, primary energy consumption, acidification, freshwater ecotoxicity. Every practice necessary for grain production and harvest was considered: cultivation, sowing, fertiliser and pesticide production and application, fuel consumption. Transportation to the grain silo and storage were subsequently considered.
The main conclusions are: (1) regarding impacts on freshwater ecotoxicity, pesticide use and fertiliser use are predominant, (2) regarding the other LCA indicators, fertiliser production and fertiliser use are predominant (3) results are highly scattered when observed among fields; much less among production basins (4) results depend on the product's technological quality.
The next step will be to identify ways to improve LCA results indicator by indicator by improving agricultural practices. InVivo's advice and decision-making tools prove suitable for mass traceability and mass environmental impacts minimization.

Keywords: Agricultural LCA, Ecodesign, wheat, traceability, mass improvement

1. Introduction

Life cycle assessment (LCA) is the leading methodology for the environmental impact assessment of a product for two main reasons: it allows for the calculation of several indicators corresponding to different environmental impacts (global warming, eutrophication, primary resources depletion…), and takes into account all relevant steps of the life cycle of a product, from the production of raw materials to the end of life.

When it comes to agricultural products, data collection constitutes a real stake. Indeed 1 kg of wheat grain sold contains wheat coming from several farms, from several fields. Each of these fields is managed following different agricultural practices and under different pedoclimatic conditions.

In light of the growing concern of the food industry for environmental responsibility, it becomes more and more important for their suppliers to know specifically what the impact of the raw products they produce is. This knowledge must cover the raw products variability, taking into account their production conditions, their technological quality…

For example, regionally aggregated data on wheat production and its potential impacts are not likely to be representative of the cooperatives members' data within the same region: mean regional data do not convey the fact that cooperatives are passing information, advice and tools on to their members, effectively influencing their way of producing wheat, as well as other agricultural products. Thus, it appears that using field-scale recorded data on real practices would be the best solution for cooperatives who want to assess the potential impacts of their own products. In addition, the field scale is relevant for impact assessment because that is the scale at which farmers make decisions and it is also the scale that allows aggregation at higher scales such as the farm or the production basin.

Thus, most LCA studies on agricultural products come up against the issue of obtaining real data on agricultural practices. The potential impacts of agricultural products are indeed closely related to those agricultural practices: inputs (fertiliser, fuel and pesticides) consumption vary a lot between fields, as well as yield and emissions that are highly dependent on both agricultural practices and pedoclimatic conditions.

InVivo is a French union of agricultural cooperatives that produces and sells, among other products and services, decision-making tools to help farmers manage fertilisation and pesticide application. With the help of cooperatives and thanks to these tools, pedologic, climatologic and agronomic (in particular fertilisation and pesticide applications practices) data is collected and recorded for each agricultural plot of a farm.

The main objectives of this study are (1) to assess the environmental impact of winter wheat, using a Life Cycle Assessment (LCA) covering field practices, transport and storage of the grains until the grain is sold to a miller, (2) to identify the most impacting phases of the winter wheat life cycle, (3) to study the scattering of the impact indicators results at two scales: field scale and production basin scale (4) to examine the effect of winter wheat technological quality on its environmental impacts. Over all, this study will make it possible to say whether, as demanding as it may be, LCA based on field-scale data is as worthwhile as is expected.

2. Methods
2.1 Data collection

An LCA was conducted on winter wheat, based on real agricultural practices recorded for winter wheat harvested in 2009, on a sample of 13 941 fields, covering 95 432 ha in different production basins of France where six cooperatives, from different regions, collect their wheat. The real agricultural practices were directly recorded with a tool used by farmers to manage their practices and to carry out their products' traceability. To use this tool farmers have to record all their field operations. Through this system, data on pedology and agricultural practices in each field where the tool is used were gathered in a database and were analysed using the *Statistic Analysis System* (SAS) software. Thus, the results of this LCA are representative of farmers who use such decision-making tools.

The fields were divided into groups, corresponding to the 6 cooperatives' production basins, which were then further divided into second-range groups identified by their own agronomic experts. These production scenarios were thus characterised by different soil conditions and/or by different technologic qualities (protein content of the grains), i.e. by different agricultural practices. Such a classification allowed for a finer analysis; however, all results cannot be detailed here for obvious conciseness reasons.

Table 1. Description of the population

Cooperative's name	Number of fields	Surface of the sample (hectares)
Coop 1	437	3 273
Coop 2	2 198	23 701
Coop 3	70	703
Coop 4	1 908	7 831
Coop 5	7 709	50 937
Coop 6	1 619	8 987

2.2 System boundaries

The functional unit studied here is "1 kg of wheat grain, ready to be exported from the cooperative".

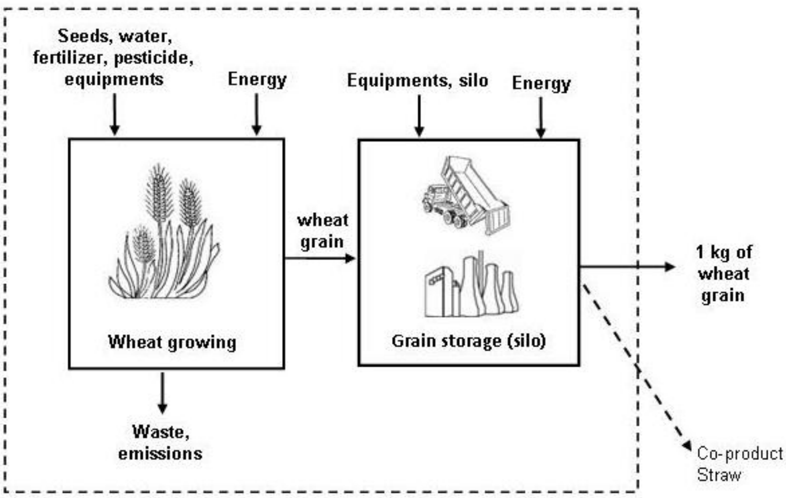


Figure 4. Studied winter wheat production system.

The agricultural practices considered in this study are cultivation, sowing, fertilising and pesticide application and fuel consumption on field to produce and harvest the wheat. Ecoinvent 2.0 life cycle inventories (Nemecek and Kägi, 2007) were used to consider the production of the inputs: wheat seed, fertilisers, pesticides, fuel, etc.

2.3 Calculation of particular fluxes and of freshwater Ecotoxicity

N_2O, NO, NH_3 NO3- and P_2O_5 emissions are fluxes that need particular calculation because they depend both on agricultural practices and on pedoclimatic and agronomic conditions. Their calculation requires the use of models, as reliable and recognized as possible.

Direct and indirect emissions of N2O and NO were assessed for each field separately, using the Bouwman emission factors (Bouwman et al., 2002). The calculation of NH3 emissions were made with the Ges'tim (Institut de l'élevage et al., 2002) and Corpen (2006) methods. The NO3- emissions were estimated thanks to a calculation tool integrated into Epicles (one of InVivo's decision-making tools) which uses the Burns model (1976) to determine the part of nitrogen that is leachable during winter rainfalls. For the P2O5 emissions, the SALCA-P method was used (Prasuhn 2006 and Nemecek et al., 2007).

The freshwater ecotoxicity impacts of pesticide applications were assessed for each field separately, using characterisation factors (CF) from USEtox (Rosenbaum et al., 2008) in line with the method previously used in a published study by Berthoud et al., (2011).

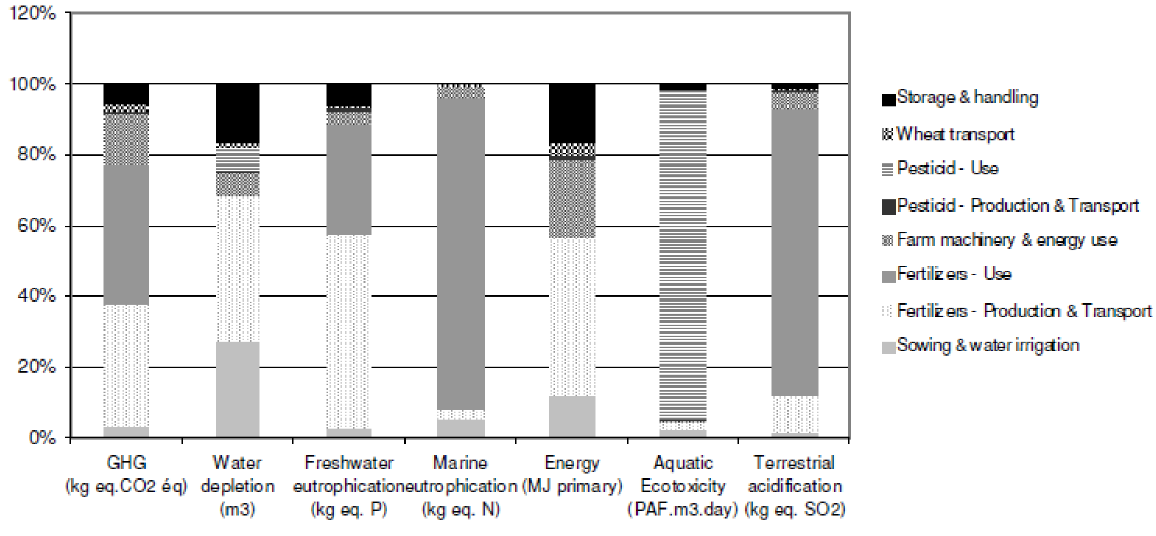
2.4 Characterisation

A tool developed on Microsoft's Excel by Bio Intelligence Service was used to make the characterisation calculations, in compliance with several characterisation methods, depending on the LCA indicator.

The global warming impacts of the inventoried fluxes were assessed with the IPCC method. The RECIPE method was used to characterise the fluxes respectively relevant to freshwater and marine eutrophication, and terrestrial acidification. Water depletion and non-renewable energy indicators were calculated according to a simple fluxes assessment. As for freshwater ecotoxicity impacts, they were assessed thanks to the USE-tox method, as detailed in the previous paragraph.

3. Results

3.1. Environmental impact of winter wheat



min	0,31	1,17E-03	8,57E-05	3,16E-03	3,21	2,28	4,50E-03
max	0,37	3,84E-03	2,21E-04	9,53E-03	4,33	88 ,3	7,98E-03
weight mean	0,34	2,13E-03	1,74E-04	4,79E-03	3,64	7,7	6,16E-03

Figure 5. Contribution of the several stages of the wheat life cycle to the LCA indicators results & Mean values of each LCA indicator for the different cooperatives involved in the study

The results show that the most impacting stages of wheat production on greenhouse gases emissions, marine eutrophication, terrestrial acidification and energy consumption are quite the same: nitrogenous fertilisers' production, transport and application on the field. This can be explained by the important emissions and energy consumption during the production of mineral fertilisers, as well as by the in-field emissions (N_2O, NO_3 and NH_3), which are linked to the application of both mineral and organic fertilisers. Farm machinery manufacturing and energy use also contribute in a significant way to greenhouse gases emissions and, logically, to energy consumption.

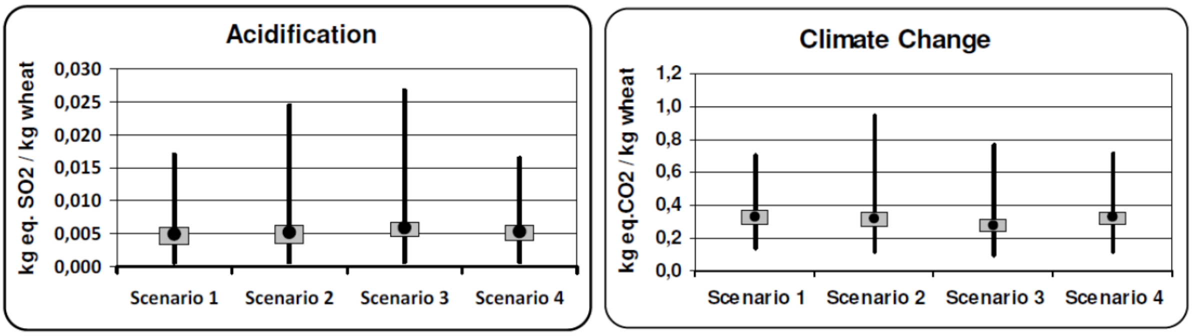
The impact on water depletion depends on the quantity of water used for crop irrigation, but also on the phosphorous fertiliser production and, to a smaller extent, on storage.

Fertiliser production is the main stage impacting freshwater eutrophication, along with their application on the fields. Field emissions are mostly due to soil erosion that carry away P2O5 particles and are slightly linked with P-fertiliser application.

Aquatic ecotoxicity is largely impacted by the ecotoxicity profile of the plant protection products applied on the field.

3.2. Scattering of the impact indicators (GES & acidification)

Results concerning the scattering of the impact indicators are based on intra-cooperative data in order to free oneself from variability between wide regional pedoclimatic conditions. Results submitted here are those of one cooperative among the 6 involved in the study. This cooperative chose to subdivide its territory into 4 scenarios depending on soil type.



Plots Number	1270	2386	1256	2797
Mean	0,004898	0,005112	0,005336	0,005919

Plots Number	1270	2386	1256	2797
Mean	0,327657	0,320375	0,326721	0,276875

Figure 6. Inter-field variability regarding the climate change and acidification indicators

GHG emissions rank from 0,36 to 0,39 kg eq.CO2 / kg wheat grain, depending on the scenario. Statistical Student test (Table 2) shows that there is a significant difference between the 4 scenarios.

Potential acidification impact ranks from $5.4.10^{-3}$ à $6.5.10^{-3}$ kg eq.SO2 / kg wheat grain, depending on the scenario. Statistical analysis (Table 2) shows that there is a significant difference between the 4 scenarios.

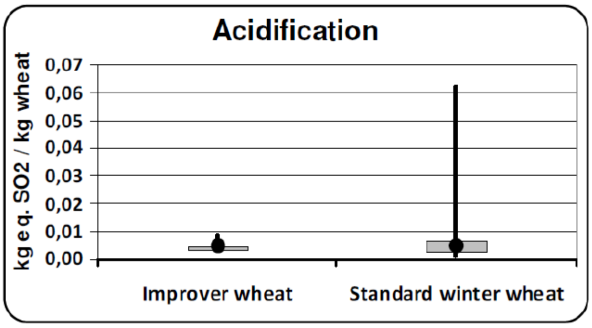
Beyond these observations on differences between scenarios, it is important to note that inter-field variability is substantial. It is much higher than variability between soil types. This is why in a given improvement strategy, it will be more efficient to work on the reduction of the impacts of the fields with the higher impact taken as a whole than to select wheat produced on the lower impacting soil.

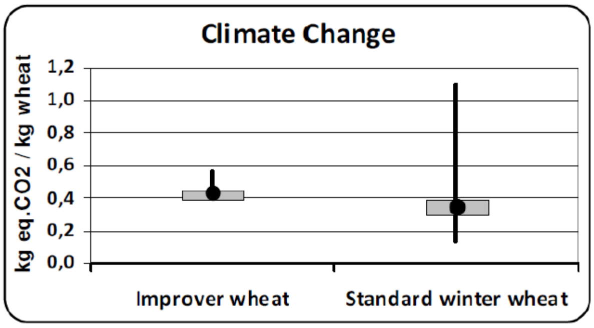
Table 2. Statistical comparison of the climate change and acidification indicators results of the 4 wheat production scenarios

Two-by-two comparison with a statistical Student test	**P-value**	
	Climate change	**Acidification**
Scenarios 3 vs. 1	6.04E-67	1.36E-38
Scenarios 3 vs. 2	3.83E-64	1.09E-35
Scenarios 3 vs. 4	3.74E-72	1.20E-13
Scenarios 4 vs. 1	1.48E-2	1.92E-06
Scenarios 4 vs. 2	3.43E-2	5.46E-3
Scenarios 2 vs. 1	7.84E-1	7.59E-3

3.3. Effect of winter wheat technological quality on environmental impacts

Results regarding the scattering of the impact indicators between high-protein wheat and winter wheat are also based on intra-cooperative data. The following results are those of one cooperative that chose to study its wheat's impacts depending on winter wheat technological quality.





Plots number	5	1 908
Mean	0,00496229	0,00531432

Plots number	5	1 908
Mean	0,43768124	0,34951817

Figure 7. Inter-field variability regarding the climate change and acidification indicators

With 0.42 kg eq.CO2 / kg wheat grain, improved wheat has a higher potential impact on climate change than standard winter wheat (0.35 kg eq.CO2 / kg grain wheat). According to the statistical Student test this difference is significant (p-value = 0.0170).

This difference can be explained by the fact that, having higher protein content grains, improver wheat needs more nitrogen to fill up its grains. When the functional unit is 110g protein rather than 1 kg of wheat grain, conclusions on the potential impact on climate change of improver wheat and standard winter wheat change: their respective GHG emissions are 0.36 and 0.40 kg eq CO2 / 110g protein.

On the other hand, when considering terrestrial acidification potential, improver wheat has a lower potential impact than standard winter wheat: its emissions are 4.4.10-3 kg eq. SO2 / kg grain when standard winter wheat ones stand at 4.9.10-3 kg eq. SO2 / kg grain wheat. This better result of improver wheat is due to the use of fertilisers that have lower NH3-emission rates (ammonium nitrate essentially). In this case, the fertiliser type enables to make up for the bigger use of fertilisers (reported to 1 kg wheat grain) by improver wheat. The change of the fertiliser type constitutes a key lever for improving the acidification indicator. It is also true for the climate change and energy consumption indicators (Berthoud and Rocca, 2011). According to the statistical Student test this difference is not significant (p-value = 0.7156).

4. Discussion

Carrying out such analyses at the field scale allows for more accurate results than at larger scales. This is especially true where cooperatives are involved, as they are key players who can thus put their agroenvironmental knowledge to good use.

However the main benefit of a field scale approach is to offer a wide range of improvement possibilities. For instance in another case study, the results of which could not be detailed here and which is separately published, we worked specifically on the ecotoxicity indicator. We were able to simulate a 50% decrease of the cooperative's mean impact regarding this indicator by identifying the most impacting practices (in this case, pesticide use) and substituting them for less impacting ones (Berthoud et al., 2011). There indeed lie the stakes of the variability study: to identify the most impacting practices in order to improve them, as well as to identify good practices.

The main limit to a field scale approach, however, is that it requires flux and impacts modelling tools and methods to be sufficiently practice-sensitive. To date, it is not (or to the least, not enough) the case for some indicators such as freshwater eutrophication.

5. Conclusion

What makes it particularly interesting to work on agricultural field practices is to address variability. The knowledge of this variability is a crucial asset: understanding, identifying the most and the less impacting practices, progressing towards an eco-conception of agricultural practices or even towards a very early consideration of potential impacts through the agricultural advising stage by informing advisers of the potential impacts of the agronomical solutions they recommend.

Similar studies have been carried out by InVivo Agro Solutions, with real data for agricultural practices, on other crops, such as barley, maize, etc, thus creating an environmental information system. Moreover, each of these studies will be conducted during 5 years, allowing for the cooperatives to have an inter-annual view of their impacts.

The recording of agricultural practices for each agricultural plot of a farm using decision-making tools or traceability tools is priceless for the cooperatives. Some of them have already started to put it to good use in their commercial relationships, meeting their clients' need for environmental information about agricultural products. The next challenge cooperatives dealing with their environmental impact will have to take up is data confidentiality and environmental services payment. Under those two conditions, the high variability of the impacts of agricultural raw materials will be reflected in the impacts of transformed products, and so will their impacts improvements be reflected in the impacts of transformed products.

6. References

Berthoud A., Maupu P., Huet C., Poupart A., 2011. Assessing Freshwater Ecotoxicity of agricultural products in Life Cycle Assessment (LCA): a case study of wheat using French agricultural practices databases and USEtox model. Int. J. LCA 16, 841-847.

Berthoud A., Rocca C., 2011. Réalisation d'une ACV sur les 3 principales formes d'azote: mise en évidence d'un effet forme au niveau technique et environnemental. COMIFER GEMAS 2011. http://www.comifer.asso.fr/images/stories/pdf/10emes_rencontres/berthoud/1%20-%20invivo%20-%20berthoud%20-%20article.pdf

Bouwman AF., Boumans LJM., Batjes NH., 2002. Modeling global annual N2O and NO emissions from fertilized fields. Global Biogeochemical Cycles 16(4).

Burns I.G., 1976. Equations to predict the leaching of nitrate uniformly incorporated to a known depth or uniformly distributed throughout a soil profile. Journal of Agricultural Science 86, 305-313.

CORPEN, 2006. Des indicateurs Azote pour gérer des opérations de maîtrise des pollutions à l'échelle de la parcelle, de l'exploitation et du territoire.

Institut de l'élevage, IFIP, ITAVI, Arvalis, CETIOM & ITB. 2009. Ges'tim - Guide méthodologique pour l'estimation des impacts des activités agricoles sur l'effet de serre. Version provisoire 1.2, mai 2010.

Nemecek T, Kägi T., 2007. Life cycle inventories of agricultural production systems. Ecoinvent report No. 15. FAL Reckenholz, FAT Tänikon, Swiss center for life cycle inventories, Zürich and Dübendorf.

Rosenbaum R.K., Bachmann T.M., Swirsky Gold L., Huijbregts M.A., 2008. USEtox- the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle assessment, International Journal of Life Cycle Assessment 13, pp. 532-546.

Prasuhn V., 2006 Erfassung der PO4-Austräge für die Ökobilanzierung – SALCA-Phosphor. Agroscope FAL Reckenholz, 22 p.

LCA of starch potato from field to starch production plant gate

Godard Caroline[1,*], Boissy Joachim[1], Suret Camille[1], Gabrielle Benoît[2]

[1] Agro-Transfert Ressources et Territoires, 2 Chaussée Brunehaut, 80200 Estrées-Mons, France
[2] INRA-AgroParisTech, UMR 1091 Environnement et Grandes Cultures, 78850 Thiverval-Grignon, France
[*] Corresponding author. E-mail: *c.godard@agro-transfert-rt.org*

ABSTRACT

To provide accurate agricultural LCA, the local production conditions, namely crop management techniques, weather and soil conditions, need to be taken into account. To develop adapted inventory methodology, a specific LCA study was carried out on a northern France starch potato supply area. It focused on the upstream steps and used specific crop management and logistics data. To improve inventory methods, the approach is based on process-based models simulating soil carbon dynamic and in-field pesticide emissions. The results obtained for 1 ton of potato showed the influence of soil carbon dynamic on climate change impact that resulted in carbon release between 10% and 18%. This level was mitigated by the soil carbon sequestration effect from the preceding catch crop. The soil type influence was limited due to rather homogenous pedoclimatic conditions. Nevertheless, the proposed approach enabled to account for specific cropping conditions and was designed to test various production scenarios.

Keywords: starch potato LCA, inventory methods, emission models, soil organic carbon, pesticide emissions

1. Introduction

Starch currently provides basic molecules for many innovative industrial applications, mainly non-food processes. Potato is the most common crop that produces starch in Northern France. To provide LCA of starch derived molecules and products with accurate and consistent data, a focus was made on the upstream processes, from potato field production to the gate of the starch processing plant. To do so, a specific LCA study was carried out on the supply area of a starch production plant located in Picardy. We were thus also able to provide local stakeholders (producers, advisers) with the environmental impacts of their production chains. There are currently scant literature references on the LCA of potato crops, moreover, most of them focus on food potato (D'Arcy et al., 2010; Williams et al., 2010), which involves crop management practices different from those used for starch potato. Hence, to provide adapted and accurate impact assessment, we used technical data from starch potato producers and specific logistics chain data. Those data were combined to in-field fluxes inventory methods using process-based models able to integrate soil and weather production conditions, and crop rotation. More precisely, two models were used to assess soil carbon dynamic and pesticide emissions. The objective of this study was thus i/ to identify the contribution of soil carbon dynamic in the global warming impact of starch potato upstream production process, and ii/ to focus on pesticide spraying which is one of the important potential environmental impacts of potato. Finally, we were also able to partly test the methodology developed for bioenergy chains (Godard et al., 2012) on another application field.

2. Methods

2.1. Studied system and functional unit definition

The studied area corresponded to the specific supply area of a starch production plant in the French Picardy region. A survey of potato growers showed that the main crop rotation including starch potato in this area was sugar beet/winter wheat/potato/winter wheat. An intermediate crop (white mustard) was sown before potato planting. The crop management technique sequence selected was the most common one described by local technicians and from producer survey (Table 1). The average distance between farm and starch production plant was considered to be 60 km, and a specific logistics chain is detailed in Figure 1.

The studied system entails all the field operations from the intermediate crop preceding potato to its harvest and transport and storage steps before starch production plant gate. All the machinery, the buildings and inputs necessary to those steps were accounted for: fuel and energy consumption, seeds, field fertilisers and pesticides, and storage treatment. The functional unit was the production of 1 t of starch potato (with a 22% dry matter content).

Table 1. Input summary of starch potato and catch crop.

Input (unit)	Value
Average annual yield (t fresh matter/ha)	52
Crop management (for 1 year)	
Stubble ploughing (runs)	1
Harrowing and catch crop seeding (runs)	1
Catch crop crushing (runs)	1
Ploughing (runs)	1
Harrowing (runs)	2
Sowing and ridging up (runs)	1
Haulm crushing (runs)	1
Lifting (runs)	1
Seeding rate (kg/ha)	2100
N fertiliser rate (kg N / ha)	180
K Fertiliser rate (kg K_2O /ha)	280
P Fertiliser rate (kg P_2O_5 /ha)	80
Magnesium fertiliser rate (kg MgO/ha)	30
Pesticide application (kg active ingredient/ha)	30.06

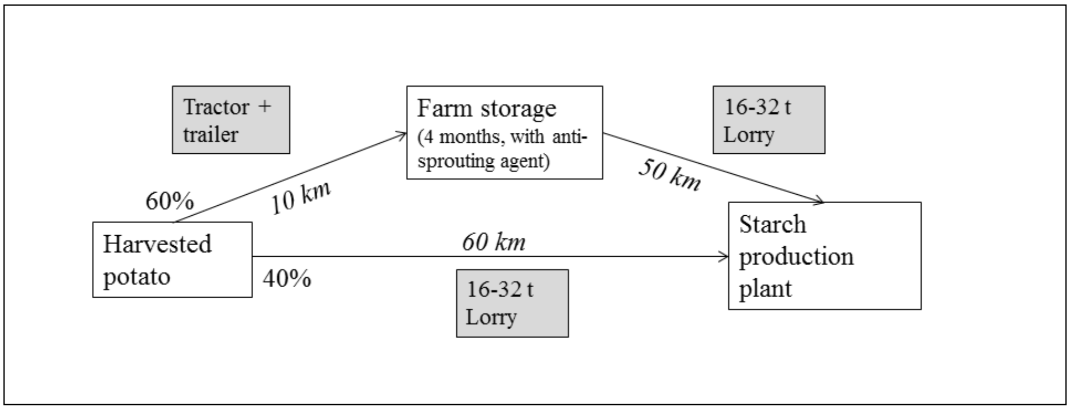


Figure 1. Logistics chain of starch potato.

2.2. Inventory methods

Life cycle inventories needed for the the manufacturing and supply of inputs and buildings were extracted from the Ecoinvent Database Version 2.2 (Swiss Center for Life Cycle Inventories, 2010).

In-field emissions of N and P were assessed using and adapting several inventory methods. Direct and indirect N_2O emissions were assessed according to the IPCC method (IPCC, 2006). NH_3 emissions to air were calculated using emission factors from Institut de l'élevage et al., 2010. The emission factor for NO_x emissions was derived from ADEME, 2010. Emissions of NO_3^- to water were estimated with a field N-balance method adapted from IFEU, 2000, and which integrates previous N fluxes (N_2O, NH_3 and NO_x). The N balance depended on crop rotation and soil type. P emissions in water by leaching, run-off and erosion were estimated according to Nemecek and Kägi, 2007. Soil erosion was estimated with the Universal Soil Loss Equation (Weischmeier and Smith, 1965).

2.3. Modelling approach for soil-carbon dynamic and pesticides emission estimates

To estimate soil C sequestration and pesticide emissions, the AMG (Saffih-Hdadi and Mary, 2008) and Pest-LCI (Birkved and Hauschild, 2006) models were used, respectively. AMG simulates the dynamics of humified organic matter, accounting for inputs from preceding and catch crop residues and their humification and mineralisation rates. The main inputs of the model are crop rotation and yields, soil management and properties (texture, organic matter and $CaCO_3$ content), and annual weather conditions. The model runs on a yearly time-step, and a 20-year series of past weather data (1988-2007) was used to simulate soil carbon sequestration. The initial Soil Organic Carbon (SOC) content, to which model predictions are very sensitive, was estimated for the typical starch potato crop-rotation and soil types combinations determined from measurements of soil organic matter changes in Picardy (Duparque et al., 2011). The major soil type in the studied area was a deep clayey loam that was selected for the parameterisation of AMG. The effect of soil type on the variations in soil C content over 20 years was also simulated. To do so, the next two soil types in terms of occurrence in the studied area (namely a deep loam and a clayey loam over chalk) were also input to the AMG model.

Pest-LCI simulates the fate of pesticides and their emissions during application and after-application, from soils and crop leaves. It simulates the fate of each fraction of pesticide reaching a compartment of the simulated system (air, crop, soil surface, water drainage system and groundwater). This model runs on a monthly time-step. It accounts for soil and climate conditions as well as bio-physico-chemical properties of the pesticide molecule. To ensure consistent results between pesticide emissions and soil C sequestration, the same past weather data and soil type as for the AMG simulation were used for Pest-LCI.

2.3. Impact assessment method

In order to focus on the main agricultural environmental impacts, five mid-point impact categories and corresponding reference substances were selected. Climate change (kg CO_2-eq), terrestrial acidification (kg SO_2-eq), freshwater eutrophication (kg P-eq), and marine eutrophication (kg N-eq) were calculated using Recipe method, version 1.05 (PRé Consultants, 2008). Ecotoxicity (Comparative Toxic Units – CTU) was assessed using USEtox method (Henderson et al., 2011), and energy consumption (MJ) was calculated according to the Cumulative energy demand method, version 1.08. All impact calculations were performed with SimaPro 7.3.2 software (PRé Consultants, 2011).

3. Results

3.1. Contribution analysis for starch potato

The hot spot for three impact categories out of six (Figure 2), namely climate change (CC), terrestrial acidification (TA) and marine eutrophication (ME) was nitrogen fertilisation which actually compounds the production of fertiliser N and the field emissions. Its share varied from 44% to 70% of the total impacts. The CC contribution of N-fertilisation mainly came from indirect greenhouse gases (GHG) emissions occurring during the production step of fertilisers, while for TA, the contribution was mostly due to the NH_3 emissions occurring after fertiliser application. The N-fertilisation contribution to ME arose from nitrate leaching.

For freshwater eutrophication (FE), the most impacting stage (with 69% of the total impact) was the other fertilisation step, PK fertilisation, mainly due to phosphate run-off and leaching after P-fertiliser application. Ecotoxicity (E) was in turn widely dominated by the contribution of pesticides (including both production step and in-field emissions) up to 67%. Contrary to other impact categories, cumulative energy demand (CED) originated from nearly all the life cycle steps with a similar level (between 6% to 19%), the transport phase (to the farm and to the plant) being the major contributor with 40% of the total impact. This transport phase often contributes as the second most impacting step to the other impact categories apart from CED.

One of the specific crop management techniques of potato is seeding. This step was the second after N-fertilisation to ME, mainly due to the nitrate leaching occurring after N-fertilisation during potato seed production step.

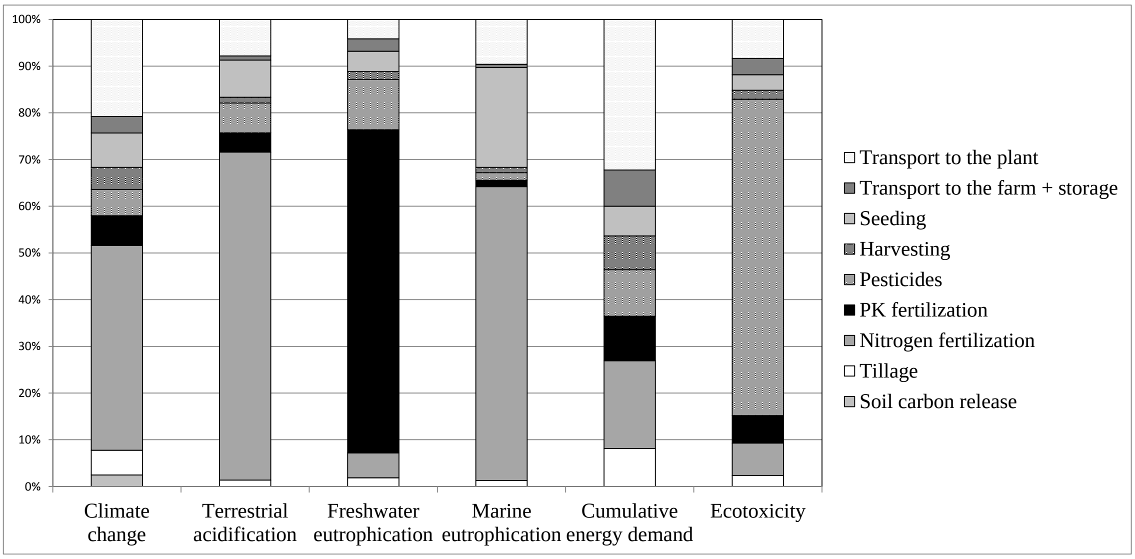


Figure 2. Contribution to the impact categories of each life cycle step from field production to starch plant gate. (CC impact includes both potato and catch crop effect on the deep clayey loam soil carbon).

3.2. Climate change impact and soil carbon dynamic: influence of soil type and intermediate crop

Soil C dynamics was influential in the CC impact, and enhanced the life-cycle GHG emissions of potato crops. Without considering the preceding intermediate crop, the latter always resulted in a release of soil C, from about 10% up to about 18% of the total GHG emissions (Table 2). Indeed all the potato haulms were exported out of field. Nevertheless, it can be noted that the catch crop preceding potato (whose impacts were allocated to the potato crop) strongly mitigated this release, with a systematic carbon sequestration reaching about 200kgC/ha/year on all the soil types. Thus the catch crop could even more than offset the soil C release of potato, and reaching, for example a C sequestration of 81 kgC/ha/year in the clayey loam over chalk.

There were few differences across the three main soil types on the CC impact. For potato crop only, the effect of the soil type on soil C dynamic was stronger than for the catch crop. Actually, the influence of soil type on soil C dynamics was limited because their properties were rather close in the AMG parameterisation (texture, organic matter and $CaCO_3$ content).

Table 2. Influence of soil type and catch crop on soil carbon variations expressed per t of potato produced. (a negative value indicates a soil C release corresponding to a CC impact increase)

Soil type (ordered by their area share)	**Soil carbon dynamic contribution to climate change impact**				**Climate change impact (kg CO_2-eq)**
	Without catch crop effect		**Including catch crop effect**		
	kg CO_2-eq	**%**	**kg CO_2-eq**	**%**	
1. deep clayey loam*	-16.7	-15.6	-2.6	-2.4	106.7
2. deep loam	-19.5	-17.8	-5.6	-5	109.5
3. clayey loam over chalk	-9.8	-9.8	5.7	5.7	99.8

* Refers to the situation represented in Fig 2.

4. Discussion

4.1 Main contributing steps and comparison with other studies

The comparison with Williams' et al., study (2010) was only possible for common indicators, as the characterising method was not the same as the one we used. The two studies showed the same order of magnitude for the CED impact (respectively 1.4 MJ/t for Williams et al., and 1.13 MJ/t in our case). Compared to the study by Williams et al., 2010, the CED proportion due to cool storage of potato is lower in our study (8% for transport+ farm storage here versus 49% for storage only for Williams et al.). This difference is certainly due to a limited storage for starch potato (40% of the harvested potato), contrary to a systematic one for food potato. In our study, the harvest step contributes to CED in the same order of magnitude as the Williams' one (7% vs 10%). D'Arcy's (2010) study showed on the contrary much higher energy consumption than in the present study (4MJ/t vs 1.13 for us). This is probably due the much lower yield they considered (28.1 t/ha in average) than the 52 t/ha we used in our case study.

4.2 Soil C dynamics integration in LCA and its effect on CC impact

We predicted the contribution of potato crop to soil C variations by simulating SOC dynamics with the AMG model. Our approach differs from Nemecek and Kägi, 2007, which is based on the C content of the biomass exported from the field, considering it a sink for atmospheric CO_2. A widespread, more practical alternative consists of considering crops as "climate-neutral", as Schmidt et al., 2004) did in their study of flax production. The latter two approaches actually disregard the effects of crop cultivation on soil C dynamics, let alone the effects of soil type, crop rotation or climate, which play a major role in the GHG balance of agricultural crops (Ceschia et al., 2010). Using a soil C model such as AMG is a means of overcoming this limitation and accurately predicts soil C sequestration or release rates. In the present example of starch potato, these rates may mitigate or, conversely, increase the global warming impact of crops, depending on soil type and climate conditions. This modelling approach was then an alternative to the French reference from Arrouays et al., 2002, who gave a single C sequestration rate for several crops. Our estimates of C release of 0.02 Ct/DMt/year was far different from the sequestration of 0.008 Ct/DMt/year given by Arrouays et al., 2002, for French food potato. Their approach was maybe too generic to account for the specificities of starch potato growing in a particular supply area.

Moreover, the AMG model includes crop rotations in its simulations of mid to long term soil C dynamics, and in the present case, the starch potato crop rotation always sequestrated soil C, despite the potato contribution as a net C release. This raises the question of the accounting of crop rotation and the allocation of catch crops in the soil C sequestration assessment. Indeed, as we showed for starch potato, the allocation of catch crop to the following main crop can result in an opposite effect on soil C dynamics.

Beyond global warming impact assessment, another reason to use AMG is that SOC is considered a relevant indicator of soil quality for LCA (Brandão et al., 2011; Milà i Canals et al., 2007a; Milà i Canals et al., 2007b). Thus, accounting for soil quality in LCA could be facilitated by the use of SOC models such as AMG.

4.3 Modelling approaches in agricultural LCA

This study showed the relevance of using emission models instead of using default emission factors in the life-cycle inventory to account for the characteristics of a crop supply area. Indeed this approach makes it possible to integrate the diversity of cropping production systems in supply areas in agricultural LCAs. Modeling approaches have already proven to be able to integrate various biophysical and technical crop production conditions in agricultural LCA, as in the studies from Adler et al., 2007; Gabrielle and Gagnaire, 2008. We were able to integrate the specific characteristics of crop management, logistics and storage in a supply area as well as its pedo-climatic characteristics by the use of the two models AMG and Pest-LCI. Beyond soil carbon dynamics and pesticides, crop models can provide precise assessments of in-field fluxes, and particularly N-fluxes which are highly dependent on local conditions. Nevertheless their use remains unusual, since they involve numerous parameters, some which are note easily available. An alternative way to these crop models are developed balance, such as Sundial used by Williams et al., 2010, thus limiting the parameterisation difficulty, and at the same time integrating crop rotation, crop management practices and pedoclimatic conditions in LCAs.

5. Conclusion

The approach proposed here has already been tested for a different context and for other crops, namely biomass feedstocks (Godard et al., 2012). It is a promising way to better account for the spatial variation of crop production conditions in agricultural LCA, by the integration of this variability range in model parameterisation. This kind of approach is relevant to test new production scenarios, such as the reduction of pesticide application, or the change in a crop supply and production area. It is also a good way to better account for geographical aspects in decision making, by providing adapted and accurate LCA results to local stakeholders.

6. References

ADEME, 2010. Analyses de Cycle de Vie appliquées aux biocarburants de première génération consommés en France. Rapport final de l'étude réalisée pour le compte de l'ADEME, du MEEDD et du MAAP et de FranceAgriMer par Bio Intelligence Service. In French.

Arrouays, D., Balesdent, J., Germon, J.-C., Jayet, P.-A., Soussana, J.-F., Stengel, P., 2002. Mitigation of the greenhouse effect Increasing carbon stocks in French agricultural soils? Scientific Assessment Unit for Expertise, Assessment Report by the French Institute for Agriculture Research (INRA) on request of the French Ministry for Ecology and Sustainable Development. Paris.

Birkved, M., Hauschild, M. Z., 2006. PestLCI--A model for estimating field emissions of pesticides in agricultural LCA. Ecological Modelling. 198 (3-4), 433-451.

Brandão, M., Milà i Canals, L., Clift, R., 2011. Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. Biomass and Bioenergy. 35 (6), 2323-2336.

Ceschia, E., Béziat, P., Dejoux, J. F., Aubinet, M., Bernhofer, C., Bodson, N., et al., 2010. Management effects on net ecosystem carbon and GHG budgets at European crop sites. Agriculture, Ecosystems & Environment. 139 (3), 363-383.

D'Arcy, A., O'Halloran, J., Sage, C., 2010. An LCA of potato production in Ireland: impacts on ecology and environment. In: Università degli studi di Bari Aldo Moro, editor. VII international conference on Life Cycle Assessment in the agri-food sector. Bari, Italy.

Duparque, A., Tomis, V., Mary, B., Boizard, H., Damay, N., 2011. Le Bilan Humique AMG, pour une démarche de conseil fondée sur des cas-types régionaux. In french. 10émes rencontres de la fertilisation raisonnée et de l'analyse COMIFER-GEMAS. Reims, France.

Godard, C., Boissy, J., Gabrielle, B., 2012. Life-cycle assessment of local feedstock supply scenarios to compare candidate biomass sources. Accepted to Global Change Biology Bioenergy.

Henderson, A., Hauschild, M., van de Meent, D., Huijbregts, M., Larsen, H., Margni, M., et al., 2011. USEtox fate and ecotoxicity factors for comparative assessment of toxic emissions in life cycle analysis: sensitivity to key chemical properties. The International Journal of Life Cycle Assessment. 16 (8), 701-709.

IFEU, 2000. Bioenergy for Europe: Which ones fit best ? A comparative analysis for the Community. Heidelberg: IFEU.

Institut de l'élevage, IFIP, ITAVI, Arvalis Institut du végétal, Cetiom, ITB, 2010. Guide méthodologique pour l'estimation des impacts des activités agricoles sur l'effet de serre - GES'TIM. version1.2. In french.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Agriculture, Forestery and Other Land Use.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., et al., 2007a. Key Elements in a Framework for Land Use Impact Assessment Within LCA. International Journal of Life Cycle Assessment. 12 (1), 5-15.

Milà i Canals, L., Romanyà, J., Cowell, S., 2007b. Method for assessing impacts on life support functions (LSF) related to the use of "fertile land" in Life Cycle Assessment (LCA). Journal of Cleaner Production. 15, 1426-1440.

Nemecek, T., Kägi, T., 2007. Life cycle inventories of Agricultural Production Systems - Data v2.0. EcoInvent report No.15. Zürich and Dübendorf: Agroscope Reckenholz-Tänikon Research Station ART. pp. 360.

PRé Consultants, 2008. Sima Pro Database Manual, Methods library PRé Consultants.

PRé Consultants, 2011. Sima Pro 7.3.2 software. Amersfoort, The Netherlands.

Saffih-Hdadi, K., Mary, B., 2008. Modeling consequences of straw residues export on soil organic carbon. Soil Biology and Biochemistry. 40 (3), 594-607.

Schmidt, A., Jensen, A., Clausen, A., Kamstrup, O., Postlethwaite, D., 2004. A comparative Life Cycle assessment of building insulation products made of stone wool, paper wool and flax. The International Journal of Life Cycle Assessment. 9 (1), 53-66.

Swiss Center for Life Cycle Inventories, 2010. Ecoinvent database v2.2. Tänikon, Switzerland.

Weischmeier, W., Smith, D., 1965. Predicting rainfall-erosion losses from cropland east of the Rocky Mountains: Guide for selection of practices for soil and water conservation.

Williams, A., Audsley, E., Sandars, D., 2010. Environmental burdens of producing bread wheat, oilseed rape and potatoes in England and Wales using simulation and system modelling. The International Journal of Life Cycle Assessment. 15 (8), 855-868.

Barilla EPD Process System to increase reliability, comparability and communicability of LCA studies

Luca Ruini[1], Laura Marchelli[1], Massimo Marino[2], Assunta Filareto[2,*]

[1] Barilla G. R. Fratelli S.p.A, via Mantova 166, 43122, Parma, Italy
[2] Life Cycle Engineering, via Livorno 60, 10144, Torino, Italy
[*] Corresponding author. E-mail: *filareto@studiolce.it*

ABSTRACT

The aim of this work is to show how a large company integrates the life cycle approach into its policies and views to reduce the footprint using reliable data suitable also for communication purposes. During 2010 an internal LCA process was implemented and in February 2011 certified, being the first private company, in compliance with the Process Certification Clarifications guidelines for International EPD® System to perform environmental impact calculation in an easy, quick and reliable way and to provide certified and published results. Barilla's EPD internal process is based on three main elements: the LCA database, the Product System and the Product Specific data. They are used together as a funnel process: data from the database and from product specific information are processes by the product system tool to have the specific LCA data sheet results, used for a type III label (EPD – Environmental Product Declaration) preparation. The reliability of the system is guaranteed by both internal and external verification.

Keywords: Product Category Rules, Environmental Product Declaration, EPD Process System, LCA for food, verified database

1. Introduction

Barilla, one of the top Italian food groups, produces more than 100 products in about 50 plants around the world. The company has been using the LCA for more than a decade. Since 2008, life cycle thinking made its way into company strategy, as an instrument to thoroughly study the production chain and localise the most substantial environmental impacts.

Barilla decides to join the International EPD System for several reasons: the System acts following the International Standards (ISO 14025); the reliability of the LCA is assured by the Product Category Rules (PCR); the System allows the comparability among the same product group, each document with a public interest (such as Product Category Rules (PCR) and General Program Instruction (GPI)) is published; public register on PCR and EPD is regularly updated; EPDs and LCAs must cover all the environmental issues not merely focusing on greenhouse gases emissions; the System gives the possibility to develop an EPD Process Certification.

Barilla's aim is to develop the EPDs for the major part of its product and the only way to make it in an easy, simple and reliable manner is to use an EPD Process System; for this reason, during 2010, it was developed and certified by Bureau Veritas in 2011.

The scope of the Process System is to prepare, verify and publish EPDs for Barilla's products related to the following Product Category Rules:

- Product Category Rules 2010:01 (CPC 2371): Uncooked pasta, not stuffed or otherwise prepared
- Product Category Rules 2012:06 (CPC 234): Bakery Products
- Product Category Rules 2010:09 (CPC 23995): sauces; mixed condiments; mustard flour and meal; prepared mustard

2. General Structure of the Barilla EPD Process

All EPDs coming from the Barilla's EPD Process System are based on the Life Cycle Assessment methodology; using the following three main elements:

1. The Product Specific data
2. The LCA dBase
3. The Product System

The system works like a "funnel process", as showed in Figure 1. product specific information are collected and elaborated by the product system using the LCA dBase, then results are collected in a specific LCA data sheet, that is then used for the preparation of the EPD.

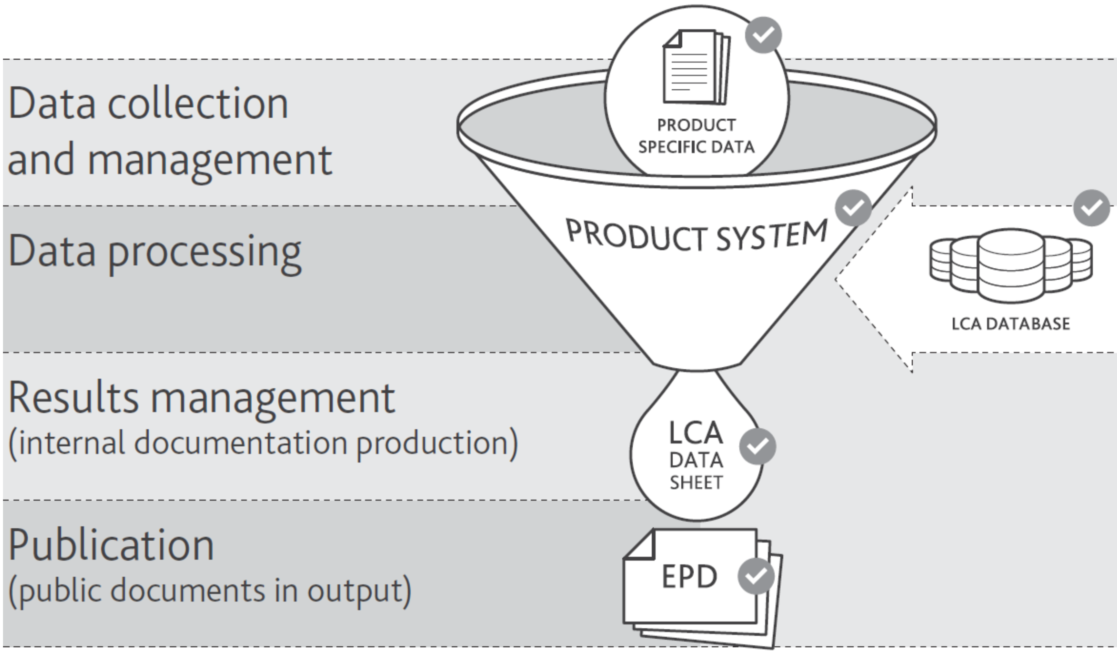


Figure 1. Scheme of the Barilla EPD Process System ("funnel process")

2.1. Product specific data

Product specific data represent all the specific information related to the product that has to be analysed, they have to be collected for each EPD and include the following specific information:

- Product recipe: includes the amount of food raw materials per unit of product (e.g. kg of sugar or vegetable oil or flour etc per kg of product)
- Bill of materials packaging list: that includes the amount of packaging materials used for product packaging
- Production plants where the product is manufacturing
- Production volume per each plant involved
- Finished product logistic distribution data (kilometers covered and transport means)
- Other relevant environmental aspects, such as liquid nitrogen and carbon dioxide consumption used for product cooling

Figure 2 provides an example of product system calculation with the relationship between product specific data and LCA dBase.

2.2. LCA dBase

The database is organised among different data modules groups:

- Raw materials: includes information about materials used for food product recipe (e.g. durum wheat cultivation for semolina production)
- Packaging raw materials (e.g. cardboard manufacturing for American box production);
- Energy: includes data about the energy mixes used in the countries in which the Barilla's plants are located. The database is updated every time new information is available;
- Plants: contains information about the processes that take place in the Barilla's plants. These data are based on the data collection and they are updated every year.
- Transports: data on the main means of transport used for the Barilla's purposes

Each data module contains all the environmental aspects related to material or process, main hypothesis applied, as requested by the ISO 14040 series (functional unit, system boundaries, data quality, data collection and treatment, allocation and cut-off rules).

All data modules are internally verified and are ready to be used for EPD purposes, they are inserted in software SimaPro®, that was selected as the modelling and calculation tool for the Barilla EPD system process.

2.3. The Product System

The Product System represents the product group model calculation tool. It is developed for each product group in a specific fashion following the Product Category Rule (PCR) and is internally vetted.

Barilla's EPD Process System includes Product Systems for pasta, bakery and sauces products. An example of product system for bakery product is reported in Fig. 2.

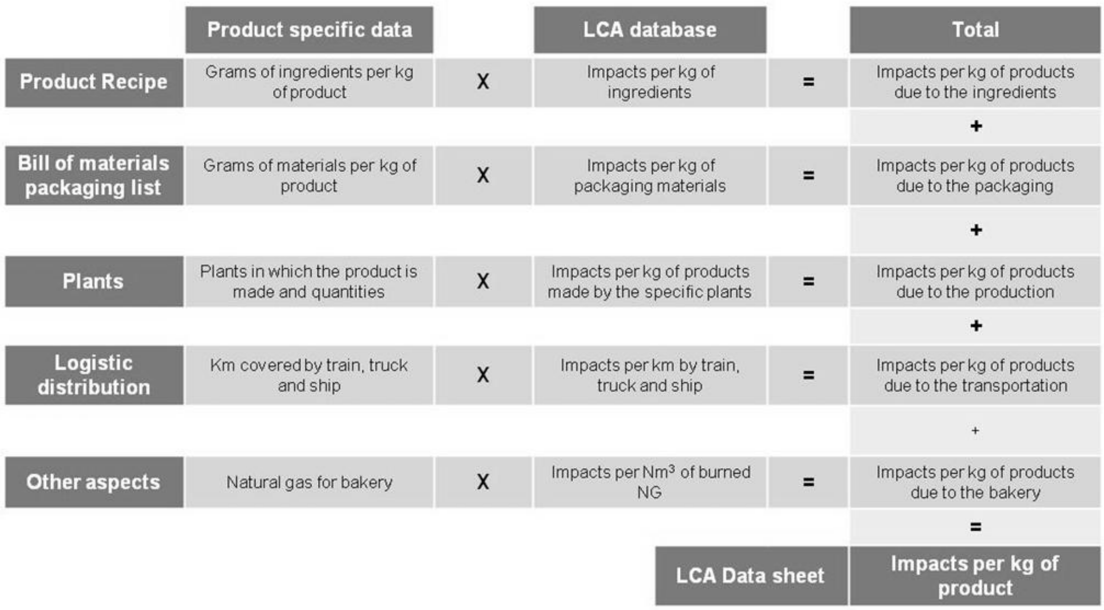


Figure 2. Example of product system for bakery products

2.4. Verification levels

The reliability of the EPDs is ensured by several verification levels done by Data Assessor, Process Assessor and Verification Body:

1. Product System and LCA Database verification is performed by the Data Assessor;
2. Product specific data, LCA data sheet and EPD Document verification is performed by the Data Assessor per each EPD realised
3. EPD Process verification by means of:
    - internal audit, performed by the Process Assessor
    - external audit, performed by a Verification Body (accredited body certified for audit of management systems)

3. Process Operations

Barilla EPD Process System is organised in three main processes, under the control of the management activities: EPD project, database update and product system update.

The management activities take into consideration all the actions that are necessary for activities coordination and organisation, such as EPDs planning, competences evaluations, process assessment planning, non conformity management and system documentation updating. An overview of the processes is given in Fig. 3.

The first activity of the system is the EPD planning, it is performed each year to organize all the works related to the EPD Process System. To reliably plan the EPD projects, the collection of all the product recipe is necessary to identify raw material still not covered by an update data module.

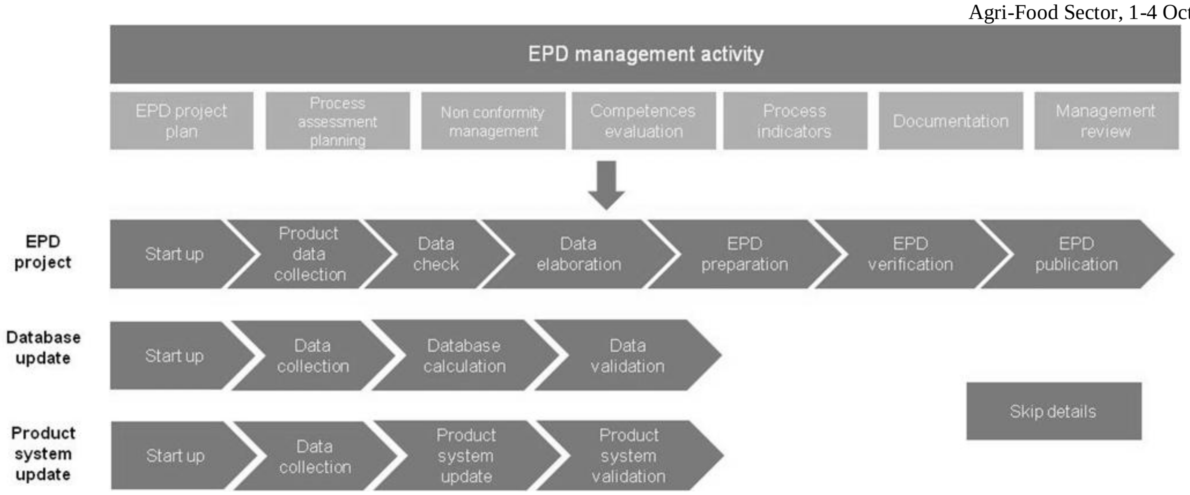


Figure 3. Overview of the process operations

The main process of the system is the EPD Project, which leads to the verification and publication of the EPD document, starting from the Product data collection and passing through data check and elaboration and EPD verification.

Database update is performed each time data must be updated (e.g. for energy mix) and at least once a year. In addition, data is updated during the data check of the EPD Project when data is unavailable for the model. This puts the EPD project process in standby and the database update process starts. The EPD Project process resumes only when all data necessary for the EPD preparation is available and validated.

The product system update process allows to update the product system model when there is a change to its product category rules and compiles a new product system when a new product must be analysed and inserted into the system.

4. Process Indicators

The Barilla EPD process performances are evaluated by mean of specific indicators (Table 1).

Table 1. Overview of the indicators used for measuring the EPD Process performances

Indicators	Unit	Description
Product volume covered by EPDs	%	Percentage of product volume covered by EPDs
Planned projects	n°	Number of the EPD projects planned each year (one EPD project may have one or more products)
Open Projects	n°	Number of the EPD projects that are still open in a specific moment
Frozen Projects	n°	Number of the EPD projects that are stopped because a database/data system update is running
Validated EPD	n°	Number of validated EPD (not all of them are published)
Published EPD	n°	Number of published EPDs available
Product System	n°	Number of product system available for all the Barilla products
Product System validated	%	Percentage of total product system validated and available for EPD realization
Product Volume covered by Product System	%	Percentage of product volume covered by Product System
Total module	n°	Total amount of the data modules that are needed for completing the EPD activities included in the running project.
Available data module	%	Percentage of the total data module available for EPD realization. It represents how much the data collection performance is completed.
Validated data module	%	Percentage of the total data module that are validated and ready for the EPD calculation. It represents the measure on how much the database is completed with validated information.

5. Actors and roles

EPD Process management is guaranteed by the mutual works of different actors: EPD process owner, LCA developer, data owners, data expert. All roles are described below:

- EPD Process owner: is the EPD system process responsible who has decision-making power and represents Top management for the EPD purposes; defines the policy and approves all documents and decisions related to EPD issues, avails himself of an EPD Process Manager;

- LCA developer: is supported by an LCA team, that manages all the activities necessary for the EPD document preparation, data modules and product system development and update;
- Data owners are in charge of providing data and information needed for LCA calculations. They usually have precise functions and are responsible for specific areas (e.g. packaging production, production process, product transport, etc). They are identified and involved in data collection according to the annual EPD work plan and they have to know the procedure for the data collection;
- Data expert represents personnel that could assist both specific data verification (peer review) during LCA calculation and EPD preparation. A data expert may be identified during the management review to support data collection and verification during LCA calculation. A data expert may be sought for strategic and relevant information such as wheat cultivation, palm oil production, etc. This figure can either be an internal or external resource;

System reliability is guaranteed by several verifiers (data assessor, process assessor and verification body), their roles are described below:

- Data assessor: is personnel responsible for the verification of the LCA calculation and of the EPD document. The data assessor conducts internal assessments at planned intervals to determine the reliability, relevance and independence of the EPD;
- Process assessor: is an internal verifier that regularly assesses the conformity of the EPD process. The process assessor is the internal verifier that has the responsibility to perform periodic audits on system application;
- Verification Body: represents an accredited body certified for audit of management systems that verifies the entire EPD process system.

Each actor in the process has qualified and formalised competences.

6. Results and Conclusion

Barilla is the first private company that has developed an EPD Process System.

About the 46% of the products put on the market by Barilla during year 2011 are covered by an Environmental Product Declaration (EPD). At 30th April 2012, fifteen EPDs were published on the website (http://www.environdec.com/en/EPD-Search/?query=barilla) and about six hundreds data modules were realised; the available data modules are over the 90% and validated data modules among the available ones are over the 75%. The use of the Barilla EPD Process System has shortened EPD publication timing, that now lasts about 6-10 weeks.

Table 2. Performance of the EPD Process System

Indicators	Unit	Data
Product volume covered by EPDs (year 2011)	%	46%
Planned projects (year 2012)	n°	39
Open Projects (point at 30/04/2012)	n°	13
Frozen Projects (point at 30/04/2012)	n°	0
Validated EPD (point at 30/04/2012)	n°	18
Published EPD (point at 30/04/2012)	n°	15
Product System (point at 30/04/2012)	n°	6
Product System validated (point at 30/04/2012)	%	67%
Product Volume covered by Product System (year 2011)	%	99,7%
Total module (point at 30/04/2012)	n°	610
Available data module (point at 30/04/2012)	%	97
Validated data module (point at 30/04/2012)	%	79

Table 2 shows the Barilla EPD Process System performances through the system indicators, from 2010 to April 2012. Looking at the table, it is important to point out that:

- There are 39 EPD projects planned for 2012; 13 of these contain more than one product to be analysed because there are several recipe variants for some products;
- There are no frozen projects because there were no problems with data availability;
- There is a higher number of validated EPDs respect to published EPD because it was decided to not publish three of the validated EPDs;

From 2010 to April 2012 forty verifications were performed: four external verifications made by Bureau Veritas, and the others made by data and process assessors for internal verifications.

7. References

ISO 14040 - Environmental management, Life cycle assessment, Principles and framework

ISO 14044 - Environmental management, Life cycle assessment, Requirements and guidelines

ISO 14025 - Environmental labels and declarations - Type III environmental declarations - Principles and procedures

International EPD Cooperation; introduction, intended uses and key programme elements; version 1 of 29/02/2008

International EPD Cooperation; General Programme Instructions for Environmental Product Declaration; version 1 of 29/02/2008

International EPD Cooperation; Supporting annexes for Environmental Product Declaration; version 1 of 29/02/2008

International EPD Cooperation; Process Certification Clarification (PCC) for the International EPD System; Guidelines; ver. 1.0 of 23/04/2010

International EPD Cooperation; Process certification clarification for the International EPD® system. Version 1.0 of 23/04/2010

International EPD Cooperation ; PCR 2010:01; CPC 2371: Uncooked pasta, not stuffed or otherwise prepared; version 1.1 of 18/06/2010

International EPD Cooperation; PCR 2012:06; CPC 234: Bakery Products; version 1.0 of 17/04/2012

International EPD Cooperation ; PCR 2010:09; CPC 23995: Sauces; mixed condiments; mustard flour and meal; prepared mustard; version 1.1 of 9/11/2010

www.environdec.com.

LCA-based hotspot analysis of food products to inform a major Chilean retailer's sustainability strategy

Jonas Bengtsson,[1] Cristián Emhart,[2] Alejandro Florenzano,[2*] Laura Guccione,[1] Carolina Jaramillo[2]

[1] Edge Environment Pty Ltd, 501/39 East Esplanade, Manly NSW 2095, Australia
[2] Fundación Chile, Av. Parque Antonio Rabat Sur N°6165, Vitacura, Santiago, Chile
[*] Corresponding author. E-mail: *aflorenzano@fundacionchile.cl*

ABSTRACT

This paper reports on a project with a major Chilean retailer to: engage their merchants and suppliers to focus efforts on science-based hotspots; provide a starting point for industry engagement to begin implementation of management practices based on life cycle information; and develop a nationally relevant product category specific measurement and reporting system. Product category life cycle assessments (PCLCAs) have been carried forward for nine food products – milk, chicken meat, blueberries, apples, grapes, avocados, wine, beer and pasta – using a commonly defined methodology. This includes normalisation and weighting factors specific to Chile, resulting in a weighted environmental impact score called the Daily Eco-Impact Indicator.

Keywords: life cycle assessment, PCLCA, food, Chile, retail, hotspot.

1. Background, aim and scope

Food production poses a major global sustainability challenge for the 21st century. Retailers hold a key position in the food supply chain to promote and enable sourcing and sale of more sustainable food products. This should be done by identifying and working towards reducing environmental impacts overall, not merely transferring impacts across media or stages of the product life cycle, but considering the whole product life cycle in improvement efforts.

This paper presenting the results of product category life cycle assessments (PCLCAs) produced by Edge Environment and Fundación Chile for a major Chilean retailer aiming to:

- Engage their merchants and suppliers to focus efforts on science-based hotspots.
- Provide a starting point for industry engagement to begin implementation of management practices based on life cycle information.
- Develop a product category specific measurement and reporting system.

To this end, PCLCAs have been carried forward to identify overall environmental hotspots of impacts and improvement opportunities for nine food products: milk, chicken meat, blueberries, apples, grapes, avocados, wine, beer and pasta. The physical scope of the studies is cradle-to-grave including raw material, production, distribution and storage, retail, consumer transport, consumption and disposal. The models have been built based on national and international studies, typical agricultural practices, and modelled using background information derived from international databases, adapted to Chilean conditions (e.g. electricity mix, recycling rates, transport distances).

Going forward, the studies and results reported in this paper are intended to be submitted for public comment, and the internationally sourced life cycle data to be improved with Chilean specific production data.

2. Methods

The studies were conducted to allow for the transparent and consistent product comparison by use of a common LCA methodology (ISO 14040 and 14044 compliant) aligned with the ongoing developments of The Sustainability Consortium (TSC) and the level playing field LCA methodology developed through the Australian Building Product Life Cycle Inventory project (BPIC, 2010) and AusLCI (2009). The modelling was performed with the SimaPro software package version 7.3.

2.1. Life cycle inventory data and functional unit

The cradle-to-grave models have been built based on published studies, typical agricultural practices and background information derived from international databases, adapted to Chilean conditions (e.g. electricity mix, recycling rates, transport distances).

The functional units used for this paper are based on a typical serving size (USDA) in order to allow the results to be presented on a similar scale. The functional units, main scenarios and key data sources are presented in Table 1. The main source of background life cycle inventory (LCI) data used within this study is a modified version of the Ecoinvent database version 2.2. The modification involves:

- Applying Chilean electrical conditions to the Ecoinvent database. All 3,952 unit processes contained in the Ecoinvent 2.2 dataset using electricity production/distribution from international regions were re-routed to the Central Interconnected System (SIC, Sistema Interconectado Central) electricity production and distribution (SIC serves the central part of the country, including Santiago).
- Applying a consistent economic allocation between products and co-products on major production input into the life cycles, primarily packaging materials for this study.

Table 1. Functional units and key data sources

Product	Serving quantity (functional unit)	Scenarios highlighted	Key data sources
Milk	1 cup (200 ml)	Cattle feed production	Franklin Associates (2006); Fundación Chile (2007); and Hospido et al., (2003)
Chicken meat	½ breast fillet (145 g)	Conventional and organic	Ahlmén et al., (2001); Boggia et al., (2010); LCA Food (2003); Pelletier (2008); RIRDC (2011); Ritz et al., (2004); and Williams et al., (2006)
Blueberry	1 cup (146 g)	Transport to market	INIA & Deuman (2010); Mila i Canals (2003); growers' manuals and best management practices
Apple	1 fruit (182 g)	Transport to market	
Grape	1 cup (92 g)	Packaging	
Avocado	½ fruit (100 g)	Transport to market	
Wine	1 glass (175 ml)	Packaging	Bosco et al., (2011); Gazulla et al., (2009); and WRAP (2007)
Beer	1 can (330 ml)	Packaging	Cordella et al., (2008); International EPD Consortium (2010); and Talve (2001)
Pasta	1 portion (57 g)	Wheat transport	Barilla & Life Cycle Engineering (2009); Bevilacqua et al., (2007); and Ruini & Marino (2008)

2.2. Allocation

The approach selected for this methodology is based on the use of economic allocation, which is one of the options included in the ISO 14044 standard for LCA when allocation cannot be avoided.

The justification for consistently apply economic allocation is that the process exists in the first place because of capital investment and the investors' anticipation of returns on that investment from the sales of the products/services that arise from the process(es). The operation of the process(es) is optimised to deliver economic return, and hence there is a clear cause (investment) to effect (economic return) from the value of the products that arise from the process(es). The extent to which each product or service contributes to the economic return from the operation of the process(es) is therefore the most appropriate unit that can be used for consistent allocation across the scope of products/services.

Other key reasons for selecting economic allocation are that it is the only means of allocation which can be applied consistently across all product sectors up and down the supply chain. It is also Edge Environment's experience that business and industry can easily and intuitively relate to using economic value as the key determinant of assigning proportional impact burden between products and co-products.

2.3. Impact Assessment

The impact assessment method used was the ReCiPe World Midpoint H (Goedkoop et al., 2009) with USEtox Recommended characterisation factors for human and eco-toxicity. The method was selected and customised based on the Sustainability Consortium's impact assessment bookshelf recommendations (version 2). Normalisation factors were calculated based on Chilean pollution and resource reports and statistics. Weighting factors were derived from a sample of public opinion and published opinion surveys. The weighted environmental impact score, which has been called the Daily Eco-Impact Indicator, is calculated so that a score of 100 corresponds to the average Chilean citizen's overall daily impact on the environment.

For normalisation Chilean factors were calculated based on national statistics, complemented with ReCiPe global factors if these factors are not available, to calculate the relative magnitude of each impact type. At this time there is only a preliminary indication of the relative importance of the mid-point indicators above based on three pilot workshops held in Santiago in September 2011. The normalised mid-point impacts have been aggregated assuming equal weighting within each weighting category (based on environmental issues used in Goedkoop et al., (2009)), as listed in Table 2.

Table 2: Weighting and normalisation factors and categories

Weighting	Weighting category/Environmental issues	Mid-point impact category	Normalisation (per day)
16.8%	Water depletion	Water depletion	6,030 L
15.5%	Climate change	Climate change	15.1 kg CO2 eq
9.7%	Human health due to PM10 and ozone	Photochemical oxidant formation	0.130 kg NMVOC
		Particulate matter formation	0.066 kg PM10 eq
9.7%	Toxicity	Ecotoxicity	12.1 CTUe
		Human toxicity, cancer	1.17E-09 CTUh
		Human toxicity, non-cancer	1.29E-08 CTUh
9.6%	Impacts of land use	Agricultural land occupation	56.9 m2a
		Urban land occupation	1.55 m2a
		Natural land transformation	0.00978 m2a
7.8%	Ozone depletion	Ozone depletion	3.48E-06 kg CFC-11 eq
7.0%	Ionising radiation	Ionising radiation	17.1 kg U235 eq
6.7%	Mineral resource depletion	Metal depletion	1.29 kg Fe eq
6.3%	Fossil fuel depletion	Fossil depletion	3.54 kg oil eq
5.5%	Acidification	Terrestrial acidification	0.177 kg SO2 eq
5.4%	Eutrophication	Freshwater eutrophication	0.00218 kg P eq
		Marine eutrophication	0.0411 kg N eq

3. Results

The results reported in this paper are based on best available data from a range of LCA and industry sources, compiled to a common methodology which includes reasonable adaptation to Chilean conditions. The underpinning PCLCA studies are intended to be submitted for public comment, and the internationally sourced life cycle data to be further improved with Chilean specific production data.

The results for the nine product categories, with two scenarios for each category, are expressed in the Daily Eco-Impact Indicator per impact category and life cycle stage. These results are not intended to be used for product differentiation in terms of environmental preference; the purpose is to illustrate how detailed results can be expressed using an equal single score metric.

3.1 Results by Impact Category

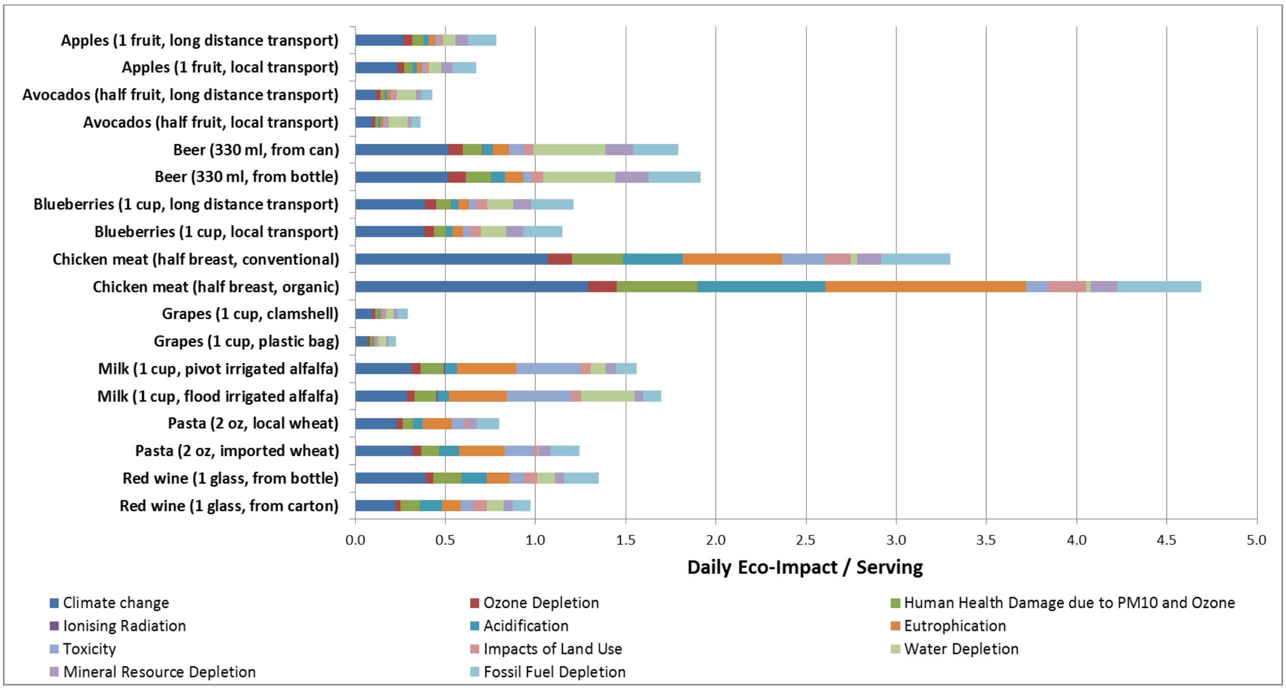


Figure 8: Impacts per weighted environmental impact category

Fig. 1 shows the results per weighted environmental impact category. The results confirm the common perception that greenhouse gas emissions, fossil fuel and water depletion are three key environmental concerns in the food life cycle. Across all product categories and scenarios:

- Climate change impacts make up between 17-35% (average 28%) of the overall impact. The highest relative impacts (>30%) in this category are in scenarios modelled for apple, grape, blueberry and conventional chicken meat life cycles.

- Fossil fuel depletion makes up between 6-20% (average 14%) of the overall impact. The highest relative impacts (>15%) in this category are in scenarios modelled for apple, grape, blueberry, pasta from local grown wheat and bottled beer life cycles.
- Water depletion makes up between 1-30% (average 12%) of the overall impact. The highest relative impacts (>15%) in this category are in scenarios modelled for milk from cows fed with flood-irrigated alfalfa, grapes in plastic bags, avocado and beer life cycles.

However on average lower range impacts:

- Eutrophication has a >20% impact for pasta, milk from cows fed with pivot-irrigated alfalfa and organic chicken meat.
- Toxicity impacts have a >20% impact for milk.

The least significant environmental impact categories, with less than 10% contribution across all assessments, are ozone depletion, ionising radiation, mineral depletion and impacts of land use. The relatively low impacts from land use are perhaps surprising considering the significant use of land resources for agricultural production.

3.2 Results by Life Cycle Stage

Fig. 2 shows the results by life cycle stage.

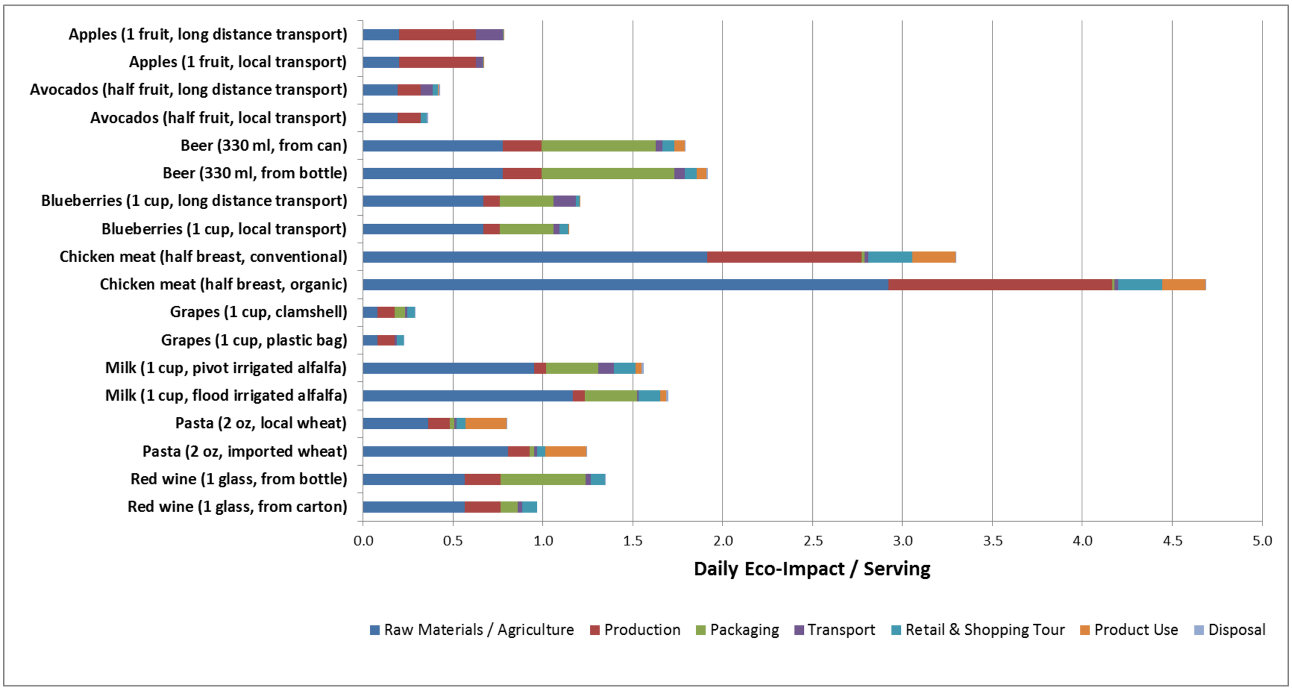


Figure 9. Weighted impacts per life cycle stage

The results show an emphasis of impact in the earlier life cycle stages with:

- Raw materials and agriculture making up 26-69% (average 49%) of the overall impacts. The highest relative impacts (>55%) in this stage are in scenarios modelled for carton-packaged wine, pasta from imported wheat, milk, chicken meat and blueberry life cycles.
- Production making up 4-63% (average 23%) of the overall impact. The highest relative impacts (>30%) in this stage are in scenarios modelled for avocado, apple, and grapes life cycles, all including chilled packinghouses.
- Packaging materials making up 0-39% (average 13%) of the overall impact. The highest relative impacts (>25%) in this stage are in scenarios modelled for beer, wine in bottles and blueberries transported locally.

Post-manufacturer's gate makes up an average of 15%, with the notable exception of pasta from local wheat where the use stage makes up 29% of the overall impact, primarily from cooking.

3.2 Benchmarking

Although not presented here, the characterised global warming potential impact results (i.e. climate change) are benchmarked with available international and national research in each PCLCA. The global warming potential impacts were consistently comparable with the other studies. This is not surprising since the underlying LCI data to a significant degree was adapted from the benchmark studies, as described in the paper. Only with indigenous data from industry can we develop a more confident product profile appropriate for Chile.

4. Discussion

The objective of identifying and reducing environmental impacts, and not merely transferring impacts across media or stages of the product life cycle, is best served by considering the whole product life cycle when setting product environmental criteria. That said, we cannot let lack of data or methods hold back progress. The spirit in which the work was progressed was to take a pragmatic and practical approach to overcome gaps, and to provide a starting point for improvement, rather than hold back in the absence of confirmed models and data. In this spirit the team developed provisional normalisation and weighting factors in order to help interpret the results. The assessments rely on unconfirmed characterisation models for regional impacts in Chile and the background data was adapted from almost exclusively non-South American process information.

An anticipated concern for the Chilean industry and stakeholders is that life cycle data and product characteristics are used correctly and appropriately in communication and use. This especially includes any tools, methods, ecoprofiles and ecolabels that draw from the participating companies' data. For this reason, a regional hub of TSC is being established to provide appropriate regional adaptations and stakeholder engagement. The retailer and Chilean government are also funding targeted producer initiatives to catalyse the use of LCA-based tools and methods by retail suppliers.

It is the authors' view that the goal should include a wide consensus on a consistent level playing field methodology. Consistency specifically in terms of LCI and allocation but also LCIA, is crucial to enable a meaningful comparison between products. Solving methodological problems to encompass the breadth of issues for products sold in retail stores must provide an approach that is similarly consistent and applicable across a large range of sectors. This is also important in order to avoid the possible misinterpretation or even misuse of the results themselves.

5. Conclusion

The LCA reported here has shed light on what the main environmental hotspots in the life cycle are. A consistent and level playing field methodology and active engagement from Chilean stakeholders must be established for the food sector in order for LCA to provide meaningful and reliable input into decision-making.

6. References

Ahlmén, K., Andersson, K., Strömberg, K., & Widheden, A., 2001, LCA Kyckling, by CIT Ekologik and MiljöledarnaCiconia for SvenskFågel.

Australian Life-Cycle Inventory Database Initiative (AusLCI), 2009, Guidelines for Data Development for an Australian Life-Cycle Inventory Database, Draft Standard, 29 June 2009.

Barilla, GERF, & Life Cycle Engineering, 2009, Environmental Product Declaration of Durum Wheat Semolina Pasta Produced in Italy.

Bevilacqua et al., 2007, Life cycle assessment of pasta production in Italy, Journal of Food Quality, 30, 932-952.

Boggia, A., Paolotti, L., & Castellini, C., March 2010, Environmental impact evaluation of conventional, organic and organic-plus poultry production systems using life cycle assessment, World's Poultry Science Journal, 66.

Bosco, A., Di Bene, C., Galli, M., Remorini, D., Massai, R., & Bonari, E., 2011, Greenhouse gas emissions in the agricultural phase of wine production in the Maremma rural district in Tuscany, Italy, Italian Journal of Agronomy 2011, 6:e15.

Building Products Innovation Council (BPIC), 2010, Methodology Guidelines for the Materials and Building Products Life Cycle Inventory, http://www.bpic.asn.au/LCIMethodology.htm [last viewed 3 March 2011].

Cordella, M., Tugnoli, A., Spadoni, G., Santarelli, F., & Zangrado, T., 2008, LCA of an Italian lager beer, LCA Journal, Springer Verlag.

Franklin Associates, 2006, Life Cycle Inventory of Container Systems for Wine, prepared for Tetra Pak Inc.

Fundación Chile, 2007, Ficha Técnica Ecónomica Para el Cultivo de Alfalfa tipo, Dormancia Media.

Gazulla, C., Raugei, M., & Fullana-i-Palmer, P., 2009, Taking a life-cycle look at Crianza wine production in Spain: Where are the bottlenecks?, Int J LCA, 15 (4), 330-337

Goedkoop M.J., Heijungs R, Huijbregts M., De Schryver A.;Struijs J., Van Zelm R, 2009, ReCiPe 2008, A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level, First edition Report I: Characterisation, 6 January 2009, http://www.lcia-recipe.net [last viewed 13 January 2012].

Hospido, A., Moreira, M.T., & Feijoo, G., 2003, Simplified life cycle assessment of Galician milk production, International Dairy Journal, 13(10), 783-796.

Instituto de Investigaciones Agropecuarias (INIA) & Deuman Ltda, 2010, Huella de Carbono en Productos de Exportación Agropecuarios de Chile, Fundación Para La Innovación Agraria, Santiago, Chile.

International EPD Consortium, 2010, Carlsberg Italia, Environmental Product Declaration, Italia.

LCA Food, 2003, Poultry Farm Production, http://www.lcafood.dk/processes/agriculture/poultryfarms.htm [last viewed 1 December 2010].

ISO, 2006, ISO14044:2006 Environmental management – Life cycle assessment – Requirements and guidelines, Geneva, Switzerland.

Milà i Canals, L., 2003, Contributions to LCA Methodology for Agricultural Systems: Site-Dependency and Soil Degradation Impact Assessment, PhD thesis in Environmental Sciences, Autonomous University of Barcelona, Spain, http://tdx.cat/handle/10803/3155 [last viewed 8 July 2011].

Pelletier, N., 2008, Environmental performance in the US broiler poultry sector: Life cycle energy use and greenhouse gas, ozone depleting, acidifying and eutrophying emissions, Agricultural Systems, 98(2): 67-73.

Ritz, C.W., Fairchild, B.D., & Lacy, M.P., 2004, Implications of ammonia production and emissions from commercial poultry facilities: A review, J ApplPoult Res., 13, 684-692.

Ruini, L., & Marino, M., 2008, LCA of Semolina Dry Pasta Produced by Barilla.

Rural Industries Research and Development Corporation (RIRDC), 2011, Environmental Assessment of Chicken Meat Production Using Life Cycle Assessment, RIRDC Project No PRJ-004596, Draft Final report April 2011, Australian Government.

Talve, S., 2001, Life cycle assessment of a basic beer, LCA Journal, ecomed Journals, Germany.

USDA, Choose My Plate; http://www.choosemyplate.gov/food-groups/ (Accessed February 9, 2012).

Williams A.G., Audsley E., & Sandars D.L., 2006, Determining the Environmental Burdens and Resource Use in the Production of Agricultural and Horticultural Commodities, Main Report, Defra Research Project IS0205, Bedford, Cranfield University and Defra.

WRAP, 2007, The Life-Cycle Emissions of Wine Imported to the UK, http://www.wfa.org.au/entwine_website/files/resources/The_Life_Cycle_Emissions_of_Wine_Imported_to_the_UK_Final_Report.pdf [last viewed 15 November 2011].

Implementation by a large-scale retailer of a multi-criteria environmental labelling for food products: the Casino Environmental Index

Marion Sarteel[*], Frédéric Croison, Bérénice Harfouf-Ponthus, Olivier Jan[1], Eric Labouze

[1] BIO Intelligence Service, France
[*] Corresponding author. E-mail: *marion.sarteel@biois.com*

ABSTRACT

In 2011, BIO IS and Casino, with the participation of suppliers, developed and tested an environmental labelling methodology called "the Environmental Index", based on LCA and communicated through a single-score. GHG emissions, water consumption and eutrophication are quantified through LCA methodology, weighted and aggregated through PRIOR® method. The score represents the environmental impact of 100g of product compared to the environmental impact of the average daily food consumption of a French person. This score, similar to on-pack nutritional information, provides a benchmark on more than 160 products and helps consumers into more responsible purchasing habits. For manufacturers, the data collection is easily handled but time spent can be important. The methodology chosen is flexible for integration of future upgrades without having to change the format of the Environmental Index, to which consumers become familiarised. Nonetheless, the aggregation method may be updated, extended to the wider European context, and improved with biodiversity information.

Keywords: environmental labelling, experimentation, consumable food products, Environmental Index, Casino, aggregation, LCA

1. Introduction

Since 2006, BIO Intelligence Service has supported Casino, a major retailer, in its efforts to evaluate the environmental impact of its own-brand products in France. The "Carbon Index," which has been displayed on-pack on more than 600 products, shows greenhouse gas emissions emitted in the production of 100 g of the finished product (Casino, 2012). In 2011, BIO Intelligence Service and Casino contributed to environmental labelling experimentation in France and developed a multi-criteria environmental labelling methodology based on Life Cycle Assessments (LCA). First designed and tested with seven partners (suppliers and NGOs), the methodology and its result, a single-score called the "Environmental Index," is progressively being implemented on all Casino-branded food products.

The goal of the study is to make simplified LCA and environmental analysis simple and accessible for industries, manufacturers and retailers. With low implementation costs and good quality results, the study demonstrates the feasibility of a multi-criteria environmental labelling. The study also demonstrates that it is possible to communicate environmental information to consumers, thanks to a single indicator, which is visible and easily understood by customers.

The study stems from an active collaboration between retailers, food manufacturers and BIO IS as an environmental labelling consultancy. The partners are:

- Alter Eco
- BIO Intelligence Service
- Environmental NGO (confidential)
- Groupe Casino – EMC Distribution
- Fruité SAS
- Glon Sanders Holding
- MerAlliance
- Monoprix
- Saint Amand
- St Michel

A technical partnership was also organised with an environmental organisation which monitored every stage of the study's development.

2. Methods

2.1. General methodological framework

The quantification of the environmental impacts of food products relies on LCA methodology, recommendations of ADEME-AFNOR platform (2011) and the draft "Food product PCR".

The functional unit is: “100g or 100ml of consumable food product”. All the stages of the lifecycle recommended in BPX30-323 are accounted for in the methodology of this study:

- raw material production
- packaging production
- production and packaging stages
- transport of raw materials, packaging, finished products
- distribution of products
- flows related to transport and energy infrastructures
- use of the products on customer premises
- end of lifecycle of finished product and packaging

Three indicators were selected as inputs to the calculation of the label: GHG emissions (g CO2 eq), raw water consumption (L) and freshwater eutrophication (g PO_4^{---} eq). These indicators were selected according to criteria on relevance, methodological feasibility (quantification) and availability of Life Cycle Inventories, determined by the ADEME-AFNOR working group on food and feed products (“Working group 1”). Furthermore, despite several proposals to account for biodiversity criteria, impacts on biodiversity could not be accounted for in this study for feasibility, data quality and relevance reasons. Should the Environmental Index be widely deployed, an additional study will be needed to work on biodiversity indicators.

Responding to the BPX 30-323 requirements, suppliers provided primary data by filling out a questionnaire in Excel form. Two to six weeks are necessary for suppliers to collect all the information necessary to calculate the impact of the product.

Table 1. Data collected among suppliers

Life cycle step	Data collected
Production of ingredients	• Recipe: ingredients and quantity • Loss of raw materials
Production of packaging	• Materials and weight for primary, secondary and tertiary packaging per consumer sales unit • Number of consumer sales units per secondary and tertiary packaging
Transformation	• Total quantity of production in the factory over a year, for all products references • Total energy and water consumed in one year in the factory • Other products consumed (refrigerant leakage, etc.) • Quantity and composition of wastewater
Retail	• Conservation method during transport and retailing
Transport	• Transport of ingredient from the field to the supplier factory: distance, mode of transportation and conservation method • Transport of packaging from the field to the supplier factory: distance and mode of transportation • Transport of the finished product from the supplier factory to Casino warehouses: distance, mode of transportation, quantity of consumer sales unit delivered of each warehouse and conservation method
Use	• Conservation method and typical cooking practices (especially heating)
End of life	-

Secondary data are extracted mainly from the EcoInvent database, except for agriculture data. The latter are derived from scientific studies or calculated by BIO IS through a simplified LCA (without peer review). More than 30 LCAs of animal products, vegetables and fruits were conducted within this study. Data used for the LCAs are from public scientific literature, methodological ADEME-AFNOR guides, CORPEN (2006), Projet CASDAR (2009), federations of producers, technical institutes and sometimes producers.

2.2 Calculating the aggregate environmental label

Although not a common practice for LCAs (ISO 14044, 2006), data aggregation is the selected methodology in this case because the environmental label is intended for the general public. Display of environmental information based on multiple criteria can affect customers or even cancel out the potential benefits of an environmental labelling scheme for products by allowing consumers to select between different environmental impact categories. In order to facilitate consumer interpretation of the environmental performance of a product and help customers in their decision-making process, LCAs resulting in weighting-aggregation methods can be used.

The three LCA indicators are compared to the environmental impacts of the average French consumer and then aggregated into a single indicator, so that the environmental label can be readily understood by customers.

2.2.2 Normalisation of the indicators

To make it easily accessible for customers, each indicator is first normalised so that it displays the environmental impact of 100 g of product compared with the environmental impact of total daily food consumption of a French consumer.

The three reference indicators are calculated for total French food consumption. They are calculated per year and then brought down to a per day basis for one French person. The GHG emissions of the global food consumption, including cooking impacts as well as their production, transport, distribution and end of life cycle, are the results of a research project led by BIO IS on the environmental impact of French food consumption (Oudet, 2011). Global water consumption is based on the water footprint of households, at a macro-economical level. It takes into account direct and indirect water consumption (as irrigation, cooling of nuclear installations, etc.). Eutrophication is evaluated by the quantity of fertilisers (N, P) used in France for food products. (The sources for global water footprint and global eutrophication are confidential.)

The environmental label is meant to provide consumers with a level of information similar to the nutritional labelling on food products. The nutrition label which is calculated per 100g of product (or in some cases, per portion) provides information on the nutritional value, contributing to better informed food choices.

The analogy with the nutrition label is relevant: to help customers opt for healthy/sustainable foods.
This principle is mirrored for the Environmental Index as it aims at proposing clear and objective information, enabling customers to form a choice that considers the impacts of global daily food consumption. The information expressed in percentage form reflects an impact and gives the size of this impact. Customers can evaluate their purchases according to the latter and their desire to have a sustainable shopping approach.

2.2.1 Aggregation method chosen

Once normalised, the three indicators are combined using specific weights defined in the PRIOR® method (Labouze, 2006). The aggregation method was elaborated by BIO IS and recognised by the ADEME and the French Ministry of Environment (MEDDTL) but never officially published. It consists in normalising results (per habitant equivalent, but here with the criteria detailed above) and then applying weighting coefficients elaborated with a panel of experts considering the French context. The output of this operation is a score without dimension.

Six aggregation methods were studied and tested during the project: methods based on the opinion of an expert panel (PRIOR®, Eco-Indicator 99 and “third” method), monetisation methods (Damage cost and Eco-tax 2002) and End point method Indicators calculation (Recipe). Although each one has its strong and weak points, the PRIOR® method was chosen. This choice was based on the acceptable degree of subjectivity, the importance allocated to the national and/or international recognition of the method, implementation feasibility (availability of data especially for the three indicators) and customer understanding.

3. Results

3.1 An environmental label designed for customers

Currently, the Environmental Index is available for more than 160 product references. The environmental label is displayed on-pack, for use as a real decision factor for purchasing. According to the consortium, a single score display placed directly on product packaging is a relevant and effective way for an environmental criterion to be considered at the time of purchase. This transmission mode of information seems to be preferred by consumers as it has been acclaimed by 86% of them (Casino-IFOP, 2012). On-pack labelling would also appear to be the best transmission mode for each company to manage its information flows and be responsible for making this available.

Figure 1. Illustration of Environmental Index for grape juice

Since results are presented as a percentage of the daily impact of food consumption by a French customer, people can truly draw a comparison between their own individual impacts and that of the average consumer. Therefore, this indicator makes it possible for people to choose their items depending on their desire to purchase sustainably.

The leaves of the label, as shown below, range from dark green to orange, so that consumers can at a glance identify the category of impact of the given product. The percentage then provides more detailed information.

Figure 2. Display on front packaging

Extensive information is available on a dedicated website for customers interested in learning more about environmental labelling. On “www.indice-environnemental.fr”, all LCA information is made available (results for each indicator and per life-cycle stage) as well as advice on recycling best practices. A web link is displayed on packaging as well as a 2D barcode for Smartphone users, which is linked to the mobile version.

Table 2. Detailed results for 100 g of grape juice

Life cycle steps	GHG emissions (g eq. CO_2)	Water consumption (L)	Wate pollution (g eq. PO_4^{3-}
Production of ingredients	68	0.54	0.47
Production of packaging	88	0.17	0.02
Transformation	4.0	0.20	0.002
Retail	9.3	0.67	0.02
Transport	37	0.14	0.04
Use	4.3	0.25	0.01
End of life	6.5	0.10	0.03
TOTAL	137	2.1	0.59

Aggregation of multiple criteria provides customers with simple, easy-to-understand information. The label displayed on the product must be quickly understood so that customers can really use it as a criterion for shopping. It also can be used as a decision-making tool to prioritize and grade the environmental stakes both by eco-design manufacturers and policy-makers.

Consumers, especially those unaware of environmental issues, clearly ask for simplicity: 61% of them prefer a scale without figures and 48% prefer a unique score (Casino-IFOP, 2012).

The results of the survey led by Casino on customers' perception of the Environmental Index [8] indicate that the Environmental Index is not easy to understand at a first glance and requires a certain amount of effort on the part of customers. Nonetheless, the score seems to be clearer than LCA indicators as 57% of customers preferred the unique score versus 24% for the separated indicators. Moreover, for young people, 76% of the customers determined correctly between two products the one which has the lowest impact with the Environmental Index whereas only 21% determined correctly with separated indicators (21% do not know and 37% have a wrong answer). Finally, it is interesting to notice that even if the Environmental Index is not known yet, the old label, the Carbon Index, is currently known by 35% of the consumers.

3.2 High involvement by suppliers

The LCA methodology was adapted to better fit environmental labelling cost and efficiency constraints. Indeed, the methodology has been established with suppliers, who, as data collectors for LCA calculations, know the kind of data they can provide at an acceptable cost. Data collection is easily handled by manufacturers, especially for those who have already participated to the Carbon Index. Nonetheless, time spent on data collection seems to be important as several departments within a company and a supplier's own supplier can be involved in the process.

The implication of manufacturers in the calculation of the Environmental index is very important and positive as it can serve as a way for companies to manage their own information flows and to be responsible for the release of this information. Because lots of data are collected from suppliers, the calculation of impact will likely be highly representative of the actual product that is to be labelled. The initiative may thus be the first step for setting up an eco-design approach on manufactured products.

The evaluation of these points will be conducted with partners during the analysis of the project feedback in June 2012.

3.3 A flexible methodology for a unique format

The methodology chosen, specifically for aggregation, will allow an interesting degree of upgrading without having to change the format of the Environmental Index, to which consumers are becoming familiarised. It can be adapted to any geographic context by modulating the references used for the normalisation and to non-food products by choosing the relevant indicators. The addition or change of environmental criteria or any methodological change due to future ADEME-AFNOR recommendations can also be implemented without impacting the format of the Environmental Index.

4. Discussion

Reference data for the normalisation method may be homogenised and further specified. A public study that assesses the impact of food consumption in France for those three indicators in the same scope may be necessary.

The aggregation method was selected among five other methods. The method is robust, elaborated by BIO IS with a panel of experts and recognized by ADEME and the French Ministry of Environment (MEDDTL). Nonetheless, it may be subsequently updated, for example:

- Each indicator may be weighed in light of the social, cultural and scientific priorities of 2012 as the PRIOR® method was developed in 2006.
- Water consumption may be weighed, as there is no specific weighting coefficient for it, the coefficient of non renewable natural resources is currently used instead, pending further precision of the method.
- The global method may be extended to the broader European context as the current method is adapted to French context only. The method will also be improved with the inclusion of biodiversity information as soon as a consensual calculation methodology is available.
- Finally, support information around environmental labelling, indicators and the Environmental Index should be prepared to improve the level of understanding of the approach.

It took several years for customers to be familiar with nutrition labelling. Now they seem to be seeking this kind of information. The same trend will likely happen with environmental information.

5. Conclusion

The Environmental Index study enabled the testing of a methodology for environmental labelling that can be expressed with a single and simple score, with the participation of various actors of the agro-industry sector.

With the aim of providing clear and objective information, this study is intended to streamline environmental display information to help consumers make sustainable purchasing choices. Indeed this approach makes it possible for products from the same or different categories to be differentiated according to environmental performance. This new approach needs nonetheless pedagogy efforts from professionals involved in environmental labelling in order to increase the level of understanding of consumers. The methodology chosen is flexible for integrating future upgrades without having to change the format of the Environmental Index, to which consumers are becoming familiarised. This study may also be the first step toward setting up eco-design approaches on food products. In this way, it may efficiently promote the reduction of environmental impacts due to food consumption.

6. References

ADEME-AFNOR, 2011. BPX30-323: référentiel de bonnes pratiques sur l'affichage environnemental des produits de grande consummation.

Casino, 2012. L'indice carbone Casino. http://www.produits-casino.fr/developpement-durable/dd_indice-carbone-indice.html

Casino-IFOP, 2012. Perception par les consommateurs de l'environnement, l'affichage environnemental et l'Indice Environnemental. Enquête réalisée auprès de 969 clients du Groupe Casino, currently unpublished.

CORPEN, 2006. Guide des émissions d'ammoniac et gaz azotés à effet de serre en agriculture.

Labouze E., 2006. PRIOR®, un outil de priorisation des enjeux environnementaux - Guide pratique destiné aux acteurs de l'éco-conception.

ISO 14044, 2006. Environmental management — Life cycle assessment — Requirements and guidelines.

Oudet N., 2011. Environmental assessment of conventional vs organic food consumption in France.

Projet CASDAR, 2009. Guide méthodologique pour l'estimation des impacts des activités agricoles sur l'effet de serre.

Fateful and faithful: the success factors of eco-labelling

Flora D'Souza*, Alexander Liedke, Sabine Deimling

PE INTERNATIONAL AG, Leinfelden-Echterdingen, Germany
* Corresponding author. E-mail: *f.dsouza@pe-international.com*

ABSTRACT

Eco-labelling schemes are on the rise – customers and environmental experts alike are confused with the booming "industry". The difference between the two groups is that the customers' decision for or against a label is the *fateful* one: it decides the ultimate success or failure of the label. At the same time the judgment of an environmental expert – the one *faithful* to *some* aspect of sustainability – carries the responsibility of steering private and public opinion in the right direction. The titular word game is meant to illustrate the core of the challenge: different stakeholder groups have very different needs and influence regarding value and success of eco-labelling initiatives. But why label food at all? And more importantly, how? Is it sufficient to declare the quality in terms of taste and nutritional value or do we need to know about the sourcing of ingredients and use of pesticides, fertilisers? Does a full Life cycle Assessment (LCA) have added value compared to type I declarations? Or to put it in a single, practical question: what makes the TüV or CarbonTrust labels successful, the EU-Bio-label popular over many other labels that spring up then die off within years of the launch? These are the questions for which this semi-theoretical paper seeks answers to.

The authors set out to explore the success factors and discovered that stakeholder engagement is in fact the make-or-break factor. The diverging interests must be harmonised and the key to that lies in asking the right questions (Table 1). In addition to evaluating some good examples from the multitude of existing labels and active stakeholder groups, the present paper discusses a stepwise protocol on "how to create a successful label" using the case study of the EU-funded Life+ project "HAproWine" (EU - LIFE08 ENV/E/000143) aimed at developing an eco-label for wine produced in the Spanish region Castilla y León.

The steps can be summarised as follows:

(1) Prepare a small-scale pilot project
(2) Engage the stakeholder groups: survey the interests of representatives of consumers, retailers, producers, and governmental or other authorities who are interested in managing, verifying or promoting the aspect of sustainability targeted by the label
(3) Define the objective of the label: which information should it carry in order to satisfy the interests and needs of the stakeholder groups?
(4) Define the requirements of awarding the label (quality criteria, award system, review/verification system, validity and organisational structure)
(5) Involve stakeholders in milestone decisions as checkpoint: is the label in fact addressing their interests?
(6) Evaluate outcomes of the pilot project based on the percentage of labels awarded, the satisfaction level of the stakeholder groups involved, the practicability of the awarding scheme, the meaningfulness of the label
(7) Roll out label with help of stakeholders involved in pilot

Through the evaluation of some key existing examples and the case study of developing a new label in the food industry based on LCA, the paper identifies underlying factors of success and failure.

Table 1. Eco-label success factors from the perspective of different stakeholder groups.

Stakeholder group	Highest priority questions	Solutions
Producer	Does the label help me sell my product?	Label may be required to sell in certain retailers, or it could enable cost reduction or higher pricing
Retailer	Does the label ensure that I can meet my sustainability targets? Can I document achievements?	Label may enable reducing the retailer's Scope III emissions and can be documented quantitatively as well qualitatively in different reporting schemes (CSR, CDP, GRI...)
Consumer	What does the label tell me and how can I find the one most attractive food product for me?	Labels must convey simple information, be it qualitative (<5% mineral fertiliser allowed) or quantitative (50g CO2e) and provide a benchmark; eco-labels should not *significantly*[a] decrease product quality or increase product cost; Labels shall be issued by some *authority*[a]
(Non-) Governmental Authority	Are the criteria for the label easy to verify and practicable for national/global targets?	Label schemes must require straightforward calculation rules and evaluation schemes; data collection rules must also be specified and verifiable; Labelled products should facilitate meeting national or regional targets (fair trade agreements, CO_2 cap agreements etc.)

[a] Words in *italics* are subjective reference points and shall be further specified

References

Díez Vázquez, J. À. et al., 2010-2012. HAproWINE (EU - LIFE08 ENV/E/000143) Integrated waste management and life cycle assessment in the wine industry. http://www.haprowine.eu/

Kwasniewski, N. 2012. "Öko-Test" und Co. Welche Lebensmittelsiegel wirklich taugen. Spiegel Online available online at http://www.spiegel.de/wirtschaft/service/0,1518,813556,00.html

J Sainsbury. 2011. Corporate Social Responsibility Report. http://www.j-sainsbury.co.uk/

Tesco. 2010. Corporate Responsibility Report. http://www.tilles.co.uk/companies/tesco/reports.html

Marks & Spencer. 2006. Corporate Social Responsibility Report. http://corporate.marksandspencer.com/

Carbon footprinting and labelling of agri-food products: practical issues for the development of Product Category Rules

Rattanawan Mungkung[1,*], Shabbir H. Gheewala[2,3]

[1] Centre of Excellence on enVironmental strategy for GREEN business (VGREEN) and Department of Environmental Science, Faculty of Science, Kasetsart University, Bangkok, Thailand

[2] The Joint Graduate School for Energy and Environment (JGSEE), King Mongkut's University of Technology Thonburi, Bangkok, Thailand

[3] Center for Energy Technology and Environment, Ministry of Education, Thailand

[*] Corresponding author. E-mail: *fscirwm@ku.ac.th*

ABSTRACT

Carbon Footprint has emerged as an approach to estimate greenhouse gas (GHG) emissions throughout the product's life cycle. The carbon footprint information is also encouraged to be provided to consumers via Carbon Label as an indicator of the climate-friendliness to take into account in their purchasing decisions. The carbon footprint and label strategy is expected to stimulate a market demand for lower carbon products to move towards the reduction of GHG emissions at both production and consumption. Agri-food is identified as one of the main industrial sectors contributing significantly to GHG emissions. Thus, they are targeted and given priority in several countries, including Thailand. Implementing carbon footprinting to put carbon labels is intended to raise awareness of consumers on the carbon footprint attached to their purchasing choices for climate change mitigation. However, it is most critical that the carbon footprinting methodology is standardized internationally to be able to compare the carbon footprint values between products. It must be emphasized that though carbon footprints are not intended in principle for comparison between "apples and oranges"; in practice such comparison by the consumers is difficult to control. Based on a number of carbon footprinting case studies conducted in Thailand in 2008-2011 (e.g. rice, tapioca, sweet corn, baby corn, asparagus, pineapple, chicken, shrimp, tilapia, etc.), some major practical issues were raised when developing the Product Category Rules (PCR). These issues are discussed in the paper along with the solutions that have been arrived at in Thailand for the first few PCRs that have been developed to date. Issues of system boundaries, product grouping, allocation and data quality are elaborated along with suggestions even for the type of carbon label based on stakeholder consultation.

Keywords: agri-food products, carbon footprinting, Environmental Product Declaration, Product Category Rules

1. Introduction

1.1. Introduction to carbon footprint

"Carbon Footprint" has emerged as a tool to assess greenhouse gas (GHG) emissions throughout the product's life cycle, aiming to identify the hot spots and stimulate emission reduction. The carbon footprint information is also encouraged to be provided to consumers via Carbon Label as an indicator of the climate-friendliness to take into account in their purchasing decisions. The carbon footprint and label strategy is expected to stimulate a market demand for lower carbon products to move towards the reduction of GHG emissions at both production and consumption. The concept of carbon footprint and carbon label is well accepted, leading to the development of ISO 14067 which is to be officially launched in 2013. Having recognised the significance of global warming and climate change impacts, several countries including Thailand have adopted carbon footprinting and labelling schemes. Agri-food is identified as one of the main industrial sectors contributing significantly to GHG emissions.

1.2. Carbon footprint and label in Thailand

Thailand is very well aware of the development and implementation of Carbon Footprinting and Carbon Labelling for climate change mitigation as well as to anticipate trade measures. Food has been the focus on this particular issue, due mainly to its contribution in GHG emissions at the household level. Also it is the main product for all consumers who should be convinced to change their consumption behaviour towards lower emissions. Initiated in 2008, carbon footprint projects on chicken (Chicken snack and Steamed diced oven chicken), tuna (Canned tuna in sunflower oil) and rice (Jasmine rice and Rice vermicelli/noodle) were conducted by Kasetsart University (KU) and the Joint Graduate School for Energy and Environment (JGSEE) (Mungkung et al., 2012). Both projects were aimed to familiarise with the PAS 2050: 2008 methodology which was available at that point of time and identify the local knowledge gaps.

In 2009, a national pilot project to promote the implementation of product carbon footprint was initiated by the Thailand Greenhouse Gas Management Organisation (Public Organisation) (TGO) and National Metal and Materials Technology Centre (MTEC), Thailand. The project was targeted at the implementation of product carbon footprinting and labelling leading to development the national guidelines on product carbon footprint as well as the national carbon footprint labelling scheme which was launched officially in December, 2009. Interestingly, food companies such as chicken, tuna, pineapple, rice, pet food, and animal feed joined the project because of their awareness on global warming issues and also because they would like to prepare themselves for reporting, reducing GHG emissions, and labelling the carbon footprint information. More studies were conducted in 2011 by KU and JGSEE, covering shrimp, asparagus, baby corn, sweet corn, sweet chili sauce, and coconut milk including the development of "FOODprint" as a tool to facilitate the industry collecting the data and calculating the carbon footprint value based on the collaboration from 40 food companies: fruits and vegetables (i.e. rice, tapioca starch, pineapple, fruit cocktail, etc.), livestock (i.e. chicken, pork, milk, etc.), fisheries (i.e. tuna, mackerel, etc.), aquaculture (i.e. shrimp, tilapia, etc.), food ingredients (i.e. sugar, palm oil, etc.), drinks (i.e. coffee, aloe vera juice, etc.) and ready-to-eat meals (i.e. cereal, instant fried rice, etc.) (Gheewala and Mungkung, 2012).

The national guideline of product carbon footprint (Available at http://www.tgo.or.th/english/index.php?option=com_content&view=article&id=194:the-national-guideline-carbon-footprinting-of-product&catid=51:publication&Itemid=68, in English) was developed from the practical experiences gained from 24 pilot companies, which included: food, packaging, textile, electrical and electronic equipment, automobile, and building materials. The key references were: ISO 14040/44, PAS 2050:2008 (Publicly Available Specification: Specification for the assessment of life cycle greenhouse gas emissions of goods and services) and TS Q0010 (Japanese Technical Specification: General principles for the assessment of carbon footprint of product). The principles of carbon footprint assessment are rooted in the life cycle approach and the main methodological issues are applicable for all kinds of products. The carbon footprint labelling is based on the methodology described in the national guideline on product carbon footprint and expressed as carbon score, XXX g or kg CO_2e (three significant digits). To date (March 2012), there are more than 480 products being labelled from 117 companies; food products are more than half of the total products [Available at http://thaicarbonlabel.tgo.or.th/carbonfootprint/index.php?page=2].

1.3. Development of PCRs for agri-food products

Different characteristics of products have been recognised thus leading to the need for development of "Product Category Rules" (PCRs) as additional requirements for carbon footprint implementation. The practices of PCR development in Thailand are at two levels: (i) by the first company who calculated the carbon footprint and applied for verification for the carbon footprint label approved by the national technical committee on product carbon footprint (PCR, company level), (ii) by TGO with the stakeholder consultation meeting and approved by the national technical committee on product carbon footprint (PCR, national level). Until now, there are 74 PCRs, at the company level (40 of these are concerning food products) and 3 PCRs at the national level (textiles divided into yarn and fabric, clothing, non-clothing; food which are milled rice and chicken meat). It was discussed if PCRs should be developed for each sector or specific product; this had led to the analysis of the related issues of PCR development and contents in this study.

2. Methods

To identify the issues related to the PCR development and contents for agri-food products, the practical experiences in applying both PAS 2050 and implementing the national guideline on product carbon footprint were gathered. The analysis is then used to provide a set of recommendations for further development of PCRs in agri-food products.

3. Results and Discussion

The results from the analysis of issues that should be taken into account and proposed solutions are discussed in the sections below.

3.1. Approach in developing PCRs

There are two approaches in developing PCRs: sector-based or individual product-based. We started from the development of PCR based on each specific product (individual product), which was drafted and proposed by the pioneering companies conducting the carbon footprinting and applying for the carbon footprint label. However, the local experts were concerned that the development of PCR at individual level would be time-consuming and will not be able to respond to the business operations and decisions. On the other hand, some experts felt that some key products should be developed individually as there are some very specific issues such as rice from different rice farming systems in each region having different emission factors. It should be noted here that some products do not yet have many companies implementing the carbon footprint label so the stakeholder consultation process could not take place. In addition, the companies applying carbon footprint based on the PCRs previously developed by the pioneering companies do not have any disagreement issues yet, so these can still be used for a while.

3.2. Contents of PCRs

3.2.1. Scope of PCRs

At the beginning, rice and chicken were selected for developing the national PCRs to test the approach used in individual product-based PCRs. Rice was of special interest as it is a main product for Thailand and the data sources of emission factors from rice fields affect the results significantly. Chicken was another product selected as it is mainly exported to EU countries which are active on carbon footprinting and labelling. Moreover, the manufacturers of these two products have been proactive in adopting carbon footprinting. Later on, the scope of PCRs of agri-food products was discussed and it was found that each sector has its own characteristics and should not be considered at the sub-sectorial rather than sectorial level such as Fruit & Vegetables, Livestock, and Aquatic products. Within each sub-sector, both non-processed and processed products should be considered as the processed products would require the raw materials such as fresh fruits and vegetables so it is best to deal with the PCR issues together. It should be noted here that it is generally agreed that processed food products with multiple ingredients (i.e. vegetable, meat, coconut milk, etc.) should be considered in a separate PCR but can refer to the related parts in PCRs of Fruit & Vegetables, Livestock, and Aquatic products as required. Meat, milk and egg are considered livestock products, whereas products from fisheries and aquaculture (from fresh, brackish and sea water) activities are reflected in aquatic products.

3.2.2 System boundary

It is generally accepted that the system boundary should cover all life cycle stages including at the point of sale (e.g. chilled storage), pre-consumption (e.g. household storage), consumption (e.g. cooking), and post-consumption (e.g. food and packaging waste disposal). This is not so for the case of food ingredients as they are only a composition of other food products in the forms of dish or meal (Mungkung et al., 2010a). It is also important that the assumptions for point of sale, pre-consumption, consumption and post-consumption are on the same basis for the same food products. The assumption at point of sale is based on the turn-over rate in the real situation. The storage prior to cooking or pre-consumption is defined based on a standard scenario. As of now, the use profile is defined according to the instructions given by producer on the labels as there is no such data available in ISO 14025 as well as the Thai Green Labelling scheme (Ecolabel Type 1). A standard scenario is also applied for the final waste disposal (assuming there is no food waste, and packaging waste is totally landfilled). It is worth mentioning here that emissions from the office, prototype, research and development, quality assurance and control, consumer's travelling are not included.

In more details, these activities should be included: land preparation (prior the crop production), seed preparation (such as soaking in water, acclimatisation in pond, etc.), chemicals (such as fertiliser, hormones, herbicides, veterinary drugs, etc.), feed, substrate materials, soil covering materials, packaging and consumables for maintenance of capital goods. The activities that should be excluded are: infrastructure, equipment, and machines. Direct land use change is included, but at the beginning only for crop production (not all land use changes) and only 20 years back to the date of implementing the carbon footprinting. Though it is understood that the effects of land use change on GHG emissions is enormous, the industry still feels that developing countries will have a major problem due to the lack of supporting data.

3.2.3. Product description

The information of product name, brand name, and quality ranking according to the national standard defined in the Thailand Agriculture Standard (TAS) documents is required in describing the product. All auxiliaries included in the finished product being sold must also be considered. A practical issue raised by the industry is related to the number of agri-food products which can be classified into some thousands at the SKU (Stock-Keeping Unit) level. The industry echoed that the associated costs (for consultancy service, verification and license) could be a barrier especially for SMEs (Small and Medium Enterprises). In this connection, to promote the adoption of carbon footprint label, product grouping is allowed for the sake of cost reduction for the license (individual carbon footprint value is still applied for each product across the whole product group), as follows:

Table 1. Example of product groups

Feature	Non-processed food	Processed food
Same product, different tastes	- Sterilised full-cream milk (chocolate, pineapple, lychee, lemon taste)	- Chicken snack in 110, 250, 500 grams
Same product, different sizes	- Canned sliced pineapple in 8, 16, 20 ounces' pack	- Instant noodles in 55, 85, 210 grams' pack
Same product, with at least 90% of the total weight being the same and the rest of the ingredients will not cause a difference in the total carbon footprint value of more than 5%		- Instant noodles in different flavours (tom yum kung, minced pork, roasted duck, etc.)

3.2.4. Functional unit

It is generally agreed at the conceptual level that the unit of analysis should not be the sold unit but reflecting the functional unit of product. The discussion on the function of food has not yet been settled; there is not yet a common consensus on which unit would be the best choice to facilitate the product comparison. The nutrition level received a wide debate, especially the edible protein from different types of meat which could favour one product over another. For instance, the carbon footprint of tilapia will be much lower than other meat sources if the functional unit is based on the content of edible protein (Table 2). More importantly, whereas the whole fish is eaten in Asian countries, in Europe, people prefer having only the fillets; the functional unit based on whole fish and fillet cannot be compared in this case. For livestock, the proposed unit is the standard unit in terms of volume or mass, e.g. meat in 1 kg, milk in 1 litre and egg is per egg according to their size and weight based on the national standard.

Table 2. Comparison of different meats with tilapia, at the farm gate, based on different functional unit

Aquatic products (Reference)	Carbon footprint value (kg CO_2e/250 g of live weight)	Carbon footprint value (kg CO_2e/100 g of protein)
Thai tilapia (Tessa and Mungkung, 2011)	0.55	0.92
French trout (Papatryphon et al., 2003)	1.13	2.20
Canadian salmon (Pelletier et al., 2009)	0.82	1.65
Norwegian salmon (Ellingsen et al., 2009)	0.75	1.51

3.2.5. Data collection

The focus of companies was around the issues of data collection, such as the primary and secondary data requirements (i.e. which ones should be collected directly and which ones sourced from literature). The difficulties in data collection in the field vary due to different levels of data recording systems in place. The companies with proper data recording systems see this as an advantage; while those with poor data recording systems see this as the way out to easily substitute with the secondary data. The quality of primary and secondary data was discussed at length, as the industry is concerned about this. It can happen that the secondary data has better quality than primary data or vice versa. This is particularly of concern among Thai food companies as the food database is being developed and it is in a very early stage of development; as a result, several substitute data are used when there is no related database; this could be a disadvantage as compared to the countries where the databases are well developed.

Table 3. Stepwise sampling rule

Data type	Agri-food products		
	Fruit & Vegetables	**Livestock**	**Aquatic**
Primary data (if it is the direct activity of implementing company)	(1) Number of farms covering at least 50% of the raw materials used for annual production; (2) If the number of farms is huge and (1) is not practicable, then sampling based on square root of the total number of farms can be used (with the justification after analysing the variation among farms);		
Secondary data (if it is the indirect activity of implementing company)	(1) Identify the sources of raw materials covering at least 50% of the raw materials used for annual production; (2) Apply the inputs and outputs based on a typical practice in a particular area at the provincial level) including yield suggested by the Department of Agriculture; an adjusted yield (the average yield from previous 3 years) should be used if the yield is affected by unusual seasonal factors at the time of implementing the carbon footprint	(1) Obtain from the national databases, which were sourced from the literature (Thailand context)	

The duration of data collection is generally acceptable to be based on annual production for fruits and vegetables, or over the life span of animal producing milk or eggs (but if they are predisposed before reaching its life span then the data collection should be finished by then). But if the yield is fluctuating more than 10%, then it is reasonable to use the data over 3 years of production to capture the variation to achieve good representative data.

3.2.6. Allocation method

For the allocation of carbon footprint between co-products, physical causality as recommended by ISO14046:2006 was found to be the most appropriate. Thus, as shown in Table 4, for most cases, allocation by mass has been selected. Only for certain cases of by-products which might almost have been considered waste but have been defined now as co-products as some economic value could be obtained by selling them, economic allocation is proposed.

Table 4. Example of discussed allocation methods

Agri-food product	Level	Product & Co-products	Allocation method
Fruit & vegetable	Farming	Different quality levels/grades	Mass
	Processing	Body, core and shells	Economic (if used as the raw material for another product system)
Livestock	Farming	Different sizes of chicken	Mass (whole body)
	Slaughtering	Different portions or parts of chicken meat	Mass (with bone)
		Body and offal-skeleton-blood-feathers	Economic
	Processing	Used cooking oil (used for biodiesel)	Carbon footprint value = 0
Aquatic	Farming	Different sizes of fish	Mass (whole body)
	Processing	Fillet and bone-skin	Economic
		Body and head-shell	Economic

3.2.7. Materiality and cut-off rules

Materiality is defined based on a value of 1% of the total carbon footprint. Items contributing lesser than 1% of the total carbon footprint can be cut off, but the cut-off must not be higher than 5% of the total carbon footprint. This is consistent with other national standards such as PAS 2050 (UK) and TS Q0010 (Japan).

3.2.8. Display of carbon footprint label

The companies proposed an idea to have a logo of the carbon footprint label without displaying a figure, but having a commitment to reduce the value when extending the license. This implies that the verifier and the authority body must ensure that the carbon footprint calculation is done according to the national guideline and the carbon footprint value reduces from the previous time.

3.2.9. Period of license

It is generally agreed to have the carbon footprint label valid for 2 years. As a consequence, it has been decided to update the databases once every two years accordingly.

4. Outlook

It is agreed in principle that PCRs are necessary for defining the additional requirements for specific products or group of products to facilitate the product comparison. However, it should be discussed at the international level whether these should be sector-based or individual product-based. There is a great concern over the standardisation of PCRs at global scale as some countries might have developed their own PCRs, for instance, the UK (i.e. Supplementary requirement of horticultural products), Japan (PCR of vegetables and fruits) and Thailand (PCR of Fruit & Vegetables, PCR of Livestock products and PCR of Aquatic products) in their own systems which could be a barrier in comparing between products from two countries. It is foreseen that some common principles can be in agreement while some issues will still need more discussion for standardisation. Limitations of available food databases or related literature particularly in developing countries including Thailand will become one of the main difficulties. Apart from that, it is noticeable that various schemes: carbon score, carbon emission rate, low carbon emission level, or carbon neutral have been applied; this can be confusing to consumers regarding the conveyed message and understanding on the contributions to GHG reduction by each scheme. Carbon footprinting and labelling should be appreciated by both manufacturers and consumers, thus the ease of adopting the methodologies and the simple message to deliver via carbon footprint labelling should be the core elements.

5. References

Ellingsen, H., Olaussen, J.O., Utne, I.B., 2009. Environmental analysis of the Norwegian fishery and aquaculture industry – A preliminary study focusing on farmed salmon. Mar Policy. 33, 479–488.

Gheewala, S.H., Mungkung, R., 2012. Food and Climate Change: FOODprint as a tool to support GHG reduction in the Thai agri-food sector. Poster presented at the 8th International Conference on LCA in the Agri-Food Sector, Rennes, France, 2-4 October 2012.

Mungkung, R., Gheewala, S.H., Kanyarushoki, C., Hospido, A., van der Werf, H., Poovarodom, N., Bonnet, S., Aubin, J., Moreira, M.T., Feijoo, G. 2010. Product carbon footprinting in Thailand: A step towards sustainable consumption and production?, Environmental Development 3, 100-108.

Mungkung, R., Gheewala, S., Poovarodom, N., Towprayoon, S. 2010a. Carbon Footprint and Label for Rice Products [in Thai]. S.P.M. Printing Co., Ltd., Bangkok, Thailand. 83 p.

Mungkung R., Gheewala, S., Bonnet, S., Hospido, A., Kanyarushoki, C., van der Werf, H. and Poovarodom, N. 2010b. Carbon Footprinting and Labelling: Experiences from the Thai food industry Available online: http://www.thaicarbonlabel.sci.ku.ac.th/

Papatryphon, E., Petit, J., Van der Werf, H.M.G., Kaushik, S.J., 2003. Life cycle assessment of trout farming in France: A farm level approach. Proceedings from the 4th International Conference, October 6-8, 2003, Bygholm, Denmark.

Pelletier, N., Tyedmers, P., Sonesson, U., Scholz, A., Ziegler, F., Flysjo, A., Kruse, S., Cancino, B., Silverman, H., 2009. Not All Salmon Are Created Equal: Life Cycle Assessment (LCA) of Global Salmon Farming Systems. Environ Sci Technol. 43, 8730-8736.

Tessa, J., Mungkung, R., 2011. Carbon footprinting of tilapia from monoculture in cage and polyculture in pond. Proceedings 5th SWU conference, 17-18 March 2011, Bangkok, Thailand.

Characterising pesticide residues and related health impacts in LCIA

Ronnie Juraske[1], Peter Fantke[2], Assumpció Antón[3], Eva Sevigné Itoiz [3], Olivier Jolliet[4,5]

[1]ETH Zurich, Schafmattstrasse 6, CH-8093 Zurich, Switzerland
[2]University of Stuttgart, Hessbrühlstrasse 49a, 70565 Stuttgart, Germany
[3]Institute of Agriculture and Food Research and Technology, 08348 Cabrils, Barcelona, Spain
[4]University of Michigan, 109 S. Observatory, Ann Arbor, Michigan 48109-2029, United States
[5]Quantis, EPFL Science Park (PSE-D), CH-1015 Lausanne, Switzerland
[*] Corresponding author. E-mail: *juraske@ifu.baug.ethz.ch*

ABSTRACT

Multiple pathways contribute to the exposure of the general population to pesticides, such as inhalation and ingestion intake of applied fractions undergoing wind-drift, runoff, and leaching, but the most important pathway being consumption of fractions directly reaching the treated food crops. However, health impacts of pesticides from food consumption are still poorly represented in existing Life Cycle Impact Assessment (LCIA) approaches. In addition, pesticide uptake and translocation mechanisms vary considerably between crop species and may demonstrate significant differences in related health impacts as discussed in Fantke et al., (2011a). Therefore, assessing pesticide residues in multiple crops plays an important role in the evaluation of current agricultural practice. In light of this, a new dynamic plant uptake model – dynamiCROP – was designed to assess the dynamics of pesticide residues in different crops and to characterise the related human intake of these residues via consumption of harvested crop components.

The model, which is fully described in Fantke et al., (2011b), is based on a flexible set of interconnected compartments (Fig. 1) and is customised to wheat, paddy rice, tomato, apple, lettuce, and potato, thereby accounting for the major mass fraction of worldwide human plant-based diet. Modelled residues are evaluated against residues measured in experimental studies, such as Itoiz et al., 2012 for lettuce. Furthermore, the functioning of the underlying dynamical system was analysed to estimate the model input uncertainty and to parameterise the complex system for use in spatial or nested multimedia assessment models currently applied in LCIA (Fantke et al., 2012). The parameterised crop-specific models are adequate to assess pesticide residues in crops and enable the user to calculate these residues by providing only a very small set of input data.

Finally, human intake fractions (Fig. 2) are connected to effect information for characterising human health impacts. When combined with substance-specific pesticide application statistics, absolute impacts per considered land use area can be estimated and compared to other LCA endpoints. Human intake fractions, effect and characterisation factors (CFs) are provided for use in LCIA for 726 substance-crop combinations. CFs were calculated for 121 pesticides applied to the six crops and were 1 to 5 orders of magnitude higher than factors estimated from fractions lost via wind-drift, runoff and leaching. Human health impacts vary up to 9 orders of magnitude between crops and 10 orders between pesticides. Main aspects influencing the fate behaviour of pesticides were identified as half-life in plants and on plant surfaces, residence time in soil as well as time between pesticide application and harvest.

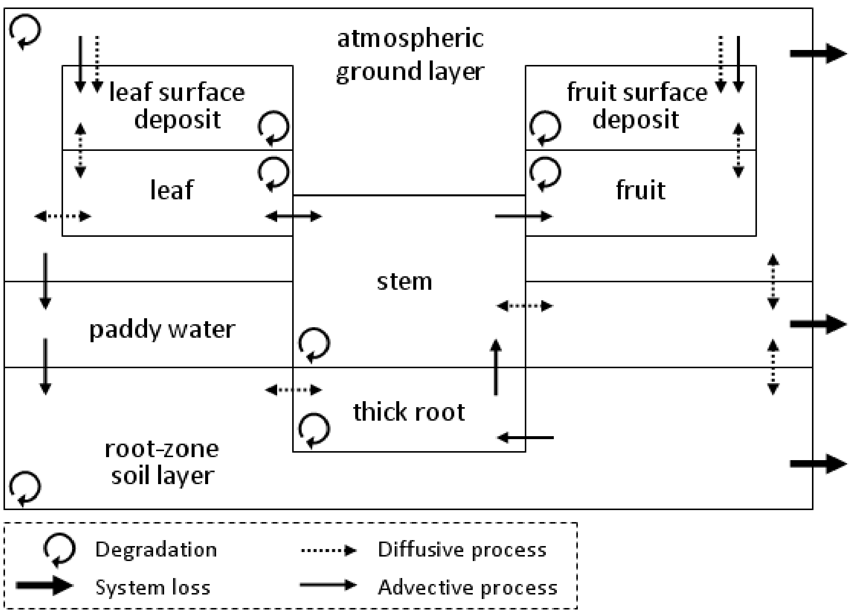


Figure 1. Graphical representation of model setup consisting of environmental compartments and processes within/between compartments.

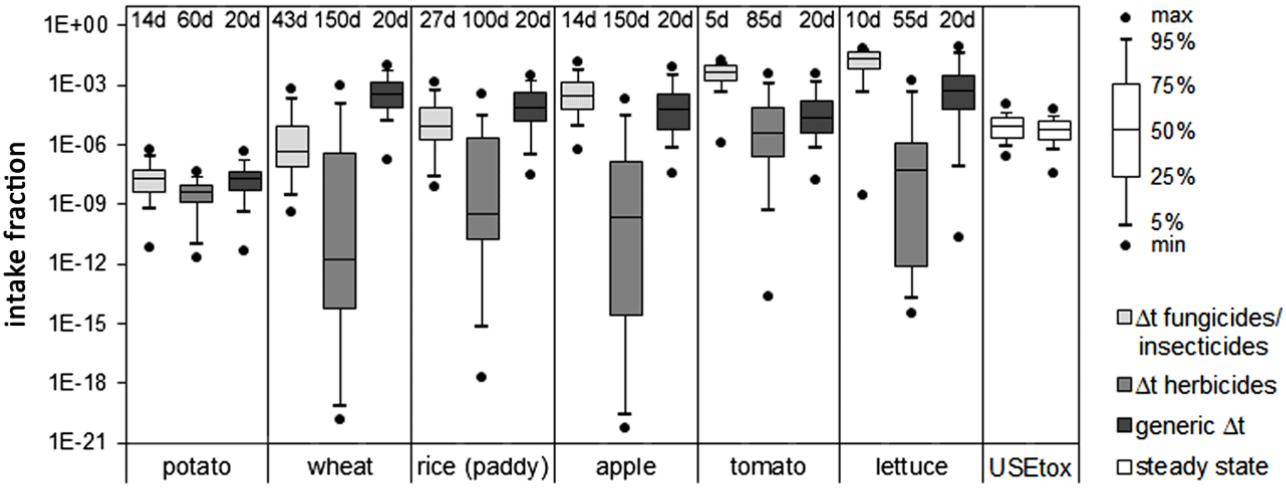


Figure 2. Intake fractions from consumption of pesticide residues for 121 substances applied to 6 food crops at different times to harvest Δt and from fractions undergoing wind-drift, runoff and leaching under steady state conditions as calculated with USEtox (www.usetox.org).

References

Fantke P, Charles R, de Alencastro LF, Friedrich R, Jolliet O. 2011b. Plant uptake of pesticides and human health: Dynamic modeling of residues in wheat and ingestion intake. Chemosphere 85: 1639-1647.

Fantke P, Juraske R, Antón A, Friedrich R, Jolliet O. 2011a. Dynamic multicrop model to characterise impacts of pesticides in food. Environ. Sci. Technol. 45: 8842-8849.

Fantke, P., Wieland, P., Wannaz, C., Friedrich, R., Jolliet, O., 2012. Dynamics of pesticide uptake into plants: From system functioning to parsimonious modeling. Environ. Modell. Softw. (submitted)

Itoiz, E.S., Fantke, P., Juraske, R., Kounina, A., Vallejo, A.A., 2012. Deposition and residues of azoxystrobin and imidacloprid on greenhouse lettuce with implications for human consumption. Chemosphere. (submitted)

Bridging the gap between LCI and LCIA for toxicological assessments of pesticides used in crop production: application to banana growing

Rosalie van Zelm[1], Pyrene Larrey-Lassalle[2,*], Philippe Roux[2]

[1] Department of Environmental Science, Institute for Water and Wetland Research, Radboud University Nijmegen, P.O. Box 9010, 6500 GL Nijmegen, The Netherlands
[2] Cemagref/Irstea, Research Unit: Information & Technologies for Agro-processes, 361 rue JF Breton, 34196 Montpellier, France
[*] Corresponding author. E-mail: *p.larrey-lassalle@irstea.fr*

ABSTRACT

In Life Cycle Assessment (LCA), the Life Cycle Inventory (LCI) provides emission data for the various environmental compartments and subsequently Life Cycle Impact Assessment (LCIA) determines the final distribution, fate and effects of substances such as pesticides. Given the overlap between the Technosphere (the studied anthropogenic system) and the Ecosphere (the natural environment) in agricultural case studies, it is, difficult to establish what LCI needs to capture with respect to degradation and partitioning of the pesticides in air, water and soil at the local scale. While evaluating the partitioning of the emitted substances, LCA practitioners must keep in mind that human toxicity and ecotoxicity models used in LCIA also include inter compartment transfers, fate, and degradation mechanisms at larger temporal and spatial scales such as long-range transmission of air pollutants at regional, continental, and global scale (Fig. 1).

Up to now, LCA practitioners have been using several hypotheses to build agricultural inventories. For example, the application of a regional or global scale model of substance transfers in the LCI phase (inducing an overlapping with LCIA models), or the application of a simplified approach assuming that pesticide emissions are entirely emitted to the soil compartment, or 85% is emitted to soil, 5% to crops and 10% to air (Audsley et al., 1997; Margni et al., 2002) are commonly used. To date, no clear distinction nor guidance are provided on how to combine LCI and LCIA models with respect to toxicological assessments of pesticides applied in agriculture.

This paper aims to provide guidance to better define the boundaries in space and time between what should be included in LCI and where LCIA takes over. A literature review was undertaken on available methods and models for both LCI (e.g. Birkved and Hauschild, 2006) and LCIA (e.g. Rosenbaum et al., 2008) with a special focus on toxicological assessments of pesticides used in crop production. The relevant biophysical phenomena are identified (Fig. 2) and guidelines are proposed to overcome the gaps between LCI and LCIA as well as to harmonise further comparisons of agricultural LCA results.

To complement these recommendations a case study on bananas is presented to i) characterise the current gap between LCI and LCIA, and ii) demonstrate the application of the proposed solution to the current LCA approach. From this case study, it is clear that impact assessment results for both human health and ecosystems are strongly influenced by the LCI hypotheses. The LCI hypotheses involving either inadequate model scales, or a too simplistic LCI, or non-equilibrated balances lead to an underestimation of up to a factor of 5.

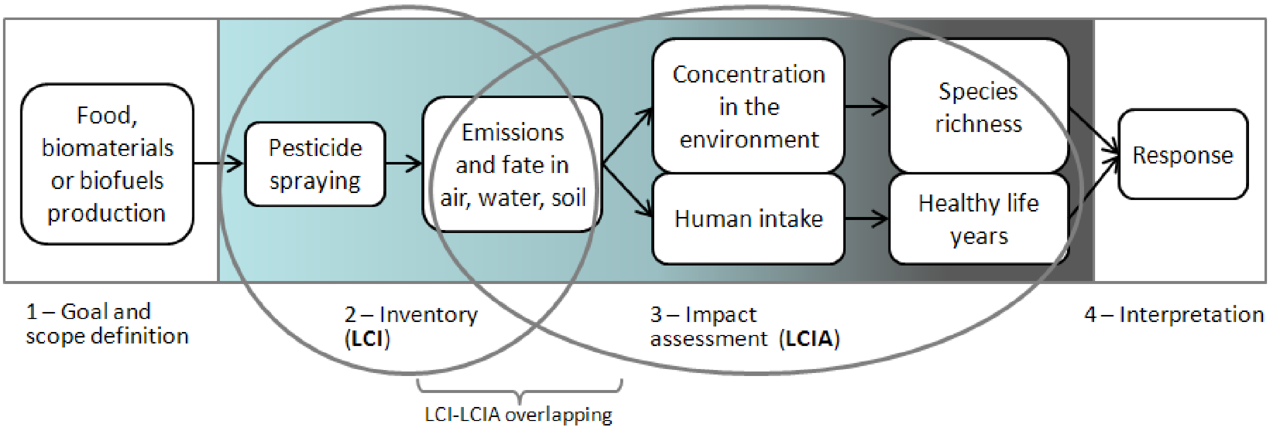


Figure 1. Overview of the 4 LCA phases of pesticides used in agricultural crop production highlighting an overlapping between LCI & LCIA.

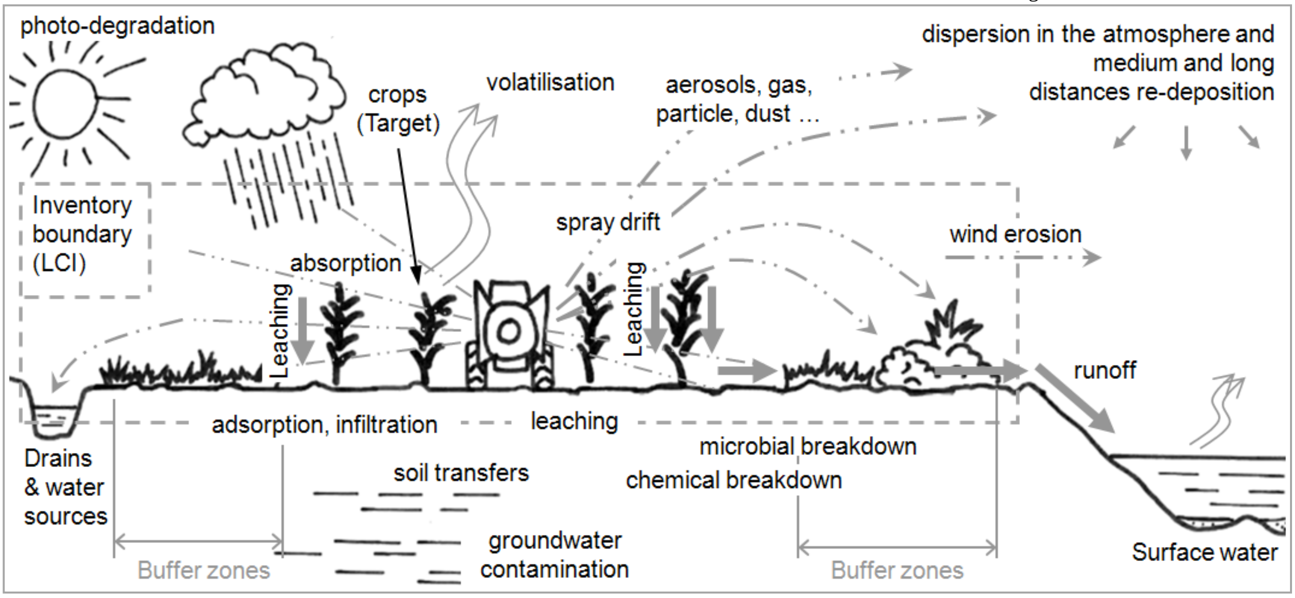


Figure 2. Proposed system boundary for LCI and associated mechanisms from pesticide spraying to emissions in air, soil and water.

References

Audsley, E., Alber, S., Clift, R., Cowell, S., Crettaz, P., Gaillard, G., Hausheer, J., Jolliet, O., Kleijn, R., Mortensen, B., Pearce, D., Roger, E., Teulon, H., Weidema, B., Van Zeijts, H., 1997. Harmonisation of environmental life cycle assessment for agriculture. Final Report Concerted Action AIR3-CT94-2028, European Commission DG VI Agriculture

Birkved, M., Hauschild, M.Z., 2006. PestLCI - A model for estimating field emissions of pesticides in agricultural LCA. Ecol. Model. 198, 433-451.

Margni, M., Rossier, D., Crettaz, P., Jolliet, O., 2002. Life cycle impact assessment of pesticides on human health and ecosystems. Agr. Ecosyst. Environ. 93, 379-392.

Rosenbaum, R.K., Bachmann, T.M., Swirsky Gold, L., Huijbregts M.A.J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H.F., Macleod, M., Margni, M., McKone, T.E., Payet, J., Schuhmacher, M., Van de Meent, D., Hauschild, M.Z., 2008. USEtox—The UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in Life Cycle Impact Assessment. Int. J. LCA 13, 532-546.

Allocation challenges in agricultural life cycle assessments and the Cereal Unit allocation procedure as a potential solution

Gerhard Brankatschk*, Matthias Finkbeiner

Technische Universität Berlin, Germany
* Corresponding author. E-mail: *Gerhard.Brankatschk@tu-berlin.de*

ABSTRACT

For agricultural life cycle assessments (LCA) several different allocation approaches are currently used. This leads to a broad range of uncertainty in LCA results (Curran, 2008; Gnansounou et al., 2009; Kim and Dale, 2002; Singh et al., 2010). ISO 14040 and 14044 give guidance on how to deal with allocation situations, but they offer a hierarchy of choices rather than a particular method (Finkbeiner et al., 2006; ISO, 2006a, b). Specific requirements for an agricultural allocation method were defined and used to test different allocation approaches. The Cereal Unit was identified as a promising denominator for an agricultural allocation procedure. Its calculation is mainly based on nutritive properties for animal feeding. It can be applied to all agricultural products. A new allocation approach for agricultural LCAs, based on the Cereal Unit, is suggested. This approach could help to solve agricultural allocation problems and might lead to more robust LCA results for services and products originated from agricultural raw materials.

Keywords: life cycle assessment, methodology, allocation, agriculture, by-product

1. Introduction

In recent years the need for the quantification of the environmental impact of products and services has grown rapidly. This is expressed by increased communication and public awareness about environmental footprints, such as product carbon footprints (Finkbeiner, 2009), in addition to full LCAs. One risk associated with this positive development is the fact that most consumers and policy makers are not fully aware of the uncertainty of LCA results related to methodological choices. Driven by the carbon footprint discussion, LCA in agriculture has gained increasing interest. There are several methodological particularities and challenges for agricultural LCAs. Here we focus on the issue of co-product allocation from agricultural systems.

1.1. Different allocation methods as a source of uncertainty in agricultural LCAs and the need for a new allocation approach

Different approaches for co-product allocation are one major reason for the uncertainty in LCA results related to methodological choices (Curran, 2008; Gnansounou et al., 2009; Kim and Dale, 2002; Singh et al., 2010) To solve the allocation problem, various strategies have been developed, but none is completely satisfying (Klöpffer and Grahl, 2009).

Agricultural LCAs are particularly error-prone, because allocation often takes place several times. The errors introduced by each allocation step propagate. Using different allocation methods, Luo et al., (2009) compare environmental effects of gasoline with those of bioethanol. The outcome is fundamentally affected by the choice of the allocation method: the results were even inverted by changing the allocation method from economic to mass or energy allocation. Lundie et al., (2007) state that “more effort needs to be invested in developing allocation procedures appropriate to specific industry sectors; if possible, physico-chemical ones”.

Another aspect of the allocation problem is the phenomenon of ignoring or double-counting of environmental burden. This systematic error might occur if the allocation approaches for two (or more) LCAs, containing co-products that are grown in the same agricultural system, are not aligned to each other. As a consequence, the sum of the sub-systems’ burdens is not equal to the total environmental burden of their common production process. This might not happen if both sub-systems are considered in one study, because ISO 14040 and 14044 requires using the same allocation approach in one study (ISO, 2006a, b). But often, LCAs are performed for each sub-product separately. Therefore this effect is likely to occur.

An agricultural example for this phenomenon is the link between dairy and biodiesel production. Rapeseed meal is used as animal feed and rapeseed oil as raw material for biodiesel. Both are obtained from the same raw material – rape seeds. If this link is ignored during the calculation of dairy LCA (rapeseed meal) and biodiesel LCA (rapeseed oil), it is likely that different allocation methods are being used. Fig. 1 shows this phenomenon in quantitative terms. Due to the use of different allocation approaches in separate assessments – here mass and economic allocation – the sum of the environmental burdens for oil and meal is not equal to the total environmental burden of the seeds. In the first example (Fig. 1, left side), 37% of the total environmental burden is being ignored by the assessments.

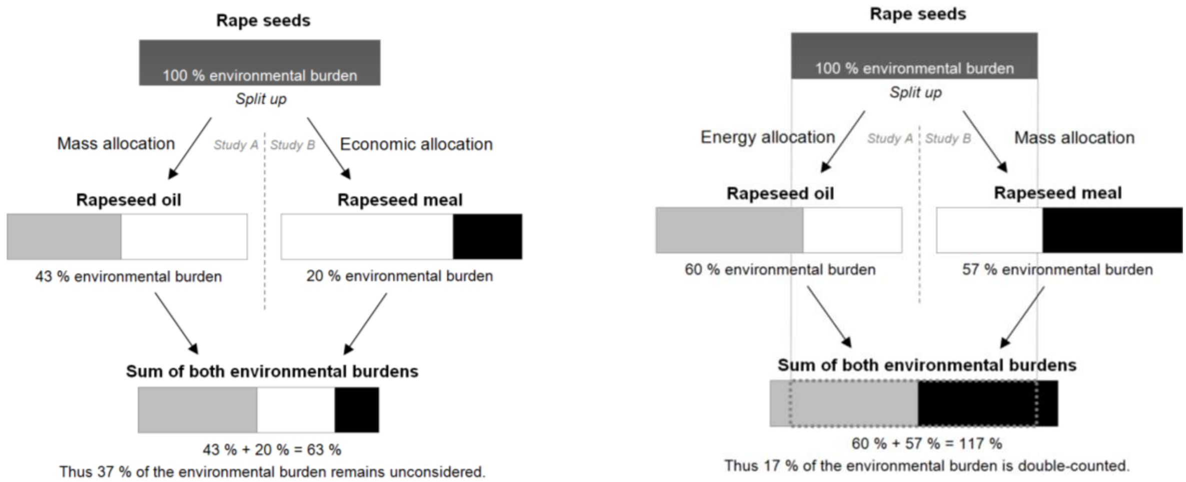


Figure 1. *Left:* Unintended ignorance of environmental burden due to different allocation approaches for by-products from the same agricultural system; *Right:* Unintended double-counting of environmental burden due to different allocation approaches for by-products from the same agricultural system. Grey and black areas represent environmental burden. Dotted lines indicate initial environmental burden (100%) and overhanging bars represent double-counted environmental burden.

The opposite effect can also be observed (Fig. 1, right side). Here the LCA practitioner might end up with an environmental burden sum for oil and meal of 117% instead of the 100% from the seeds, i.e. 17% of the burden is double-counted. That means the sum of environmental burden of one agricultural system cannot be calculated by adding up the environmental burdens of the sub-systems when different allocation methods are used. For the general interpretation of agricultural LCAs this aspect carefully needs to be considered. In terms of a contribution for solving the allocation problem for agricultural purposes, in the next chapters a new allocation approach is described.

2. Methods

2.1. The Cereal Unit and its use as new allocation approach

In the past, different estimation approaches for agricultural productivity were discussed. A simple sum of the masses of all agricultural products was recognized to be an inappropriate solution, because different levels of efforts for the production, various functionalities of the products and fulfilment of services would be neglected. To make agricultural productivity better comparable and measurable, it was realised that a weighted sum rather than a sum of masses is necessary (Becker, 1988). For this aggregation step, one common denominator with a conversion key is necessary. Amongst others, the Cereal Unit is one such common denominator for the aggregation of agricultural data. The Cereal Unit was developed by German agricultural authorities decades ago in the field of agricultural statistics and is continuously updated (Becker, 1988; BMELV, 2012; Klapp, 2011; Mönking et al., 2010). Using this Cereal Unit as common denominator, all agricultural products and by-products can be brought to the same level and thereby become comparable. We suggest using this Cereal Unit as a basis for allocation in agricultural Life Cycle Assessments.

The calculation of the Cereal Unit is mainly based on the feeding value of agricultural products. Calculation example: For barley the animal specific metabolisable energies [MJ ME / kg fresh weight] (a common parameter in animal feeding; cattle: 11.30; pigs: 12.63; etc.) are being weighted, using the share of actual feeding practices of barley (here for German conditions; cattle: 5%; pigs: 94.5%; etc.). Result is the specifically aggregated metabolisable energy content of 12.56 MJ ME / kg barley. This value was defined as 1 Cereal Unit (therefore formerly also called barley unit). The specifically aggregated metabolisable energy contents for other products are compared with the value of barley. This comparison leads to the crop specific Cereal Unit. For wheat the specifically aggregated metabolisable energy content is 13.06 ME MJ / kg wheat – therefore 1 kg wheat equals 1.04 Cereal Units.

For the sake of practicability the Cereal Unit is based on fresh weight. Because the storability of agricultural products is affected by the moisture content, each agricultural product is traded in the market with a specific moisture content or dry matter content, i.e. fresh cereals: 88% dry matter.

For agricultural products that are not feasible or not intended for animal nutrition (e.g. fruits or animal products) Cereal Unit conversion factors can be calculated via auxiliary calculations. E.g. the Cereal Unit for animal products is the quotient from fodder energy to produce 1 kilogram of product [in Megajoule metabolisable energy] and the animal specific energy content of barley [in Megajoule metabolisable energy] (Mönking et al., 2010). In consequence, for all agricultural products, Cereal Unit conversion factors theoretically can be calculated (Table 1).

Table 1. Cereal Unit conversion factors for selected agricultural products and by-products in Germany; Sources: (BMELV, 2012; Mönking et al., 2010)

Field crops and by-products (basis: fresh weight)	**Cereal Unit conversion factor**
Barley	1.00
Malt sprouts	0.74
Malt spent grains / brewers' spent grains / draff	0.75
Beer yeast	0.91
Wheat	1.04
Distillery spent wash from wheat	0.06
Rye	1.00
Cereal straw	0.43
Maize corn	1.08
Maize starch	1.07
Maize gluten	1.22
Maize gluten feed	0.82
Soybeans	1.15
Soybean oil	2.81
Soybean meal	0.96
Rape seeds	1.30
Rapeseed oil	2.74
Rapeseed meal	0.77
Sugar beet	0.23
Sugar beet leaves	0.13
Cassava / manioc	1.03
Potato	0.22
Grass, fresh	0.16
Grass silage	0.27
Grass hay	0.61
Cherries, sweet	2.73
Blueberries	3.42
Strawberries	1.16
Broccoli	0.87
Cabbage	0.18
Lettuce	0.46
Asparagus	1.71
Cattle, total, live weight	5.98
Calf, at birth, live weight	4.52
Dairy cow, live weight	6.30
Milk, unskimmed, for human nutrition	0.80
Laying hen	4.60
Eggs	2.28
Sheep, live weight	9.10
Wool, raw	1.90

3. Results

3.1. Case study for the Cereal Unit as allocation procedure

A comparison of allocation approaches was elaborated for selected agricultural products from cereal-, sugar- and oilseed-sector (Figure 2). For example, the environmental burden of a barley plant is allocated to barley grain and straw as follows: when mass allocation is applied, 56% is allocated to the grain and 44% to the straw; when energy allocation is used, 55% is allocated to the grain and 45% to the straw; when economic allocation is used 95% is allocated to the grain and 5% to the straw; when the Cereal Unit allocation is applied 77% is allocated to the grain and 23% to the straw. The results of Cereal Unit allocation are well between the outcomes of mass-, energy- and economic allocation.

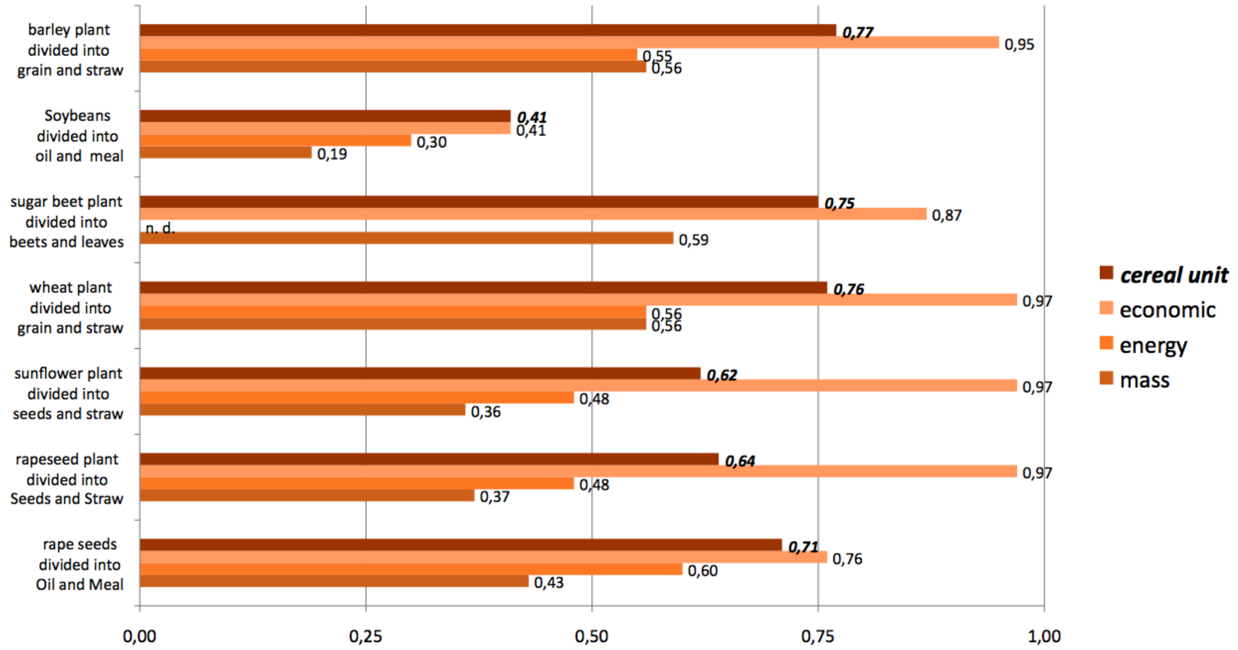


Figure 2. Comparison of mass-, energy-, economic- and Cereal Unit allocation for selected agricultural products; the sizes of bars indicate the allocation ratio between the environmental burden of the first product and the total environmental burden

4. Discussion

Usually the users of agricultural LCA results typically deal with different agricultural products. For the sake of credible results, it is their aim to treat all products and by-products as fairly and as adequately as possible. This aim could be achieved by a reliable allocation solution for all agricultural processes. This is crucial to improve the credibility of LCA results in agriculture. The way of allocating environmental burden in agricultural LCAs or within LCAs for products derived thereof, is crucial for the results, because allocation steps often take place several times. Errors introduced by each allocation step propagate. This emphasises the necessary accuracy of allocation approaches for convincing LCA results.

Cereal Unit conversion factors already are available for a large number of agricultural crops and their by-products (BMELV, 2012; Mönking et al., 2010). A suitable allocation method both for products and by-products allows LCA practitioners to use the same allocation method within independent LCAs – even if the practitioners do not know each other. For that reason the Cereal Unit allocation approach is suitable regardless of the product's final use. The suitability of the Cereal Unit for all products and by-products helps to avoid the use of different allocation methods within one product system and therefore avoids an unintended ignorance or double counting of environmental burden.

Agricultural production includes both vegetable and animal production. In many product systems animal and vegetable production are interlinked. Because the Cereal Unit is valid for vegetable and animal products, it allows LCA modellers better to depict agricultural reality. An artificial split between of crop and livestock farming becomes no longer necessary.

If Cereal Unit conversion factors are not yet available for a particular product, they can be developed based on the published calculation method (BMELV, 2012; Mönking et al., 2010). In a strict sense, the existing factors are valid for Germany only, because livestock composition and feed consumption of the region are part of the calculation of Cereal Unit conversion factor. Further conversion factors for the European and worldwide context could be developed using the published calculation method. This step seems to be necessary for the application of the Cereal Unit in other regions.

Due to the use of the Cereal Unit in agricultural statistics for decades, good balanced allocation results between different products and the acceptance for this method within the agricultural sector is expected to be high. Because the Cereal Unit combines different aspects and parameters it seems to be a promising compromise to reflect the intention of different users of agricultural products.

Decreasing the arbitrariness of LCA practitioners' choice of allocation method increases the reliability of agricultural LCAs. Improved accountability, predictability and credibility of agricultural LCAs are the basis for new, effective strategies for Sustainable Consumption and Production (SCP) for products based on agricultural raw materials.

When it comes to the allocation of environmental burden in agricultural LCAs it seems useful to think about the biggest user of agricultural area and agricultural products. Eighty percent of global agricultural land is being used to feed livestock (FAO, 2009), and the Cereal Unit reflects its needs.

5. Conclusion

While the Cereal Unit allocation approach from a theoretical point of view has a large potential, broad experience of its practical application is not yet available. To test the proposed approach and either reconfirm its suitability or identify practical drawbacks, LCA practitioners are invited to use the Cereal Unit allocation in their published results.

Application of the Cereal Unit allocation might reduce the variability and potential bias in LCA results in this sector. If the Cereal Unit allocation approach is established as a universal approach for agricultural processes, it will support the use of LCAs by providing decision makers with more robust recommendations and to achieve the aim of SCP.

6. References

Becker, J.W., 1988. Aggregation in landwirtschaftlichen Gesamtrechnungen über physische Maßstäbe - Futtergersteneinheiten als Generalnenner - [Aggregation in agricultural accounts using physical standards - feed barley units as a common denominator -], Fachbereich Agrarwissenschaften. Justus-Liebig-Universität Gießen, Gießen, p. 366.

BMELV, 2012. Statistisches Jahrbuch über Ernährung, Landwirtschaft und Forsten der Bundesrepublik Deutschland 2011 [Statistical Yearbook on Food, Agriculture and Forestry of the Federal Republic of Germany 2011]. Landwirtschaftsverlag Münster Hiltrup.

Curran, M.A., 2008. Development of life cycle assessment methodology: a focus on co-product allocation. Erasmus University Rotterdam, p. 204.

FAO, 2009. The state of food and agriculture 2009: livestock in the balance. Food and Agriculture Organization of the United Nations (FAO), Rome.

Finkbeiner, M., 2009. Carbon footprinting-opportunities and threats. Int. J. Life Cycle Assess. 14, 91-94.

Finkbeiner, M., Inaba, A., Tan, R.B.H., Christiansen, K., Klüppel, H.J., 2006. The new international standards for life cycle assessment: ISO 14040 and ISO 14044. Int. J. Life Cycle Assess. 11, 80-85.

Gnansounou, E., Dauriat, A., Villegas, J., Panichelli, L., 2009. Life cycle assessment of biofuels: Energy and greenhouse gas balances. Bioresource Technology 100, 4919-4930.

ISO, 2006a. 14044 Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organization for Standardization (ISO), Geneva.

ISO, 2006b. ISO 14040 Environmental Management – Life Cycle Assessment – Principles and Framework. International Organization for Standardization (ISO), Geneva.

Kim, S., Dale, B.E., 2002. Allocation procedure in ethanol production system from corn grain - I. System expansion. Int. J. Life Cycle Assess. 7, 237-243.

Klapp, C., 2011. Getreide- und Vieheinheitenschlüssel als Bewertungsmaßstäbe in der Landwirtschaft: internationaler Vergleich und Konsequenzen alternativer Viehbewertungen [Grain and livestock units as assessment measures in agriculture: international comparison and implications of alternative livestock reviews] Cuvillier, Göttingen.

Klöpffer, W., Grahl, B., 2009. Ökobilanz (LCA): ein Leitfaden für Ausbildung und Beruf. Wiley-VCH-Verl., Weinheim

Lundie, S., Ciroth, A., Huppes, G., 2007. Inventory methods in LCA: towards consistency and improvement - Final Report, UNEP-SETAC Life Cycle Initiative. Life Cycle Inventory (LCI) Programme Task Force 3: Methodological Consistency, p. 135.

Luo, L., van der Voet, E., Huppes, G., de Haes, H.A.U., 2009. Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol. Int. J. Life Cycle Assess. 14, 529-539.

Mönking, S.S., Klapp, C., Abel, H., Theuvsen, L., 2010. Überarbeitung des Getreide- und Vieheinheitenschlüssels - Endbericht zum BMELV-Forschungsprojekt 06HS030 [Revision of cereal unit and livestock unit - Final report on research project 06HS030 BMELV]. Georg-August-Universität Göttingen Fakultät für Agrarwissenschaften Forschungs- und Studienzentrum für Landwirtschaft und Umwelt, Göttingen, p. 537.

Singh, A., Pant, D., Korres, N.E., Nizami, A.-S., Prasad, S., Murphy, J.D., 2010. Key issues in life cycle assessment of ethanol production from lignocellulosic biomass: Challenges and perspectives. Bioresource Technology 101, 5003-5012.

LCA applied to pea-wheat intercrops: the significance of allocation

Christophe Naudin[1*], Hayo van der Werf[2,3], Marie-Hélène Jeuffroy[4], Guénaëlle Corre-Hellou[1]

[1]LUNAM Université, Groupe ESA, UR LEVA - SFR 4207 QUASAV, 55 rue Rabelais, 49007 Angers cedex 01, France
[2]INRA, UMR1069 Sol, Agro and hydroSystem, 35000 Rennes, France
[3]Agrocampus Ouest, 35000 Rennes, France
[4]UMR Agronomie INRA-AgroParisTech, Bâtiment EGER, 78850 Thiverval-Grignon, France
[*] Corresponding author. E-mail: *c.naudin@groupe-esa.com*

ABSTRACT

Cereal-legume intercrops (ICs) are a promising way to combine high productivity and several ecological benefits in temperate agro-ecosystems. This study aims to apply LCA to ICs by (i) testing several methods of allocation, or system expansion; (ii) in order to assess the environmental impacts of the co-products.

Agronomic performances of winter pea-wheat IC, sole-cropped (SC) pea and wheat were assessed in field experiments in France. LCA was carried out from sowing to harvest, including grain sorting. Functional unit is 1 kg of wheat grain (bread making quality).

ICs allowed producing wheat with lower impacts than SC wheat (at least -20%). Allocation method strongly affected results and system expansion was shown to be inappropriate, as it did not take into account benefits from interspecific complementarity. Then we redefined our functional unit to assess impacts of ICs from those of SCs (equivalence of area, or of production). ICs always decreased impacts compared to SCs (from -13% to -54%).

Keywords: system expansion; cereal; legumes; environmental impacts

1. Introduction

During recent decades, it has become obvious that the design of cropping systems has to take into account the environmental impacts of agricultural practices (Altieri, 1989) in order to limit the use of non-renewable natural resources and chemical inputs and to improve their efficiency (Tilman et al., 2002).

Intercropping is the simultaneous growing of two or more species in the same field, with variations in the species used, the densities of each species and their spatial arrangements (Willey, 1979). Apart from its frequent use in pastures, this practice is not very widespread in temperate agroecosystems. However cereal-legume intercrops are gaining interest in Europe due to the increasing awareness of environmental damage arising from the intensive use of fertilisers and pesticides and the increasing cost of these inputs (Anil et al., 1998). Moreover, intercrops are mainly widespread in organic farming but may have interesting potential uses in conventional farming systems, in particular for the development of low-input multi-use crops.

Intercropping is known to increase yields, yield stability and grain N concentration of the cereal, and to decrease weed pressure and N leaching. Indeed, cereal-legume intercropping has been demonstrated to be an interesting way to improve the nitrogen efficiency of agroecosystems (Hauggaard-Nielsen et al., 2009) and limit losses to the environment. These advantages are assumed to be mainly linked to the complementary use, in time and space, of N sources by the different components of the intercrop (Corre-Hellou et al., 2006; Jensen, 1996; Naudin et al., 2010). Thus intercrops can contribute to the development of cropping systems which combine high productivity and several ecological benefits. However, few studies have attempted to assess environmental benefits based on multicriteria approaches (Pelzer et al., 2012), and, to our knowledge, none using life cycle assessment (LCA).

LCA is a method to assess impacts of a product considering all stages of its life cycle. This methodology, so called "from cradle to grave", assesses resource use and emissions to the environment, from the extraction of resources, through each step of the production process, including product parts and recycling or final disposal (Guinée et al., 2002). In the case of product systems yielding several co-products, impacts have to be allocated among the co-products. This is the case for cereal-legume IC, and the estimated impacts of the co-products may well be dependent on the choice of the impact allocation method (Ekval and Finnveden, 2001; Heijungs and Guinée, 2007).

The aim of this study was to apply LCA to ICs by (i) testing several methods of impact allocation (mass, economic, based on N yield in grains) or avoided allocation by system expansion (ii) in order to assess the environmental impacts of the co-products.

2. Methods

2.1. General design of experiments

Field experiments were carried out in France in 2007–2008 at La Jaillière (see Exp B in Naudin et al., 2010) at the experimental station of ARVALIS Institut du Végétal, in western France (47°26'N, 0°58'W).

Winter field pea (*Pisum sativum* L.) cv. Lucy, and winter wheat (*Triticum aestivum* L.) cv. Cézanne, were sown as sole crops at 80 and 260 pl m^{-2}, respectively. Winter pea-wheat intercrops (IC) were grown in a substitutive design, each species being sown at half its sole crop density, both species being mixed within the rows. All the experiments were arranged in randomized complete block designs with three replicates. The soil was a clayey sandy loam (27.7% clay, 42.1% silt, 27.8% sand).

2.2. Crop management and analytical methods

In all experiments, pests were controlled with pesticides when required. No irrigation was provided. Inorganic soil N, measured in February (end of winter), varied from 55 to 60 kg N ha^{-1} in the 0–90 cm soil layer. N was applied as NH_4NO_3 as liquid fertiliser (Table 1). The fertiliser was enriched with ^{15}N ($\delta^{15}N$= 200‰) in order to follow the dynamics of the amount of nitrogen derived from air (Ndfa) and accumulated in pea shoots. Pea sole crops were always grown without applied N. For other details on methods concerning field experiment, see Naudin et al., (2010).

Table 1. Treatments, N fertilisation, mean grain yields and mean LER.

Crop design	Treatments	Treatments in Naudin *et al.* (2010)	Time of N-fertilization	Rate of N-fertilization (kg N ha^{-1})	Wheat GY ($g.m^{-2}$)	Pea GY ($g.m^{-2}$)	LERw	LERp	LER
SC	P100 N0	B-Psc N0	—	0	—	397 (±51)	—	—	—
SC	W100 N	B-Wsc N	07/03; 20/03; 14/05	80; 70; 40	857 (±36)	—	—	—	—
IC	P50W50 N0	B-IC N0	—	0	275 (±48)	417 (±60)	0.32 (±0.05)	1.11(±0.26)	1.43 (±0.22)
IC	P50W50 N	B-IC4	07/03	45	413 (±60)	335 (±31)	0.48 (±0.06)	0.89 (±0.18)	1.37 (±0.12)

W100: wheat sole crop. P100: pea sole crop. P50W50: substitutive intercrops of pea and wheat ("50" indicates half of the recommended plant density when sole cropped). Crops are N-fertilised ("N": 190 kg N ha^{-1} on sole cropped wheat, and 45 kg N ha^{-1} at the beginning of stem elongation on N-fertilised intercrops), or not ("N0"). LERw: partial land equivalent ratio for wheat. LERp: partial land equivalent ratio for pea. Values for grain yields are means (n=3)±SE (Standard Errors)

2.3. LCA

2.3.1. Evaluation methodology.

Potential impacts were estimated according to LCA methodology, from soil tillage for sowing to harvest (including the grain sorting process). The functional unit is 1 kg of wheat grain of bread making quality. Direct emissions were estimated based on the field experiment and International Panel on Climate Change (IPCC) 2006 recommendations. Indirect emissions were estimated with the help of the Ecoinvent 2007 database, version 2.0 (Nemecek and Kägi, 2007). The production of seed for sowing was taken into account: we assumed that inputs required for seed production were similar to those required for the corresponding crop.

2.3.2. Calculation of emissions.

Emissions to air were estimated for NH_3, N_2O and NO_x. Emission factors for NH_3 volatilisation following application of mineral fertiliser were based on Nemecek and Kägi (2007). Emission factors for N_2O were based on IPCC (2006), and emissions of NO_x were estimated according to Nemecek and Kägi (2007) at 21% of emissions of N_2O. Losses of NO_3^- to groundwater were estimated from experimental measurements. Phosphate emissions to water were estimated according to Nemecek and Kägi (2007) considering leaching to groundwater and run-off to surface water for soluble phosphate, as well as erosion of soil particles containing phosphorus.

2.3.3. Characterisation factors.

The following impact categories were considered: climate change (CC) (corresponding to greenhouse gas emissions, kg CO_2 eq.), eutrophication (EU) (g PO_4^{3-} eq.), and cumulative energy demand (CED) (MJ eq.). The indicator result for each impact category was determined by multiplying the aggregated resources used and the aggregated emissions of each individual substance with a characterisation factor for each impact category to which it may potentially contribute.

Climate change and eutrophication were calculated using the CML2 'baseline' and 'all categories' 2001 characterisation methods as implemented in the Ecoinvent v2.0 database. Cumulative energy demand (CED) was calculated according to its version 1.05 as implemented in the Ecoinvent v2.0 database. For climate change, we updated values of characterisation factors (Forster et al., 2007) for biogenic methane (new value 25 kg CO_2 eq.) and nitrous oxide (new value 298 kg CO_2 eq.). A description of the CML 2001 and CED methods can be found in Frischknecht et al., (2007).

2.3.3. Methods for allocations and system expansion

Economic allocation was calculated based on mean official price from 2000-2010 (Agreste database: www.agreste.agriculture.gouv.fr). Mean official price for wheat and pea was of 129 euros ton^{-1}, and 150 euros ton^{-1}, respectively.

System expansion was carried out by replacing impacts attributed to intercropped pea by the respective impacts estimated for sole cropped pea.

2.4. Calculations

The Land Equivalent Ratio (LER) for grain yield for pea–wheat intercrops was calculated according to De Wit and Van den Bergh (1965). The LER is the sum of the partial LER values for wheat (LERw) and pea (LERp):

$$LER = LERw + LERp = \frac{GY_{WIC}}{GY_{WSC}} + \frac{GY_{PIC}}{GY_{PSC}} \qquad \text{Eq.1}$$

where GY_{WIC} and GY_{PIC} are yields of wheat and pea in the intercrops, respectively, and GY_{WSC} and GY_{PSC} are the yields of wheat and pea in sole crops, respectively. LER values above 1 indicate a benefit of inter-cropping over sole cropping.

Nitrogen leaching was estimated based on experimental measurements using mass balance as follows:

$$N_{leaching} = N_{soil\ S} - N_{soil\ W} - N_{abv} - Ndfa \qquad \text{Eq.2}$$

where:

$N_{leaching}$ is the estimated quantity of leached N
$N_{soil\ S}$ is the N soil content observed at sowing (0–90cm soil layer)
$N_{soil\ W}$ is the N soil content observed in February after the leaching period (0–90cm soil layer)
N_{abv} is the amount of N observed in the aboveground canopy
Ndfa is the amount of N derived from air

3. Results

3.1. Estimated impacts related to wheat grain production

Estimated impacts per kg of sole cropped wheat (W100 N) were of 0.37 kg CO_2 eq., 1.17 g PO_4^{3-} eq., and 2.18 MJ eq., concerning climate change (CC), eutrophication (EU), and cumulative energy demand (CED), respectively (fig.1). Concerning the un-fertilised intercrop (P50W50 N0), mass allocations of impacts among co-products brought about impacts of 0.15 kg CO_2 eq., 0.71 g PO_4^{3-} eq., and 1.31 MJ eq., for CC, EU, and CED, respectively. N-fertilisation of the intercrop increased environmental impacts by 36%, 32%, and 17% for CC, EU, and CED, respectively (fig.1).

Methods for co-product allocation affected impacts. Concerning the un-fertilised intercrop, economic al-location impacts were 0.14 kg CO_2 eq., 0.64 g PO_4^{3-} eq., and 1.19 MJ eq., for CC, EU, and CED, respec-tively. Allocation based on the N content of grains resulted in lower impacts than for economic allocation: 0.10 kg CO_2 eq., 0.48 g PO_4^{3-} eq., and 0.90 MJ eq., for CC, EU, and CED, respectively (fig.1).

System expansion resulted in the lowest values for all impacts. In case of un-fertilised intercrops, this method always yielded negative impacts (-0.10 kg CO_2 eq., -0.68 g PO_4^{3-} eq., and -0.88 MJ eq., for CC, EU, and CED, respectively) (Fig.1).

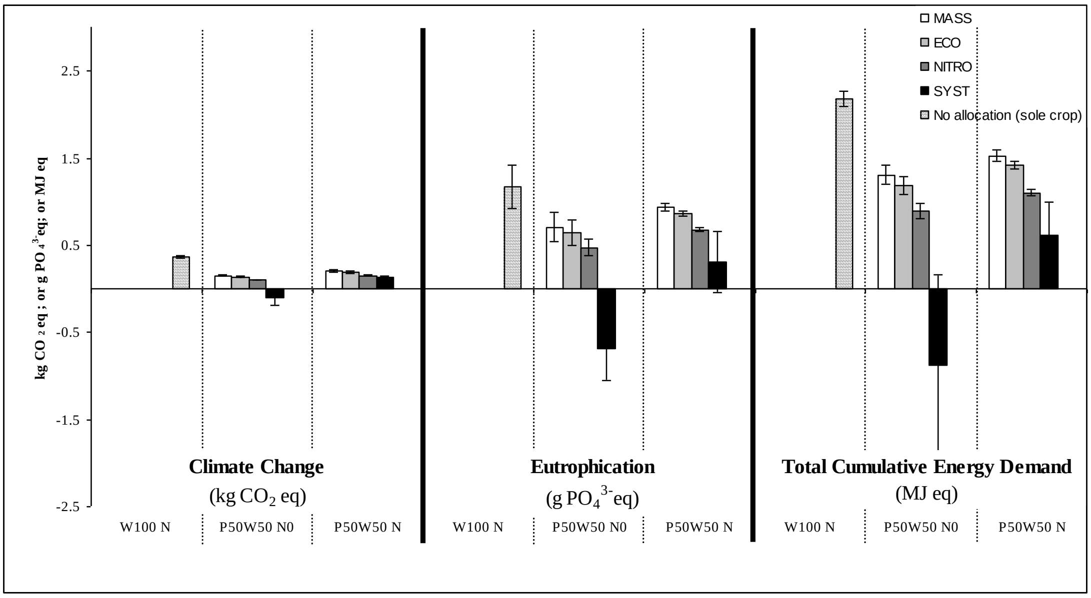


Figure 1. Potential impacts per kg of wheat grains (bread making quality) from sole cropping, and according to different propositions for allocation of impacts among grains of wheat and pea in intercrop yield. W100: wheat sole crop. P50W50: substitutive intercrops of pea and wheat ("50" indicates half of the recommended plant density when sole cropped). Crops are N-fertilised ("N": 190 kg N ha^{-1} on sole cropped wheat, and 45 kg N ha^{-1} at the beginning of stem elongation on N-fertilised intercrops), or not ("N0"). MASS: mass allocation. ECO: economic allocation. NITRO: allocation according to the quantity of Nitrogen in wheat and pea grains. SYST: avoided allocation by system expansion. Values are means (n=3) ±SE (Standard Errors).

3.3. Estimated impacts related to ICs grain production

Two comparisons were made: i) equivalence of area: impacts of IC cropped on 1 ha, compared to 1 ha of a combination of SCs according to relative sown densities in IC; ii) equivalence of production: impacts of IC cropped on 1 ha, compared to a combination of SCs producing the same quantity of wheat and pea grains as in IC (fig. 2). These comparisons took into account grain yield and LER (table 1) to perform comparison on the basis of equivalence of cropped area, and equivalence of production, respectively.

Comparison on the basis of equivalence of cropped area showed that, relative to sole crops, impacts of un-fertilised intercrops were reduced by 52%, 41%, and 36%, for CC, EU, and CED, respectively. Comparison on the basis of equivalence of production shown that, relative to sole crops, impacts of un-fertilised intercrops were reduced by 54%, 49%, and 46%, for CC, EU, and CED, respectively. N-fertilisation increased impacts relative to un-fertilised intercrops, but impacts were less than those of sole crops, whatever the way of comparison (Fig. 2).

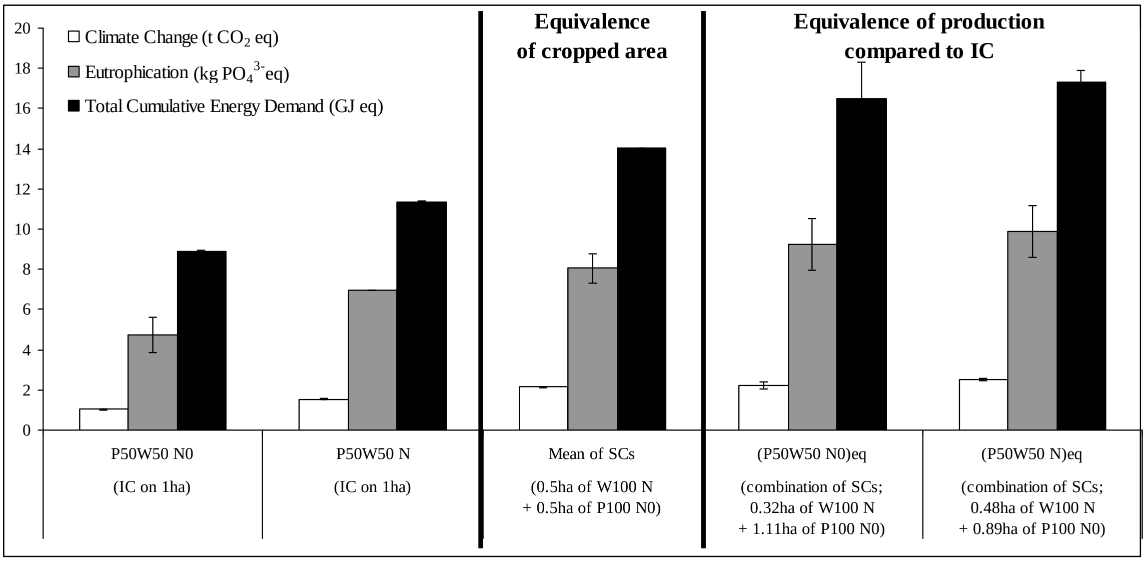


Figure 2. Potential impacts of 1 ha of pea-wheat intercrops relative to combinations of respective sole crops on a basis of equivalence of cropped area or of equivalence of production (including impacts of the grain sorting process). W100: wheat sole crop. P100: pea sole crop. P50W50: substitutive intercrops of pea and wheat ("50" indicates half of the recommended plant density when sole cropped). IC: intercrop. SC: sole crop. Crops are N-fertilised ("N": 190 kg N ha^{-1} on sole cropped wheat, and 45 kg N ha^{-1} at the beginning of stem elongation on N-fertilised intercrops), or not ("N0"). Values are means (n=3)±SE (Standard Errors).

4. Discussion

Concerning sole cropped wheat, estimated impacts for CC and CED were in accordance with previous results (Mosnier et al., 2011). Our estimated impacts for EU were lower than those exhibited by Mosnier et al., (2011) (1.17 and 3.8 g PO_4^{3-} eq., respectively). Mosnier et al., (2011) calculated mean values for France. Therefore, differences may result from the specific soil N dynamics in our experiments.

ICs allowed producing wheat with lower impacts than sole cropped wheat, whatever the allocation method (reduction was at least -20%; fig. 1). However, allocation method strongly affected results. The system expansion method did produce surprising results, as shown by negative impacts for wheat from unfertilised ICs. In our study, data used for system expansion was solid, as we estimated impacts for IC pea from the impacts associated with the production of the same quantity and quality of SC pea grains. Indeed, previous results have shown that sole cropped pea grains are of the same quality than those from intercrops (Naudin et al., 2010). In fact, inconsistent results may be explained by interspecific complementarity which allows a better efficiency in resource use but was not taken into account in the system expansion approach. Our proposition was to avoid allocations or system expansion by assessing impacts of ICs from those of SCs. Indeed, wheat and pea grains from ICs are of the same quality as those from sole cropping (Naudin et al., 2010), and introduction of ICs in cropping systems is to be considered as a substitution for a combination of both SC wheat and SC pea. Two comparisons were proposed: i) equivalence of area; and ii) equivalence of production (Fig. 2). The originality of this proposition is the introduction in LCA of approaches linked with two indicators currently used for assessing the performances of a multispecific canopy: net biodiversity effect (Loreau and Hector, 2001), and LER (de Wit and Van den Bergh, 1965). Indeed, the net biodiversity effect analyses production of biomass on the basis of an equivalence of area, and the LER on the basis of equivalence of production.

Using these ways of comparison, results showed that ICs always decreased impacts compared to SCs (from -13% to -54%). Finally, our proposition involves redefining our functional unit and estimating impacts for a mix of wheat and pea grains (including grain sorting).

5. Acknowledgment

We gratefully acknowledge the technical staff of LEVA (Laboratoire d'Ecophysiologie Végétale et Agroécologie), and of the experimental station of La Jaillière (ARVALIS Institut du Végétal) for their excellent technical assistance. This work was jointly founded by ADEME (Agence de l'Environnement et de la Maîtrise de l'Énergie), CASDAR n° 431 and n° 8058 (Compte d'Affectation Spéciale pour le Développement Agricole et Rural), Conseil Général of Maine-et-Loire, UNIP (Union National Interprofessionnel des plantes riches en Protéines), and FESIA (DIVeRE research program).

6. References

Altieri, M.A., 1989. Agroecology: a new research and development paradigm for world agriculture. Agr. Ecosyst. Environ. 27, 37-46.

Anil, L., Park, J., Phipps, R.H., Miller, F.A., 1998. Temperate intercropping of cereals for forage: a review of the potential for growth and utilization with particular reference to the UK. Grass and Forage Science 53, 301-317.

Corre-Hellou, G., Fustec, J., Crozat, Y., 2006. Interspecific competition for Soil N and its interaction with N_2 fixation, leaf expansion and crop growth in pea-barley intercrops. Plant and Soil 282, 195-208.

De Wit, C.T., Van den Bergh, J.P., 1965. Competition between herbage plants. Netherlands Journal of Agricultural Science 13, 212-221.

Ekvall, T., Finnveden, G., 2001. Allocation in ISO 14041 - a critical review. Journal of Cleaner Production 9, 197-208.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., SchulzMand van Dorland, R. 2007. Changes in atmospheric constituents and in radiative forcing. In Climate Change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, (ed. S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller), pp. 129–234. Cambridge University Press, Cambridge, UK and New York, NY, USA.

Frischknecht R., Jungbluth N., Althaus H.J., Bauer C., Doka G., Dones R., Hischier R., Hellweg S., Humbert S., Köllner .T, Loerincik Y., Margni M., Nemecek T., 2007. Implementation of life cycle impacts assessment methods. Ecoinvent report no. 3, v2.0. Swiss Centre for Life Cycle Inventories, Dubendorf, Switzerland.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., et al., 2002. Life cycle assessment. An operational guide to the ISO standards. Leiden, The Netherlands: Centre of Environmental Science, Leiden University.

Hauggaard-Nielsen, H., Gooding, M., Ambus, P., Corre-Hellou, G., Crozat, Y., Dahlmann, C., Dibet, A., von Fragstein, P., Pristeri, A., Monti, M., Jensen, E.S., 2009. Pea-barley intercropping for efficient symbiotic N2-fixation, soil N acquisition and use of other nutrients in European organic cropping systems. Field Crops Research, 8.

Heijungs, R., Guinée, J.B., 2007. Allocation and 'what-if' scenarios in life cycle assessment of waste management systems. Waste Management 27, 997-1005.

Intergovernmental Panel on Climate Change (IPCC) 2006. 2006 IPCC guidelines for national greenhouse gas inventories (ed. HS Eggleston, L Buendia, K Miwa, T Ngara and K Tanabe), IGES, Japan.

Jensen, E.S., 1996. Grain yield, symbiotic N2 fixation and interspecific competition for inorganic N in pea-barley intercrops. Plant and Soil 182, 25-38.

Loreau, M., Hector, A., 2001. Partitioning selection and complementarity in biodiversity experiments. Nature 412, 72-76.

Mosnier, E., van der Werf, H.M.G., Boissy, J., Dourmad, J.-Y., 2011. Evaluation of the environmental implications of the incorporation of feed-use amino acids in the manufacturing of pig and broiler feeds using Life Cycle Assessment. animal 5, 1972-1983.

Naudin, C., Corre-Hellou, G., Pineau, S., Crozat, Y., Jeuffroy, M.-H., 2010. The effect of various dynamics of N availability on winter pea-wheat intercrops: crop growth, N partitioning and symbiotic N_2 fixation. Field Crops Research. 119, 2-11.

Nemecek T., Kägi T., 2007. Life cycle inventories of Swiss and European agricultural production systems. Final report Ecoinvent report v2.0, no. 15, Agroscope Reckenholz-Taenikon Research Station ART. Swiss Centre for Life Cycle Inventories, Zurich and Dübendorf, Switzerland.

Pelzer, E., Bazot, M., Makowski, D., Corre-Hellou, G., Naudin, C., Al Rifaï, M., Baranger, E., Bedoussac, L., Biarnès, V., Boucheny, P., Carrouée, B., Dorvillez, D., Foissy, D., Gaillard, B., Guichard, L., Mansard, M.-C., Omon, B., Prieur, L., Yvergniaux, M., Justes, E., M.-H., J., 2012. Pea–wheat intercrops in low-input conditions combine high economic performances and low environmental impacts. European Journal of Agronomy 40, 39-53.

Tilman, D., Cassman, K.G., Matsons, P.A., Naylor, R., Polasky, S., 2002. Agricultural sustainability and intensive production practices. Nature 418, 671-677.

Willey, R.W., 1979. Intercropping - Its importance and research needs. Part 1. Competition and Yield advantages. Field Crops Abstract. 32, 1-10.

Life cycle assessment at the regional scale: innovative insights based on the systems approach used for uncertainty characterisation

Ludivine Pradeleix[1,*], Véronique Bellon Maurel[2], Philippe Roux[2], Olivier Philippon[3], Sami Bouarfa[1]

[1] Cemagref/Irstea, Joint Research Unit: G-eau, 361 rue JF Breton, 34196 Montpellier, France
[2] Cemagref/Irstea, Research Unit: Information & technologies for agro-processes, ELSA pole, 361 rue JF Breton, 34196 Montpellier, France
[3] Montpellier SupAgro, Irc (Institut des Régions Chaudes), 1001 avenue Agropolis, 34093, Montpellier, France
[*] Corresponding author. E-mail: *ludivine.pradeleix@irstea.fr*

ABSTRACT

Rapidly increasing population growth and food requirements call for increases in agricultural production, especially in irrigated areas. Environmental impacts arising from farming intensification in groundwater irrigated areas worldwide are manifold and the Life Cycle Assessment (LCA) is very relevant for assessing these impacts. But a regional LCA can not be done by transferring the "standard" product-oriented methodology to this meso-scale, especially in a context of data scarcity. Our objective is to propose a methodology to build a regional-scale Life Cycle Inventory (LCI) that would account for farming system diversity, avoid double counting and make clear allocation rules within this multi product system. We propose to base this methodology on the Agrarian System Diagnosis (ASD). This approach leads to a typology of farming systems which reflects the different agricultural exploitation modes existing on a regional scale. Enquiries are then carried out in farms representative of each type in order to build the inventory, which leads to a reduction of the uncertainty. This approach was applied on a case study located in Tunisia. Nine existing farming system archetypes and their main agricultural practices were identified and linked to their natural and socio-economic conditions. This typology goes beyond the farming system structure to describe its functioning and dynamics. Being a valuable approach for building a regional LCI, the agrarian system diagnosis could also be useful when assessing the environmental impacts of agricultural products at farm and crop scale. Indeed, this method allows us to build a typology of realistic situations instead of a virtual average system, and to support better allocation for multi product systems.

Keywords: variability, LCI, irrigation, agrarian system diagnosis

1. Introduction

Rapidly increasing population growth and food requirements call for an increase in agricultural production, out of 40% is provided by irrigated areas. The International Water Management Institute has called for "more crop per drop" (Molden, 2007). And drip irrigation is endorsed as it is less water intensive par area than surface irrigation. But eventually this technique led to manifold indirect environmental impacts such as cropland extension, global input intensification and groundwater overexploitation due to individually managed tube wells . It is therefore essential to develop tools in order to assess the impact of various agricultural planning scenarios in irrigated areas.

Life Cycle Assessment could be a candidate tool. As public water management and decision making is carried out, Life Cycle Assessment (LCA) should be applied at regional scale, which is still a methodological challenge (Guinée et al., 2011). In spite of being product oriented, several authors underline LCA relevancy at regional scale, namely for a farming region, defined as a set of farms in a given geographical area (Aubin, et al., 2011; Payraudeau and Van der Werf, 2005).

Accounting for the variability of farming systems and management practices is one of the main challenges in agricultural LCA (Nemecek et al., 2010). Unlike the case described by Payraudeau et al., (2005), the farm population in an irrigated area is too large to be surveyed one by one. Reviewing 70 LCA studies conducted in tropical and semi-arid locations, Basset-Mens et al., (2010), highlighted the failure to account for farming system diversity and the lack of specific data and data collection methods. Building up farm typologies is a way of dealing with the great variability of flows related to agricultural practices (Dalgaard et al., 2006). Some studies already performed LCA at regional scale. At this scale, statistics e.g. the Farm Accounts Data Network (FADN) were used for building farm typologies (Dalgaard et al., 2006, Mishima et al., 2005), mainly because they are in line with LCA for being product oriented (Weidema and Meeusen 2000). Nonetheless, this approach cannot be widely applied: on the one hand, very few countries in the world offer agricultural statistics device and on the other hand these statistics are based on economic inputs and outputs (I/O) and thus present several drawbacks for LCA purposes. Agriculture is poorly described by I/O because it is mainly dependent on many self produced resources (Haas, et al., 2000) and also because data are expressed in monetary units; a large uncertainty (Huijbregts, 1998) is then linked with major environmental impacting flows such as fertilisers and chemicals. Moreover, the European Commission, (2010) mentioned that when performing a LCA, data related to the foreground system, i.e. field scale agricultural production, should be inventory specific.

Facing a double challenge of data scarcity in the context of Southern countries and high variability of crop management practices in irrigated areas, we propose a new methodological framework to model activity data and build a regional model of agriculture, with the objective of reducing the uncertainty. In this paper, we propose to adapt the methodology of agrarian system diagnosis for modelling farming systems to create as accurate as possible Life Cycle Inventories. This methodological framework enables us to characterise the uncertainty linked to the "real world variability" and to imprecision also called epistemic uncertainty (Huijbregts, 1998). Preliminary results of the agrarian system diagnosis conducted in Tunisia are presented. Finally relative pros and cons of this methodological framework for conducting a regional LCA are discussed.

2. Material and Methods

2.1 Adapting the methodology of Agrarian System Diagnosis for building a Life Cycle Inventory at regional scale

Below is a brief description of the several steps of the Agrarian System Diagnosis (ASD) hereafter designated "diagnosis" and of adaptations made for LCA purposes (right column). This framework for building LC inventory should allow us to reduce the global uncertainty in the LCA; this demonstration is part of the results.

Table 1. Proposed methodology for adapting the Agrarian System Diagnosis (ASD) to Life Cycle Assessment.

Step n°	Major Steps of Agrarian Systems Diagnosis	Original Agrarian Systems Diagnosis	Agrarian Systems Diagnosis LCA oriented
0	Choice of a pilot zone Study of available information sources: Maps, previous studies (soil, slope, climate, water resources)	Selection criteria: Most of knowledge	Selection criteria: Worst case with regard to potential environmental impacts
1a	Landscape analysis / identification of agro ecological units (soil, slope) and pre-types of cropping and livestock systems	Identify cultivation dynamics, spatial distribution	Identify vulnerable areas with regard to major impacts
1b	Historical analysis / interviews Climate hazards frequency	For capturing past differentiation processes.	Identify innovative systems Foresee potential evolutions
2	Surveys of cropping and livestock systems (diversity, varieties, soil fertility management, animal feeding calendar…)	Focus on spatial distribution, crop sequences, crop-animal interactions	Investigate co-product destination, material flows between farms
3a	Sampling design of farms to be surveyed for each farming system pre-type (steps 1 &2). Sampling criteria: maximise diversity, farms chosen according to criteria explaining diversity.	Farms illustrating differentiation processes	Focus on potential environmental impacts drivers (contrasted yields, fertilisers and agrochemicals). Select farms with most records.
3b	In depth interviews of Farming Systems: techno-economic characterisation: cross-checking of qualitative and quantitative information, iterative process, systems triangulation procedure	Focus on Practices and Economy Focus on farm strategy, opportunities and bottlenecks linked to capital, labour force, etc.	Focus on input/output quantification (e.g. fertilisers & agrochemicals), including internal flows
4	Extrapolation to the whole area	Based on local knowledge about the representation of each type in the whole area.	Only necessary for "snapshot" LCA, not for agricultural planning scenarios.

We propose to mobilise the Agrarian System Diagnosis (ASD), hereafter designated "diagnosis", to model the agricultural region for LCA purposes. ASD was initially designed for targeting farm diversity in development projects. For being systems oriented, ASD aims at understanding the diversity and complexity of regional agricultural production modes at different scales and then model them into a farming systems typology. Farming systems all together are interconnected and compose the agrarian system at the regional scale (Cochet, 2011). Each farming system is modelled as functionally representative of a set of comparable production units. These units carry out a given combination of cropping systems (crop rotation and associated cultivation practices) and livestock systems and rely on comparable resources and socio-economics constraints (Cochet and Devienne, 2006; Moreau et al., 2012).

The modelling process is progressive but not linear, and iterative with several feedback procedures. Table 1 is a brief description of the several steps of the ASD and of some adaptations made for LCA purposes. Starting from a global standpoint by analysing landscape heterogeneity on maps (# 0) and in the field (#1a), several hypothesis about spatial distribution of cropping systems are formulated. Then, assumptions are checked during field surveys (#2) and cropping systems are modelled; other hypothesis on their combination are made into a pre-typology of farming systems. For each pre type, a set of representative farms is sampled (#3a) and in depth interviewed (#3b). Finally, an archetype is designed, whose agricultural practices and economical values are modelled for a “normal year”, i.e. exceptional events are not modelled (#1b). The archetype is modelled for being for the most probable case according to the farm structure, its objective, opportunities and constraints. This approach is system-oriented: it uses triangulation for ensuring data reliability, cross checking structural, functional and historical information about farming systems. In the same vein, disciplinary viewpoints and scales of analysis enrich data consistency. Finally, technical and economic thresholds are calculated for each farming system for outliers identification. A restitution to surveyed people and local expert allows us checking data completeness and validating their reliability.

2.2 Characteristics of the irrigated plain of Kairouan, Tunisia

Located in central Tunisia, in semi arid to arid climate, area under study is an alluvial plain of 30 000 ha and comprises around 2 000 farms. Agriculture has much evolved with drip irrigation introduction, from sheep herding and rain fed cereals and low density olive groves to irrigated vegetables, fruit orchards and high density olive groves. Groundwater provides irrigation water and is overexploited. Nonetheless, economic profitability of irrigated crops led people to drill unauthorised boreholes. A pilot area of 6 000 ha out of 30 000 ha was selected for being a hotspot in terms of water exploitation and intensification of agricultural management practices, i.e. several crop cycles per year and numerous intercropping. Data were collected by two students during a three month stay.

3. Results

Hereafter, we demonstrate how the new framework based on ASD for LCI can support the characterisation of uncertainty sources in a regional LCA and public decision making for land planning options. Uncertainty sources are manifold when aiming at modelling the Life Cycle Inventory of an agricultural region. They are usually separated into variability of the “real world” and uncertainty (Huijbregts, 1998).

3.1 Methodological output: the Agrarian System Diagnosis as a methodology to characterise uncertainty in agricultural Life Cycle Inventories

Figure 1 describes the sources of uncertainty in LCI of agricultural systems and the solutions proposed to characterise them via the ASD framework

In the upper part of the figure, uncertainty sources found at the regional scale are listed, in line with uncertainty classification proposed by Huijbregts (1998). In the lower part the way the Agrarian System Diagnosis contributes to characterise each uncertainty source is explained. In the very bottom part we describe how regional LCA outcomes can support public land planning decision making.The *Variability between Sources and Objects* (VBSO) stands for the “differences in inputs and emissions of comparable processes in a product system”; *parameter uncertainty* is caused by inaccurate, unrepresentative, incomplete data e.g. chemicals specifications; *uncertainty due to choices* originates from choices made regarding allocation, Functional Unit, and the LCIA stage that is out of our scope; model uncertainty also occurs during the LCIA stage.

Five out of the six categories of uncertainty sources are addressed by the ASD. Only model uncertainty is totally beyond our study scope.

As explained above, uncertainty related to variability of farming systems and management practices is addressed by building a functional typology of farming systems and cropping / livestock systems based on practices modelling. Practices are contextualised and different from standard technical guidelines.

The typology allows accounting for VBSO❶and spatial variability over the studied region. Then, a sample set of farms is in-depth interviewed for each identified type. Intra type variability (VBSO❷should be less than inter-type variability (VBSO❶. If not, a new type should be designed by splitting the type with high variability. Exceptional values due to temporal variability (e.g. climate hazards) for instance and looking inconsistent with regard to the functioning identified are discarded. The archetype built at farm scale is drawn after parameter uncertainty has been reduced and allocations have been made clear; This is done respectively by running several procedures of data consistency checking (cf. part 2.2) and by surveying the whole farming system and interconnections between farms.

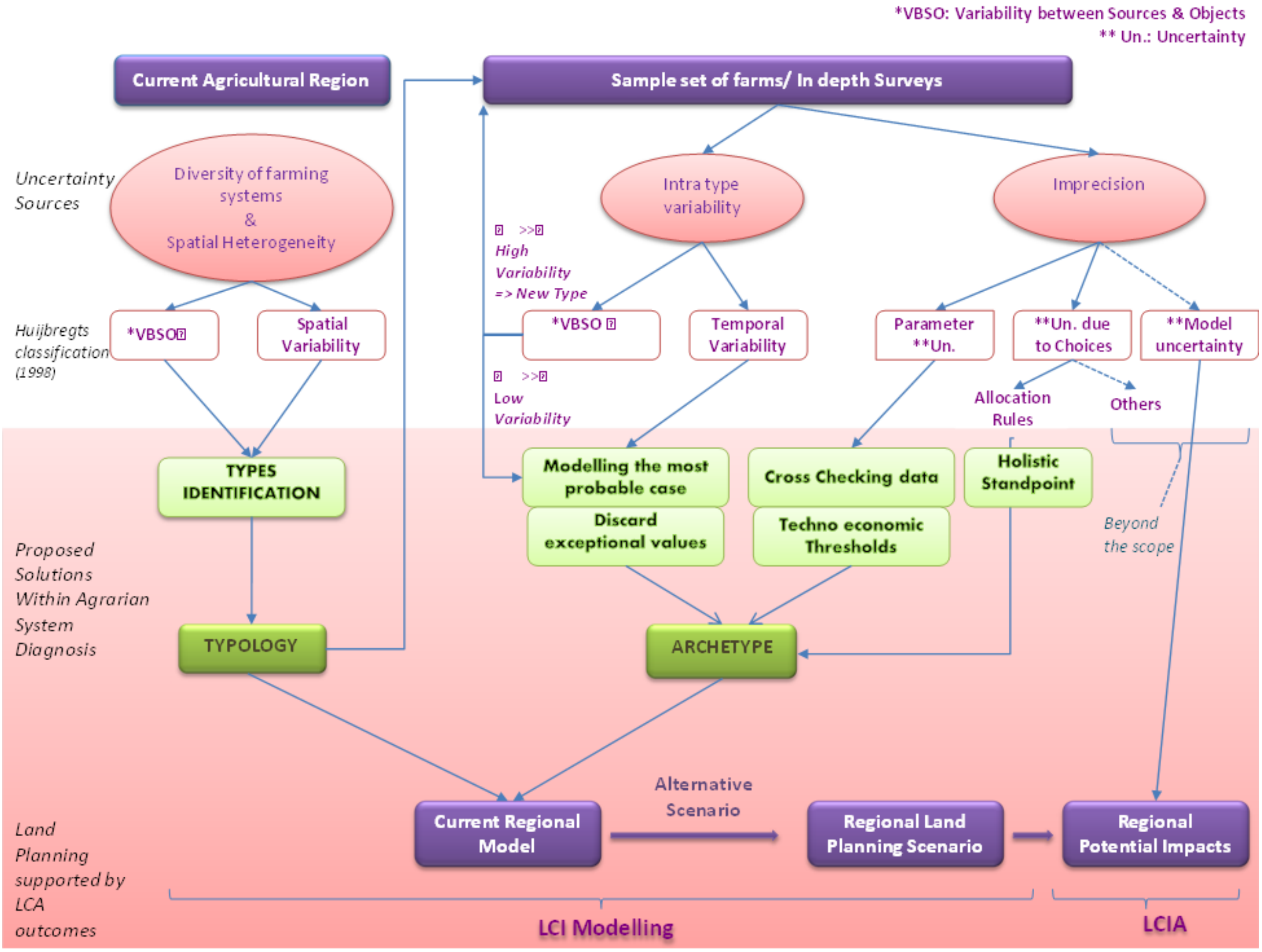


Figure 1. Sources of uncertainty in LCI of agricultural systems and solutions proposed to characterise them within the Agrarian System Diagnosis (ASD) framework

3.2 Preliminary results

Nine archetypes of farming systems and sub systems (cropping and livestock systems) were modelled in the pilot area, illustrating a high variability. A large range of cultivated species, crop sequences and associated crop management practices were identified. This is usual in irrigated areas because rainfall is no more a limiting factor for growing crops. Olive groves, a crop shared by most of farming systems, is part of very diverse cropping systems. Indeed, density, irrigation management and intercropping vary among olive-grove based cropping systems. Vegetables intercropping is widespread, mostly during unproductive stage of perennial crops. The smaller the cultivated area is, the more crops are intercropped. The level of intensification regarding crop inputs and water quantities vary from a factor one to five. Land productivity is high in case of several crop cycles per year or overlapping crop cycles. These results illustrate the degree of complexity of cropping systems in irrigated areas, especially in case of intercropping, and the need for having a typology-based inventory.

Indeed, the methodology proposed for archetype modelling enable us to design the typology, according to uncertainty source and also according to its magnitude in case of "Variability Between Sources and Objects".

4. Discussion

The objectives of a farming system typology for LCA purposes were: “to lower data variability, thereby allowing a better selection of representative farms for detailed research; better determine the marginal effects of a studied change.” (Lindeijer and Weidema, 2000).

The methodological framework we propose, namely the ASD-based LCA is a powerful method for LCA-oriented data collection that accounts for farming systems and practices variability and provide LCA specific activity data “for a marginal supplementary effort.” ASD is of higher interest since farming systems and sub systems modelling are based on in depth analysis of agricultural practices that are to be turned into LCI data. Consistency of each type is ensured by calculations of technical and economical thresholds. The consistency of activity data is also enhanced by collecting data at different scales, crossing field observations and farm and literature surveys.

The farm archetype that is modelled is neither a virtual farm average (Basset-Mens et al., 2009; Dalgaard et al., 2006) nor a single farm chosen by experts for being representative (Haas et al., 2005). On the contrary, regarding temporal variability, the farm archetype is built for a “normal year” whereas statistics or farm accounting data would refer to a single year. Thus, yields were “modelled” to represent a normal year, instead of being averaged. For example, due to climate hazards yields of pepper ranged from 1.6 to 3.2t/ha; the “modelled” yield was 3t/ha and the average one 2.5t/ha, i.e. there is a 25% difference which would change significantly the LCA outcomes, especially if impacts are expressed per mass unit.

Parameter uncertainty can be large if used data are not specific. In ASD-based LCA we are able to calibrate our own data collection to LCA requirements and include critical flows expected to heavily impact conversely to statistics and FADN data (Dalgaard et al., 2006). Indeed, fertilisers elements, agrochemicals properties and details about intercropping are crucially lacking into these generic databases. Extension services acknowledge that some values, such as cultivated areas in intercropping systems, can be registered twice in local statistics (personal comm.); this may be highly misleading since this practice is widespread on our field study.

ASD allows us to identify innovative cropping systems and current tendencies even if in minority, conversely to statistics already outdated when published. The holistic standpoint provides important insights regarding allocation rules. Indeed, statistics prevent the LCA practitioner from designing allocation rules among the numerous products of the farm, especially mixed ones which are highly represented in our study area. Efole Ewoukem et al., (2012) highlighted that mixed up farming systems tend to make the most of their limited resources and allocate biomass among several productions, thus making allocation rules more complex. Other limitations of the statistic approach for building farming systems typology is that farm functioning cannot be described (Dalgaard, 2000), by-products and near-to-zero values are overlooked (Lindeijer and Weidema, 2000), and data could be too much aggregated (Cochet and Devienne, 2006).

5. Conclusion

ASD demonstrated its ability to help us designing and characterising typical farming systems and their crop and livestock components, in qualitative and quantitative terms. This method of typology is particularly relevant when data are scarce and key criteria for classifying the population of farms cannot be taken from statistics nor from expert knowledge (Dalgaard et al., 2006; Haas et al., 2005). The agrarian system approach decreases uncertainty linked to the inherent variability of “real agriculture” (i.e. farming systems and management practices). Unlike statistics that process data and deduce mathematical correlations between variables, this approach make causalities clear within the frame of each farming system functioning.

Moreover, by revealing material and energy interconnection flows between farms or within a farm, it allows for clearer burden allocation rules among the different product systems. Double counting could also be avoided through the ASD holistic standpoint. Indeed, farming systems that are diversified are likely to support interconnection flows and thus would particularly benefit from this approach.

In addition, this work supports the identification of each farming system’s room for manoeuvre to mitigate their environmental impact, within agro-ecological and technical values that define their range of existence.

Based on this methodology and the typology, our next objective is to complete the characterisation of farming systems archetypes, conduct LCA and lastly assess alternative land planning scenarios for agriculture based on the diagnosis outputs.

6. References

Aubin, B., Donnars, C., Supkova, M., Le Perchec, S. et Dorin, B., 2011. Méthodes D’évaluation. Panorama Critique. in duALIne - Durabilité De L’alimentation Face à De Nouveaux Enjeux. Questions à La Recherche, (Coords.), , in: N. Esnouf, C., Russel, M. et Bricas (ed.) Pour Une Alimentation Durable. Réflexion Stratégique duALIne. Paris, pp. 164–184.

Basset-Mens, C., Benoist, A., Bessou, C., Tran, T., Perret, S., Vayssières, J. and Wassenaar, T., 2010.Is LCA-based Eco-labelling Reasonable□? The Issue of Tropical Food Products, in *Lcafood2010.uniba.it*.

Basset-Mens, C., Kelliher, F. M., Ledgard, S. and Cox, N., 2009. Uncertainty of Global Warming Potential for Milk Production on a New Zealand Farm and Implications for Decision Making. The International Journal of Life Cycle Assessment 14, (7) 630–638.

Cochet, H., 2011. L’agriculture Comparée, Quae, Paris.

Cochet, H. and Devienne, S., 2006. Fonctionnement Et Performances Économiques Des Systèmes De Production Agricole : Une Démarche à l 'échelle Régionale. Cahiers Agricultures 15, 578–583.

Dalgaard, R., Halberg, N., Kristensen, I. S. and Larsen, I., 2006. Modelling Representative and Coherent Danish Farm Types Based on Farm Accountancy Data for Use in Environmental Assessments. Agriculture, Ecosystems & Environment 117, (4) 223–237.

Dalgaard, T., 2000. Farm Types – How Can They Be Used to Structure, Model, and Generalise Farm Data? , in: B. P. Weidema and M. J. . Meeusen (eds) Agricultural Data for Life Cycle Assessment.Vol. 2. The Hague, pp. 98–114.

Efole Ewoukem, T., Aubin, J., Mikolasek, O., Corson, M. S., Tomedi Eyango, M., Tchoumboue, J., Van der Werf, H. M. G. and Ombredane, D., 2012. Environmental Impacts of Farms Integrating Aquaculture and Agriculture in Cameroon. Journal of Cleaner Production 28, 208–214.

European Commission, 2010. ILCD Handbook - General Guide for Life Cycle Assessment-detailed Guidance, Ispra, Italy.

Guinée, J. B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T. and Rydberg, T., 2011. Life Cycle Assessment: Past, Present, and Future. Environmental science & technology 45, (1) 90–6.

Haas, G., Geier, U., Frieben, B. and Köpke, U., 2005. Estimation of Environmental Impact of Conversion to Organic Agriculture in Hamburg Using the Life-Cycle-Assessment Method, .

Haas, G., Wetterich, F. and Geier, U., 2000. Life Cycle Assessment Framework in Agriculture on the Farm Level. International Journal of Life Cycle Assessment 5, (6) 345–348.

Huijbregts, M. A. J., 1998. Application of Uncertainty and Variability in LCA. Part I: A General Framework for the Analysis of Uncertainty and Variability in Life Cycle Assessment. International Journal of Life Cycle Assessment 3, (5) 273–280.

Lindeijer, E. and Weidema, B. P., 2000. Conclusions of the Working Group on Farm Typologies , in: B. P. Weidema and M. J. . Meeusen (eds) Agricultural Data for Life Cycle Assessment.Vol. 2. The Hague, pp. 145–146.

Meeusen, M. J. . and Weidema, B. P., 2000.Agricultural Data for Life Cycle Assessments, in *European Invitational Expert Seminar on Life Cycle Assessment of Food Products*.

Mishima, S., Taniguchi, S. and Komada, M., 2005. Adaptation of Life Cycle Assessment (LCA) to Agricultural Production on a Regional Scale in Japan. Ecology 81, 671–678.

Molden, D., 2007. Water for Food, Water for Life: A Comprehensive Assessment of Water Management in Agriculture., London.

Moreau, P., Ruiz, L., Mabon, F., Raimbault, T., Durand, P., Delaby, L., Devienne, S. and Vertès, F., 2012. Reconciling Technical, Economic and Environmental Efficiency of Farming Systems in Vulnerable Areas. Agriculture, Ecosystems and Environment 147, (1) 89-99.

Nemecek, T., Knuchel, R. F., Alig, M. and Gaillard, G., 2010.The Advantages of Generic LCA Tools for Agriculture : Examples SALCAcrop and SALCAfarm, in *Life Cycle Assessment in the Agri-food Sector*.

Payraudeau, S., Van Der Werf, H. M. G. and Vertès, F., 2005. Evaluation of an Operational Method for the Estimation of Emissions of Nitrogen Compounds for a Group of Farms. International Journal of Agricultural Resources, Governance and Ecology 5, 224-246.

Payraudeau, S. and Van der Werf, H. M., 2005. Environmental Impact Assessment for a Farming Region: a Review of Methods. Agriculture, Ecosystems & Environment 107, (1) 1-19.

Generic model for calculating carbon footprint of milk - applied to Denmark and Sweden

Randi Dalgaard[1,*], Anna Flysjö[2], Jannick Schmidt[1]

[1] 2.-0 LCA consultants, Skibbrogade 5, 1, 9000 Aalborg, Denmark
[2] Arla Foods, Sønderhøj 14, 8260 Viby J, Denmark
[*] Corresponding author. E-mail: *rad@lca-net.com*

ABSTRACT

The aim of the study is to develop a tool, which can be used for calculation of carbon footprint/LCA of milk both at a farm level and at a national level. The functional unit is '1 kg energy corrected milk (ECM) at farm gate' and the applied methodology is life cycle assessment (LCA). The model includes switches that enables for, within the same scope, transforming the results to comply with 1) consequential LCA, 2) allocation/average modelling (or 'attributional LCA'), 3) IDF (2010) guide to standard LCA methodology for the dairy industry, and 4) PAS2050 (Carbon Trust et al., 2010). Results are presented for average Danish and Swedish milk produced in 2005using the four different 'switches'.

Keywords: milk, farming system, LCA methodology, system expansion, carbon footprint, attributional and consequential LCA

1. Introduction

Dairy production – from farm to retail – represents 2.7% (±26%) of global anthropogenic greenhouse gas (GHG) emissions, where about 80-90% occurs before farm gate (Gerber et al., 2010). The dairy company Arla Foods is one of many companies committed to reduce GHG emissions in the whole value chain. One action is to develop a tool to assess the carbon footprint at farm level.

The aim of the study is to establish a generic model, which can be used for calculation of carbon footprint/LCA of milk. The model has certain facilities, which makes it useful for carbon footprint calculations both at farm level and country level. The functional unit is '1 kg energy corrected milk (ECM) at farm gate'. Indirect land use (ILUC) is included in the modelling and N balances are established for all cattle and crops. The model is applicable to any country for crops and animal systems.

2. Methods

The Arla model is established as a parameterized agricultural model. Farm specific parameters e.g. milk yield, weight and ages of cattle, purchased feed and fertiliser, land use, crop yields, housing systems, crop residue fate, diesel use etc. can be entered in the model by the farmers or their advisors. The entering of some of the parameters is mandatory, while the model suggests default values for parameters that are less easily obtainable. The model consists of a cattle system, a plant cultivation system, a food industry system and some general activities (e.g. energy, transport) as described below.

The cattle system is divided into a dairy system and a beef system and includes both the adults (dairy and suckler cows respectively) and their offspring. The input of feed to the cattle system is calculated from the cattle's milk yield, weight and ages and based on IPCC (2006) and Kristensen (2011). Methane emitted from enteric fermentation and methane and nitrous oxide from manure management are calculated according to IPCC (2006). Brazil is identified as the marginal supplier of beef to the world market and an LCI of Brazilian beef is established on basis of production data from Cederberg et al., (2009), but by using the same methodologies as for the Danish and Swedish cattle systems.

The plant cultivation system contains 34 different crop activities (barley, wheat, soybeans, corn, permanent grass etc. in different countries) and the crop yields are based on the farm specific data entered or FAOSTAT (2012). GHG emissions from the plant cultivation system is calculated according to IPCC (2006) and the fraction of removed crop residues from fields cultivated in Denmark and Sweden is based on Danish data from Statistics Denmark (2012). Crop residues removed from fields cultivated in other countries are assumed to be negligible. Fertilisation levels for calculating the Danish and Swedish baseline are primarily based on Plantedirektoratet (2004) and Flysjö et al., (2008).

The food industry system produces food, but a significant share of the outputs from the food industry is used as animal feed; soybean meal, rapeseed meal, molasses, beet pulp, wheat bran etc. The inventory of the food industry system is generally based on literature data. The soybean meal system is mainly based on Dalgaard et al., (2008) and the rapeseed oil and palm oil systems are mainly based on Schmidt (2010).

ILUC (indirect land use changes) are caused by occupation of land mainly in the crop/pasture stage. The applied inventory data are obtained from the ILUC-project version 3 (Schmidt et al., 2012).

Nitrogen balances are established for both cattle-, plant cultivation- and food industry systems and it is thereby secured that all nitrogen is accounted for, hence all nitrogen inputs to the systems will leave as products (e.g. milk, meat, manure, crops residues) or emissions (e.g. nitrate, ammonia).

The applied methodology in the present paper is consequential LCA (Weidema et al., 2009) and thereby follows the international standards on life cycle assessment: ISO 14040 and 14044 and the international standard on carbon footprints on products: ISO 14067-1. (The default method to be used within Arla foods is IDF (2010), which is the method developed by the dairy industry.) System expansion is used in all multiple output processes. For example milk production results in co-production of beef, hence an equal amount of beef production in Brazil is avoided and the saved emissions from this beef production are deducted from the environmental impact of milk production. Coproduction of soybean oil together with the determining product soybean meal is another example of system expansion.

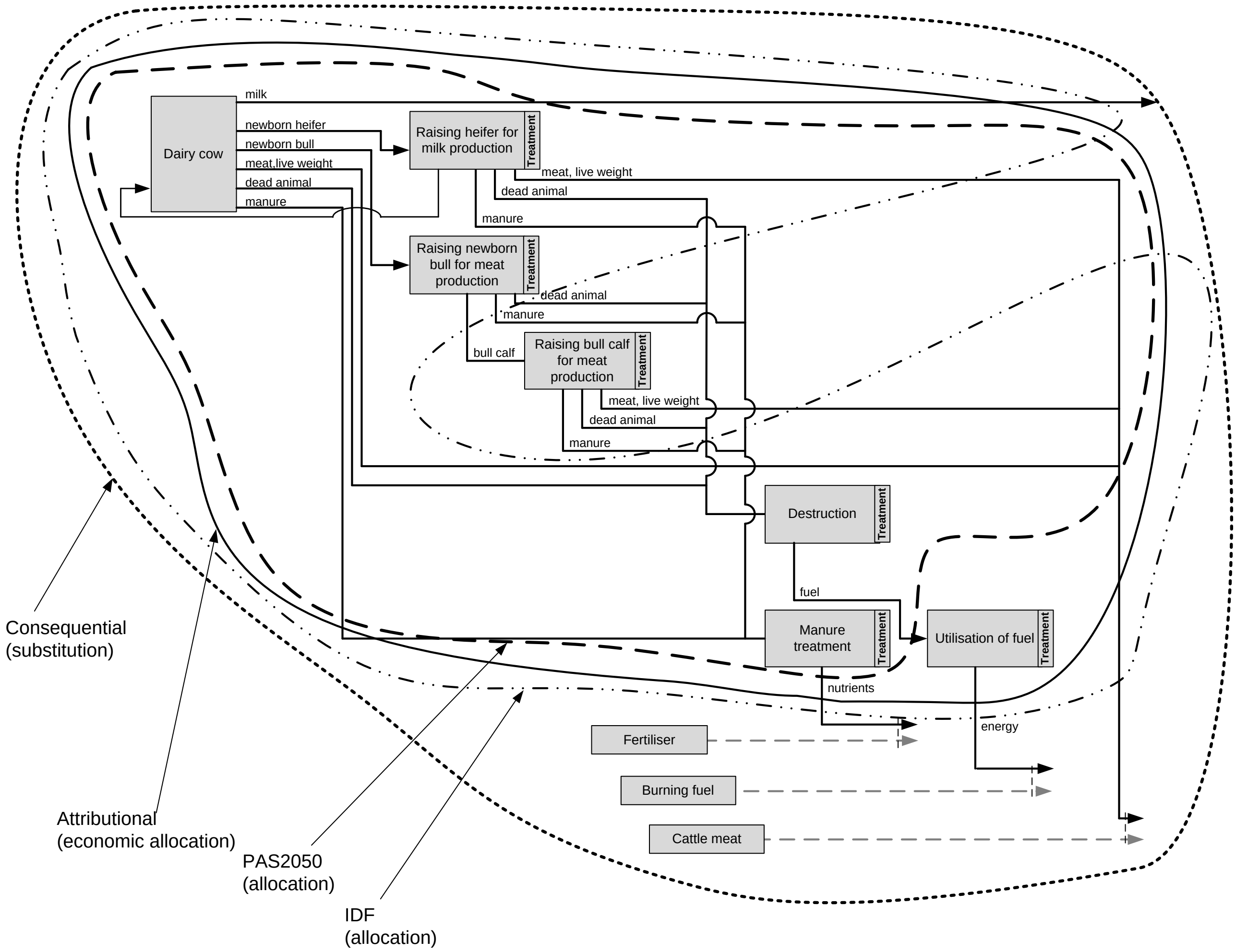


Figure 1. Dairy system with the boundaries for substitution/allocation.

The model also includes switches that enables for, within the same scope, transforming the results to comply with allocation/average modelling, IDF (2010) guide to standard LCA methodology for the dairy industry and PAS2050 (Carbon Trust et al., 2010). When these switches are turned on, allocation and average market mixes are used for the calculations. The system boundaries applied for the different standards are presented in Figure 1, where the dairy system is used as an example. The dairy system produces milk, meat, energy (from destruction of dead animals), nutrients contained in the manure and dead cattle for destruction. In the consequential modelling all the activities are included. When the allocation/average switch is turned the GHG emissions are economic allocated between milk, meat, energy (from destruction of animals) and the nutrients in the manure. When the IDF switch is turned on, the GHG emissions are allocated between milk and meat. But opposed to the allocation/average and PAS2050 switches, the raising of bull calves for meat production is not part of the milk system, i.e. the export of small bulls for further raising for meat production are excluded from the inventory. Allocation between milk and meat is based on a method using relationship between the energy content in the feed ration and milk and meat production. When PAS2050 is turned on

GHG emissions are economic allocated between milk and meat, and the allocation to nutrients (from manure) and energy (from destruction of animals) is zero.

Note that the same unallocated inventory data are used for all four switches and only the system boundaries are changed. This implies that the farm specific data can be entered in the model, and then the results can easily be presented according to the desired standards mentioned above. Similarly, the results for the Danish and Swedish baseline can be transformed to be compliant to attributional modelling, IDF or PAS2050 guidelines.

3. Results

The result for Danish baseline milk produced in 2005 is presented in Table 1. The table is divided into three major parts: 1) direct emissions of CH_4 and N_2O from the animals and their manure (from housing and storage). 2) upstream emissions from the production of feed, land use changes, utilisation of manure as organic fertiliser, fuels and their combustion etc. Note that services and capital goods are included in these figures. 3) Avoided emissions related to the substituted beef production caused by the supply of meat from the dairy system.

Table 1. GHG-emissions for 1 kg ECM, Danish baseline 2005. Switch: Consequential (ISO14044). Unit: kg CO_2-eq per kg ECM.

Denmark baseline	**Dairy cow**	**Raising heifer**	**Raising newborn bull**	**Raising bull**	**Total**	**Total**
Direct emissions						
CH_4, enteric fermentation	0.414	0.094	0.00138	0.0380	0.548	
CH_4, manure handling and storage	0.0697	0.00838	0.000347	0.00764	0.0860	
N_2O direct	0.0312	0.00605	0.000274	0.00459	0.0422	
N_2O indirect	0.00524	0.000731	0.0000216	0.000474	0.00647	0.682
Emissions outside the animal activities (incl. capital goods and services)						
Feed inputs, excl. ILUC	0.170	0.0388	0.000568	0.0156	0.225	
ILUC related to feed	1.47	0.336	0.00492	0.135	1.95	
Manure land appl. incl. subst. mineral fert.	-0.0353	-0.00125	-0.0000160	-0.00165	-0.0382	
Fuels incl. Combustion	0.00994	0.00229	0.000255	0.00175	0.0142	
Electricity	0.0369	0	0	0	0.0369	
Transport	0.0138	0.00315	0.0000461	0.00127	0.0183	
Destruction of fallen cattle incl. subst. energy	-0.00314	-0.000530	-0.000269	-0.000414	-0.00435	
Farm, capital goods	0.0113	0.0113	0.000580	0.00397	0.0271	
Farm, services	0.0149	0.0150	0.000767	0.00525	0.0359	2.26
Substituted beef system (incl. capital goods and services)						
Direct emissions (CH_4 and N_2O)					-0.399	
Feed inputs, excl. ILUC					-0.0324	
ILUC related to feed					-1.20	
Other					-0.245	-1.88
Total						1.06
Results with lower degree of completeness						
Total (result without ILUC)						0.316
Total (result without ILUC and services)						0.254
Total (result without ILUC, services and capital goods)						0.199

It appears from Table 1, that the most important contributions are ILUC (sum of ILUC from several crops/grass), avoided beef (sum of contributions from several activities within the beef system), direct emissions from the animal activities (where enteric fermentation is the most important), and the production of feedstuff (sum of all feedstuff incl. upstream activities such as diesel for traction, farm capital goods and services, and production of fertiliser and pesticides).

Transport of materials (mainly feed) to the dairy farms, burning of diesel for traction etc., and electricity do not contribute significantly to the overall result. Also the inputs of capital goods and services to the milk system are not major contributors to the overall result. It appears that the land application of manure has a negative contribution. This is because the avoided emissions from the substituted production of mineral fertilisers are larger than the direct emissions related to the application of the manure on crops. Also the destruction of animals is associated with a negative contribution because the by-products from the activity substitutes energy that alternatively would have been produced by the burning of fossil fuels.

In the lower part of Table 1, the results are shown with lower degrees of completeness. The results without ILUC are significantly lower than when including ILUC. The results without capital goods and services show, that the overall result is only affected with approximately 0.117 kg CO_2-eq. by the inclusion/exclusion of capital goods and services.

The contribution from ILUC includes contributions from transformation of land not in use (primary and secondary forest) to arable land and from intensification of land already in use. The major contribution is the one from intensification, where the emissions from additional fertiliser application are the most significant source of the GHG-emissions.

The GHG emission per kg Swedish ECM is 1.15 kg CO_2-eq and is thereby 8% higher than the Danish milk (Figure 2, first column). The overall results, i.e. the relative magnitude of different contributing activities, for Swedish milk are not significantly different from the ones for Danish milk in Table 1. The underlying reasons for the difference between Danish and Swedish milk are described in the following. The direct emissions are higher for Sweden, because the activity ‘Raising bull’ contributes more in Sweden. The reason for this is that these animals are kept for longer time and grown bigger before they are slaughtered in Sweden than in Denmark. The contributions from activities outside the animal activities are higher for Sweden, and this is also because the bulls grow bigger before they are slaughtered and thereby consume more feed.

Furthermore, the Swedish cows eat relatively less maize ensilage and grain crops and more permanent grass, which results in higher GHG-emissions. However, carbon sequestration is not included in the model and including that could result in a lower CF for permanent grass (Doreau and Dollé, 2011). The avoided emissions in Sweden are higher than in Denmark, which again is related to the higher meat output from the Swedish milk system.

Fig. 2 shows the GHG emissions per kg Danish and Swedish milk by use of the different switches as explained previously. The result for ‘Attributional’ (average/allocation) is not significantly different from the result based on consequential modelling (including ILUC). But it should be noted, that these ‘similar’ results are more a matter of incident than an indication that similar results can be expected when using consequential and attributional modelling assumptions. The most important deviation from consequential modelled results are that economic allocation is used, which implies that only 82% of the emissions and inputs are ascribed to the milk, and there is no subsituted beef system. Also, ‘ILUC related to feed’ is much lower, because attributional modelling of ILUC considers all inputs to the market for land (land tenure) as flexible and a market average mix is applied. There is no substituted beef system.

The modelling assumptions in the IDF and PAS2050 switch mode are to a large extent similar to the average/allocation attributional switch mode and the results excluding ILUC are rather similar. However, including emissions from ILUC drastically increase the CF for milk using IDF and PAS2050. The reason for the high total results for the IDF and PAS2050 switch mode is the contribution from land use changes in soy cultivation in Brazil (and minor contributions from oil palm in Malaysia). It should be noticed that the way land use changes are modelled in PAS2050 are direct land use changes (DLUC) and by applying a 20 year historical amortisation period. This approach is substantially different from the modelling of ILUC which is applied in the switch modes for ISO14040/44 consequential and average/allocation attributional. DLUC here only considers impacts from cultivated fields that have been transformed within the recent 20 years. The results when using the IDF switch mode are slightly higher than the ones of PAS2050. The reason for this is a higher degree of completeness, i.e. capital goods and services are included.

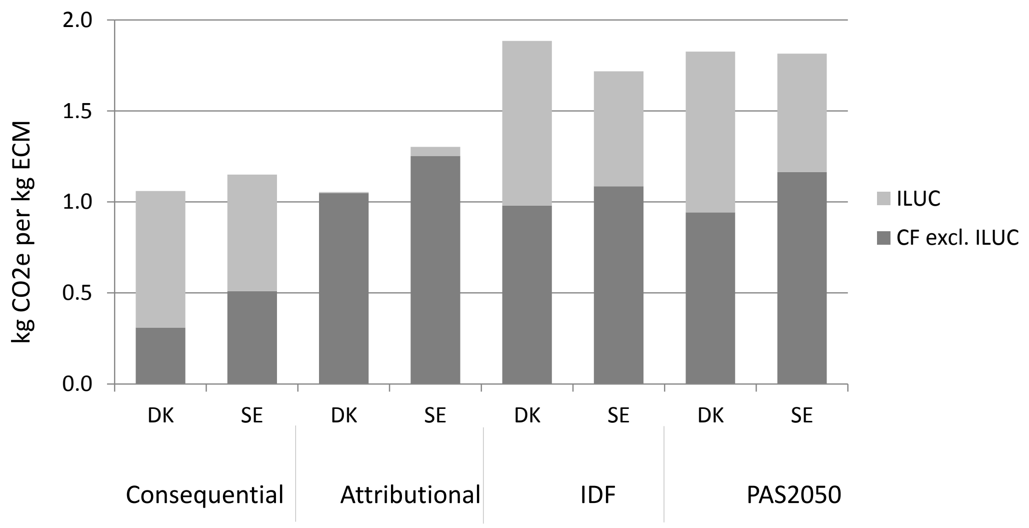


Figure 2. GHG emissions from 1 kg ECM for the Danish and Swedish baseline by use of the different switches.

4. Discussion

Sensitivity analyses are performed on region of substituted beef system (Denmark and Sweden versus Brazil), crop yields (reduced by 25%) and milk yield (reduced by 10%). The carbon footprint of Danish and Swedish milk increased by 28-31% if Danish and Swedish beef systems are used as substituted beef system. This is because these systems are more efficient compared to the Brazilian beef system. However, the beef market is global, so there is no reason to belief that the Danish and Swedish beef market are affected when extra beef is produced by the dairy cows. The effects of decreasing the crop and milks are limited.

For the national baselines, the most important assumptions relate to the identification of substituted beef system, the animal turnover, the feed composition and land use changes. The collected data on animal turnover and feed composition in Denmark and Sweden are regarded as being related to a low degree of uncertainty.

The switch mode and the exclusion of land use changes obviously affect the results significantly as shown in the results.

5. Conclusion

The Arla model which is documented in the current study is prepared for the calculation of farm specific carbon footprint as well as Danish and Swedish national baselines for milk at the farm gate.

The model is characterised by being parameterised, so that in principle any country or any specific milk farm carbon footprint can be calculated – just by entering the relevant input parameters. Of course there are some limitations in data which are entered as background data in the model, e.g. a milk farm in a country outside EU obviously uses feed inputs with other origins (countries) and maybe also types of feed that is not included in the model. Currently, the model is prepared for country and farm specific carbon footprints for Danish and Swedish milk farms and background data are based on year 2005.

Further, the model enables for applying different modelling assumptions or carbon footprint standards: Consequential (ISO 14040/44); Attributional (average/allocation); IDF guideline and PAS 2050. No additional data are required for switching between the mentioned standards. It is also possible to operate with different levels of completeness in the results by switching on and off capital goods, services and land use changes.

The GHG emissions are slightly higher for Swedish milk than for Danish milk. The major contributions to the overall result include enteric fermentation (methane emissions from the cattle) and the cultivation and production of feed inputs. A major part of the impact related to the feed inputs is associated to land use changes. However, carbon sequestration is not included in the current model, and including this might change the results (i.e. the CF could be lower for permanent pasture, which stands for a relatively higher share of the feed in Sweden compared to Denmark). Finally, the results are highly dependent on the choice of modelling switch mode.

6. References

Carbon Trust, Dairy UK and DairyCo, 2010. Guidelines for the Carbon Footprinting of Dairy Products in the UK. Carbon Trust, Dairy UK and DairyCo

Cederberg, C., Sonesson, U., Henriksson, M., Sund, V., Davis, J., 2009. Life cycle inventory of greenhouse gas emissions and use of land and energy in Brazilian beef production. SIK Report No 792. ISBN 978-91-7290-283-1.

Dalgaard, R., Schmidt, J.H., 2012. National and farm level carbon footprint of milk - Life cycle inventory for Danish and Swedish milk 2005 at farm gate. Arla Foods, Aarhus, Denmark

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W.A., 2008. LCA of soybean meal. Int J LCA (3) 240-254.

Doreau, M., Dollé, J-B. 2011. Strategies for reducing greenhouse gas emissions in dairy production. A European perspective. Conference Proceedings. Eastern Nutrition Conference, Montreal, Canada, 12-13 April 2011, 57-77.

FAOSTAT, 2012. FAOSTAT, Food and agriculture organization of the United Nations. http://faostat.fao.org/ (Accessed January 2012)

Flysjö, A., Cederberg, C., Strid, I., 2008. LCA-databas för konventionella fodermedel –miljöpåverkan i samband med produktion. SIK Report No 772. ISBN 978-91-7290-265-7. In Swedish.

Gerber, P., Vellinga, T., Opio, C., Henderson, B., Steinfeld, H., 2010. Greenhouse Gas Emissions from the Dairy Sector, A Life Cycle Assessment. FAO, Food and Agriculture Organization of the United Nations, Animal Production and Health Division, Rome, Italy.

IDF, 2010. A common carbon footprint approach for dairy – The IDF guide to standard life cycle assessment methodology for the dairy sector. Bulletin of the International Dairy Federation 445/2010

Kristensen, T., 2011. Notat: Tilpasning af funktion til estimering af foderbehov afhængig af ydelsesniveau hos malkekøerne (English: Note: Adaption of the function for establishing feed intake as a function of milk yield for dairy cows). Department of Ecology and Environment, Aarhus University, Denmark

Plantedirektoratet, 2004. Vejledning og skemaer. Gødningsplanlægning, gødningsregnskab, plantedække, harmoniregler, ændringer i ejer- og brugerforhold. 2004/2005. Ministeriet for Fødevarer, Landbrug og Fiskeri. In Danish.

Schmidt, J.H., Reinhard, J., Weidema, B.P., 2012. Modelling of indirect land use changes in LCA, report v3. 2.-0 LCA consultants, Aalborg, http://www.lca-net.com/projects/iluc_model/

Schmidt, J.H., Dalgaard, R., 2012. National and farm level carbon footprint of milk. Methodology and results for Danish and Swedish milk 2005 at farm gate. Arla Foods, Aarhus, Denmark

Schmidt, J.H., 2010. Comparative Life Cycle Assessment of Rapeseed Oil and Palm Oil. International Journal of Life Cycle Assessment 15:183–197

Statistics Denmark, 2012, http://www.dst.dk/ (Accessed January 2012)

Weidema, B.P., Ekvall, T., Heijungs, R., 2009. Guidelines for applications of deepened and broadened LCA. Deliverable D18 of work package 5 of the CALCAS project. http://fr1.estis.net/includes/file.asp?site=calcas&file=7F2938F9-09CD-409F-9D70-767169EC8AA9

LCI-dataset gap bridging strategies in the program Agri-BALYSE

Peter Koch[1,*], Gérard Gaillard[1], Thibault Salou[2,3] Anne Paillier[4]

[1]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zürich, Switzerland
[2]INRA, UMR1069, Soil Agro and hydroSystem, F-35000 Rennes, France
[3]Agrocampus Ouest, Soil Agro and hydroSystem, F-35000 Rennes, France
[4]ADEME, F-49004, France
[*]Corresponding author. E-mail: *peter.koch@art.admin.ch*

ABSTRACT

The situation is well known and each LCA-practitioner has to deal with it: Once the system model is designed and the unit processes as well as their in- and outputs are identified, they should be assigned to the appropriate life cycle inventory (LCI) in an existing database (e.g. ecoinvent). The question is only: what is appropriate? Is it appropriate to assign a “6-row self propelled tanker harvester with a 20 tons tank” used to harvest sugar beets in France e.g. to the existing ecoinvent LCI-dataset “harvesting, by complete harvester, beets, CH” knowing that this process uses a 1-row harvesting machine? The number of agricultural activities and inputs used in “real world” agricultural practice applied in France exceeds the number of available and accurate LCI-datasets by far (see table 1). While this issue may be of minor concern for a one-product LCA study, it will become a very important question when creating a multi-product LCI-database with the aim of comparison: How to deal with these “upstream-dataset gaps”? How to ensure comparable quality of the resulting LCA’s? Recently, Milà i Canals et al., (2011) suggested four different strategies to bridge data gaps (scaled, direct and averaged proxies as well as extrapolated data). In the framework of its program, Agri-BALYSE adopted this approach to state a clear strategy to face upstream-dataset gaps.

The program Agri-BALYSE is an initiative launched by the French authorities in order to develop a public LCI-database of agricultural products in France (including a small panel of imported tropical products) by the end of 2012. The program is managed by a consortium consisting of fourteen partners (ADEME, INRA, ART, CIRAD and ARVALIS, CETIOM, UNIP, IFV, ITB, CTIFL, ASTREDHOR, IFIP, ITAVI, Institut d’Elevage). As data collection is not performed centrally, the fourteen partners have developed several tools that ensure the comparability and consistency of the data: A data collection tool, an accompanying data collection guide as well as a framework of data processing tools in order to calculate the LCI data (see figure 1). An important step in this phase is the assignment of the raw data to the existing LCI-datasets.

According to the ILCD Handbook (2010), methodological consistency is a “shall-criterion” when selecting secondary data sets. Hence, the program Agri-BALYSE abstained from using datasets from several LCI-databases. Focussing on a single LCI-database, the gap-bridging-strategies proposed by Milà i Canals et al., (2011) are a suitable resort. Agri-BALYSE defined for each category of agricultural input category a specific strategy to treat inputs with missing LCI-datasets (see table 1). For fertilisers, Agri-BALYSE uses the average proxy approach by creating a proxy-LCI dataset reflecting an “average French fertiliser” based on the fertiliser consumption 2005-2009 (differentiated by N-, P- and K-fertilisers), whereas for active ingredients direct proxies are applied on the basis of their chemical structure. Agricultural machines as well as processes are extrapolated by adopting the available data sets with their main activity parameters (life time and weight for machines; energy consumption and working time for processes).

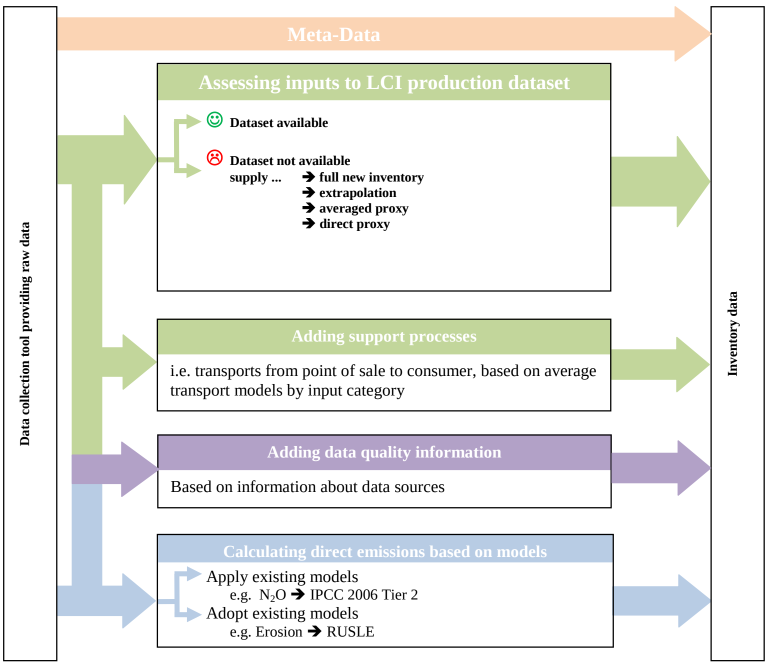


Figure 1. Assessing inputs to existing LCI-datasets is an important step when converting raw data to LCI-data. The figure shows all parts of the data processing phase in the framework of the program Agri-BALYSE: LCI-dataset assessing as well as adding of support process, quality information, meta-data and direct emissions.

Table 1. Number of available inputs per input category in the data collection tool of Agri-BALYSE

Input category	No. of inputs (real word)	No. of LCI datasets in ecoinvent V2.2	Gap bridging strategy and number of additional inventories (LCI)
Active ingredients	236	94	Direct proxies (no new LCI)
Agricultural processes	276	39	New inventories (by extrapolation) and standardisation (➔ ca. 70 LCI),
Buildings and facilities	58	18	New inventories (+58 LCI)
Feedstuffs	164	19	New inventories (+ 63 LCI), direct proxies
Fertilisers	119	25	Averaged proxies (+ 3 LCI)
Machines	200	6	New inventories (by extrapolation) and standardisation (➔ 14 LCI),
Seeds	48	28	New inventories (by extrapolation ➔ ca. 10) , direct proxies

References

Milà i Canals L, Azapagic A, Doka G, Jefferies D, King H, Mutel C, Nemecek T, Roches A, Sim S, Stichnothe H, Thoma G, Williams A, 2011. Approaches for addressing Life Cycle Assessment data gaps for bio-based products. J Ind Ecol 15(5) 707-725.

European Commission Joint Research Centre, 2010. ILCD Handbook: General Guide for Life Cycle Assessment - Detailed guidance, 1st edition, European Union, p. 247.

Cradle to gate life cycle inventory and impact assessment of glyphosate

Evan Griffing[1,*], Kiyotada Hayashi [2], Michael Overcash[1]

[1] Environmental Clarity LLC, Montgomery Village, MD, USA
[2] National Agriculture and Food Research Organization, 3-1-1 Kannondai, Tsukuba, Ibaraki, 305-8666 Japan
[*] Corresponding author. E-mail: *egriffing@environmentalclarity.com*

ABSTRACT

Glyphosate is the active ingredient in the popular broad-spectrum herbicide Roundup. The cradle to gate life cycle inventory of glyphosate was modelled with the design-based method. Industrial literature, patents, and input from agricultural chemists were used to select a representative chemistry and develop detailed models of each manufacturing process in the supply chain. The results were analysed using an energy analysis, and the global warming potential was calculated. The cumulative energy demand of 181 MJ HHV/kg glyphosate is compared to other life cycle inventory results. Process efficiencies and energy hot spots in the supply chain are described.

Keywords: life cycle inventory, herbicide, pesticide, design-based

1. Introduction

Agricultural production requires application of large numbers of herbicides and pesticides. Total herbicide use in the United States (US) in 2006 was 226,000 metric tonnes. Of this, 37% was glyphosate, a broad spectrum herbicide marketed by Monsanto as Roundup (USEPA, 2011). As of 2006, US use of herbicides as a percentage of global use was 40% by market price and 25% by mass. Glyphosate was developed as an agricultural product in the 1970s. Patent protection in the US expired in 2000. As of 1998, non-Monsanto producers represented 40,000 tonnes / yr capacity (Woodburn, 2000). By 2009, capacity in China had grown to 655,000 tonnes/yr (R & M, 2011). Global Industry Analysts predicts glyphosate production to reach 1.35 million metric tonnes by 2017.

Commercial glyphosate formulations are typically salts, which are more soluble in water. Common salts are isopropylammonium, monoammonium, diammonium, and potassium (Green, S. and Pohanish, 2007; BCPC, 2010). In this article, we present results for glyphosate as a solid. In addition, select results for the potassium salt of glyphosate are given as an example of a commercial formulation.

Commercial production routes were summarized by Bryant (2003). The Monsanto routes and Chinese chloracetic acid route described by Bryant all go through iminodiacetic acid (IDA) and phosphonomethyl iminodiacetic acid (PMIDA). The primary production route for IDA in world production, which is used in the new Monsanto route, involves diethylamine.

In the chloroacetic acid route given by Bryant, chloroacetic acid is reacted with hydrazine (NH2NH2) to form an intermediate that is converted to IDA. In another variation, chloroacetic acid can be reacted with ammonia to produce glycine, which is a starting material in a route specified by Unger (1996). Although the glycine route has not been used commercially outside of China (Bryant, 2003), it appears to be the favoured route in China (Yin, 2011).

Other life cycle inventory data on glyphosate production are available from Ecoinvent. The current version 2.2 (2010) shows production from the glycine route (Sutter, 2010) starting with acetic anhydride, formaldehyde, ammonia, sodium hydroxide, chlorine, and phosphorus trichloride. Earlier versions were based on data from Green and were given only in cradle to gate form. Results from this study are compared to the ecoinvent data.

2. Methods

Industrial literature, patents, and input from agricultural chemists were used to select a representative production route. Several routes for glyphosate were provided by Unger (1996) and Bryant (2003). The newer Monsanto route was selected as representative.

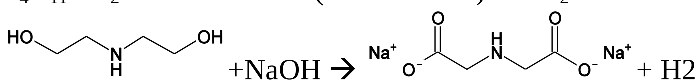
To provide complete transparency, production of each chemical was divided into gate-to-gate (gtg) processes that include a small number of primary chemical reactions. Each gtg was modelled using standard process engineering methods as outlined by Overcash (1995 and Jimenez, et. al. (2000). Reports were generated on a gate-to-gate level and include (1) all necessary chemistries, with reaction and overall process yields, (2) process descriptions and literature reviews, (3) detailed process flow diagrams including all material flows into and out of the process and process temperatures and pressures, (4) mass flow tables (5) energy flows at the unit process level.

3. Results

The chemical supply chain for glyphosate is shown in Fig. 1. Each block represents one gtg. A chemical name with a bold font indicates multi-output processes. The three gtgs with a grey background, glyphosate, PMIDA, and DSIDA may be run as an integrated glyphosate process. However, PMIDA and DSIDA are available as commercial products, and thus these were created as separate gtgs.

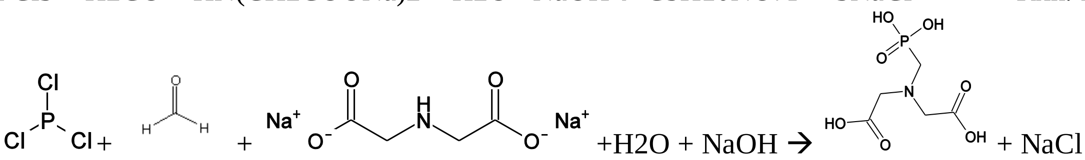
The disodium salt of iminodiacetic acid (DSIDA) is produced from diethanol amine and sodium hydroxide (Rxn. 1). The hydrogen formed is assumed to be combusted for potential heat recovery, leading to a net negative energy consumption in this gtg. DSIDA, phosphorus trichloride, formaldehyde, sodium hydroxide, and water are converted into phosphonomethyl iminodiacetic acid (PMIDA) and sodium chloride in a separate gtg. The overall reaction takes place in several steps and variations on the order of reactions are described in the patent literature (US 4,724,103; US 5,527,953; US 5,312,972; EP 599,598; US 5,527,953; US 5,688,994; US 6,515,168; WO 00/22888). Large quantities of NaCl are formed in the net reaction (Rxn. 2), and the NaCl is considered to be a chemical emission to wastewater. In the final step, PMIDA is oxidized by sparging oxygen through a series of reactors with a dilute aqueous solution of PMIDA (Rxn. 3). Carbon dioxide is a direct by-product in this reaction. Additionally, 20% of the formaldehyde by-product is assumed to be oxidized to CO2 and water. The remainder of the formaldehyde exits in a dilute aqueous waste stream.

$C_4H_{11}NO_2$+2NaOH → HN(CH2COONa)2+ $4H_2$ Rxn. 1



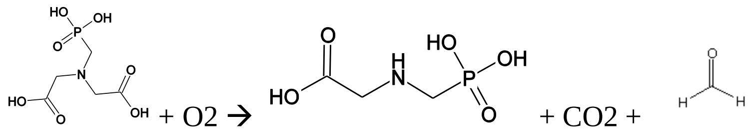
Diethanolamine + sodium hydroxide → DSIDA (disodium salt of iminodiacetic acid) + hydrogen

PCl3 + H2CO + HN(CH2COONa)2+ H2O +NaOH→ C5H10NO7P + 3NaCl Rxn. 2



phosphorus trichloride + formaldehyde + DSIDA + water + sodium hydroxide → PMIDA + sodium chloride

C5H10O7NP + ½ O2 → C4H8NO4 + CO2 + H2CO Rxn. 3



PMIDA + oxygen → glyphosate + carbon dioxide + formaldehyde

The mass yields and stoichiometric yields for each of these three gtgs are shown in Table 1. The stoichiometric yields were about 93% from each of the primary inputs (diethanol amine to DSIDA to PMIDA to glyphosate). The stoichiometric yield from diethanol amine is 80%. The mass yields, which are calculated as the total product divided by the total reactant (to each gtg) from Fig. 1, are relatively low (50 – 90%). This is due in part to the reaction stoichiometry, which produces a number of reaction byproducts that are treated as waste. The overall yield from ethylene oxide in this chemistry is < 50%, because two ethylene oxides are used per molecule of PMIDA. One of these ethylene oxide groups is oxidized to CO2 and formaldehyde in the glyphosate gtg as discussed in US 7,750,180.

Table 1. Yields for glyphosate gtgs. Apparent mass yields are from Figure 1. The other yield values are stoichiometric, and refer to overall process yield based on inputs to the process.

Gtg	Mass yield	Yield I1	Yield I2	Yield I3	Inputs corresponding to yield data
Glyphosate[a]	61%	93%	47%		I1 = PMIDA, I2 = oxygen
PMIDA	47%	93%	71%	84%	I1 = DSIDA, I2 = formamide, I3 = PCl3
DSIDA	86%	92%	88%		I1 = diethanol amine, I2 = sodium hydroxide

[a] The mass yield for DSIDA excludes the oxygen input, which is used to recover heat from hydrogen.

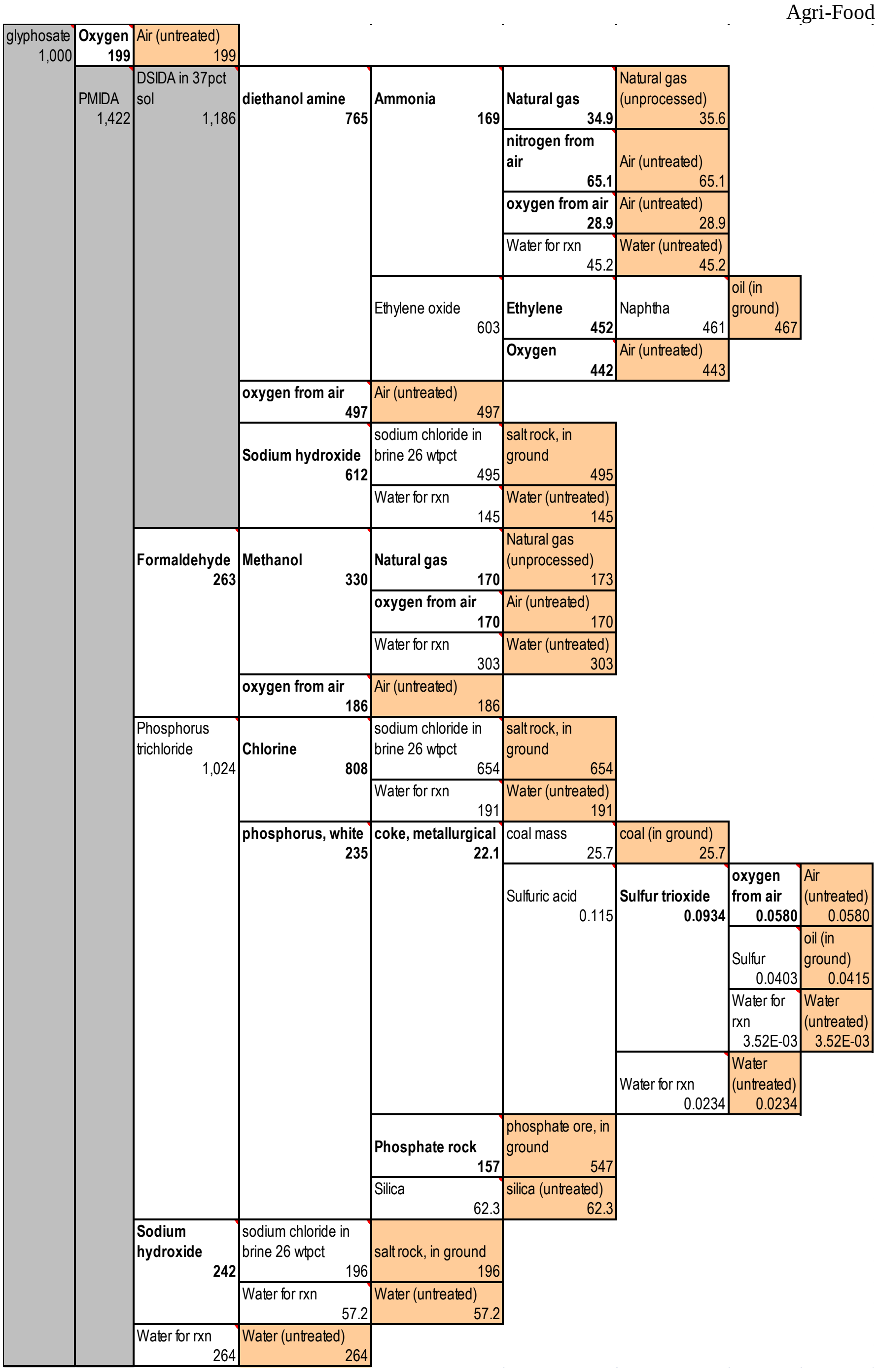


Figure 1. Supply chain for production of glyphosate from commodity chemicals

Process energy consumption includes all inputs of heat and electricity. This includes heat inputs from steam, Dowtherm, direct combustion of fuel, electricity or other sources. During cradle-to-gate (ctg) production of electricity, steam, etc., fuels are combusted, resulting in an inventory of energy emissions and fuel consumption. The ctg energy value of the fuels used (fuels in making energy plus precombustion fuel to deliver fuel to point of energy production) is the natural resource energy (nre). The sum of the nre and the energy value of the feedstocks used to build the glyphosate molecule (right-most components in Fig. 1) is comparable to the cumulative energy demand LCA metric.

The process and natural resource energies are good proxies for the environmental impact inherent in production of materials. Table 2 shows the mass of each product used to produced 1000 kg of glyphosate. For each material, the mass of byproducts and the allocation parameter are shown. For example, for each kg of ammonia produced 1.18 kg of CO_2 are produced. For mass allocation, the gtg energies, inputs, and emissions are multiplied by 1 / 2.18 = 0.459. For each kg of ammonia produced, 12.6 MJ of process energy are used. Of this, 12.6*.459 = 5.8 MJ are allocated to ammonia. In the supply chain for 1000 glyphosate, 169 kg of ammonia are used. Thus, 976 MJ of process energy are used.

At the gtg level, other processes with significant energy use and significant byproduct formation are diethanolamine, ethylene, white phosphorus, chlorine, and sodium hydroxide. In this rollup, chlorine and sodium hydroxide are used in quantities similar to their production rates, and so, the impact of allocation choice is minor. The ethylene and diethanol amine gtgs each produce products with similar utility and chemical properties. White phosphorus has a fairly small energy contribution, however, the allocation to lower value products is high. Thus, the phosphorus allocation may be important in the result.

The total process energy is 88.1 MJ/kg glyphosate. Process energies are scaled to account for cradle to gate delivery of fuel and again scaled to account for energy generation efficiencies. The total high heat value (HHV) of fuels used ctg is 148 MJ/kg glyphosate. Using this presentation, it is very easy to apply alternative energy production models and update these data as energy supplies change.

The process and natural resource energies for glyphosate production are summarized in Fig. 2.
In this view, energies for commodity chemicals are shown as ctg values. The energy profile dominated by the glyphosate gtg, which uses 63 MJ of the total 88 MJ of process energy/kg glyphosate. We can learn more about the energy use, potential variability between manufacturers, and potential for improvement by looking more closely at the glyphosate gtg. The full glyphosate gtg report contains a detailed description of the process, a process flow diagram, and Tables detailing mass and energy flows at the unit operation level. This document can be obtained by contacting the article authors. The glyphosate process takes place in a dilute aqueous solution, and about 80% of the energy use in the glyphosate gtg is for evaporation of the water. Based on most of the patent data (US 6,921,84; US 7,799,571; US 5,962,729; US 3,969,398), the water use ratio (kg water/kg PMIDA) was 10:1 to 50:1. In our gtg model, the water use was 36 kg/kg PMIDA.

Evaporation of water is an energy intensive process, and this energy can be reduced dramatically by using multi-effect evaporators. These units split the evaporation into multiple stages and utilise the steam from each stage as an energy supply for successive stages. An economic tradeoff between capital cost and operating (energy) costs favours more evaporation steps in larger processes. In the glyphosate model, a single effect evaporator was used. Thus some industrial plants may achieve significantly lower energy use by increasing reactor concentrations and using multiple effect evaporators.

Feedstock use of fossil fuels can be seen in Fig. 1. Natural gas is used to make formaldehyde and ammonia, petroleum is used to make ethylene oxide, and coal is used to make metallurgical coke for phosphorus production. Using high heat values of 29 MJ/kg for coal, 54 MJ/kg for gas, and 45 MJ/kg for oil, the total feedstock energy is 33,000 MJ/Mt glyphosate. Thus, the cumulative energy demand is 181,000 MJ/Mt glyphosate.

Two allocation choices in the glyphosate supply chain are in the ammonia gtg, which produces CO2, and the white phosphorus gtg, which produces slag and fuel gas. We tested the impact of these allocation choices by setting the allocation to ammonia and phosphorus to 1. Thus, no inventories were allocated to CO2, slag, or fuel gases. In that case, the process energy, nre, and cumulative energy demands were 97, 187, and 235 MJ/kg glyphosate.

Table 2. Cradle to gate energies for 1000 kg glyphosate.

Chemicals	Mass	By-products	Allocation factor	Energy with allocation, MJ/1000kg glyphosate						
kg / 1000 kg glyphosate		kg / kg chemical		Electricity	Dowtherm	Steam	Direct use of fuel	Transport*	Potential recovery	Total net energy
glyphosate	1,000		1.00	500	0	7.72E+04	0	440	-1.48E+04	6.33E+04
Oxygen	641	0.0553 kg Argon; 3.26 kg Nitrogen;	0.231	403	0	0	0	0	-2.88	401
PMIDA	1,422		1.00	25.1	0	2,172	0	626	-745	2,079
DSIDA in 37pct sol	1,186		1.00	340	0	1,570	0	0	-7,323	-5,413
diethanol amine	765	1.25 kg Ethanol Amine; 0.235 kg Triethanol amine; 0.0141 kg triethanol amine, 85 wt pct;	0.400	149	0	3,094	0	337	-1,043	2,536
Ammonia	169	1.18 kg CO2;	0.459	126	0	738	642	74.5	-606	976
Natural gas	205	3.00E-03 kg Butane; 0.0400 kg Ethane; 7.00E-03 kg LPG condensate; 0.0100 kg Propane;	0.943	0	0	0	698	0	0	698
nitrogen from air	65.1	0.358 kg oxygen from air;	0.736	0	0	0	0	0	0	0
oxygen from air	882	2.79 kg nitrogen from air;	0.264	0	0	0	0	0	0	0
Water for rxn	1,005		1.00	0.809	0	0	0	0	0	0.809
Ethylene oxide	603		1.00	1,169	0	22.5	0	265	-3,426	-1,970

Chemicals	Mass	By-products	Allocation factor	Energy with allocation, MJ/1000kg glyphosate						
Ethylene	452	0.480 kg C4 stream; 0.0881 kg fuel oil; 0.0484 kg Hydrogen; 0.569 kg CH4; 0.634 kg Propylene; 1.03 kg pyrolysis gas;	0.260	643	0	1,070	5,237	199	-1,280	5,869
Naphtha	461	1.55 kg heavy gas oil, from distillation; 0.644 kg kerosene, from distillation; 0.542 kg light gas oil, from distillation; 0.711 kg residum, from distillation;	0.225	110	0	51.7	910	0	0	1,072
Sodium hydroxide	854	0.886 kg Chlorine; 0.0252 kg Hydrogen;	0.523	4,045	0	2,740	0	376	-46.3	7,114
sodium chloride in brine 26 wtpct	1,345		1.00	19.2	0	1,212	0	0	0	1,232
Formaldehyde	263		1.00	202	0	1,459	0	116	-1,176	600
Methanol	330	0.0103 kg Dimethyl ether;	0.990	415	0	694	7,280	145	-6,935	1,600
Phosphorus trichloride	1,024		1.00	6.98	0	0	0	450	0	457
Chlorine	808	0.0285 kg Hydrogen; 1.13 kg Sodium hydroxide;	0.464	3,826	0	2,592	0	355	-43.8	6,729
phosphorus, white	235	1.000 kg CO; 1.41 kg Dust; 0.192 kg Ferrophosphorus; 8.10 kg Slag;	0.0855	1,172	0	0	0	103	-599	677
coke, metallurgical	22.1	7.92E-04 kg Ammonia; 7.25E-03 kg Ammonium sulfate; 0.0133 kg Benzene; 0.0533 kg coal tar from coking; 0.0703 kg coke oven gas;	0.873	0.620	0	0.0758	0.120	9.72	0	10.5
coal mass	25.7		1.00	5.61	0	0	23.0	11.3	0	39.9
Sulfuric acid	0.115		1.00	8.68E-06	0	0.0829	0	0.0506	-0.0366	0.0970
Sulfur trioxide	0.0934	0.989 kg Sulfur dioxide; 0.506 kg Sulfuric acid;	0.401	0.0474	0	0.0178	0.0523	0.0411	-0.384	-0.226
Sulfur	0.0403	already allocated	0.200	0	0	0	0.197	0.0177	0	0.215
Phosphate rock	157		1.00	34.0	0	0	4.73	69.0	0	108
Silica	62.3		1.00	0	0	0	0	27.4	0	27.4
Total process energy				1.32E+04	0	9.46E+04	1.48E+04	3,604	-3.81E+04	8.81E+04
Multiplying by pre-combustion factor to account for energy consumed prior to point of use.				1.45E+04	0	1.09E+05	1.70E+04	4,325	-4.38E+04	1.01E+05
Natural resource energy, HHV				4.54E+04	0	1.36E+05	1.70E+04	4,325	-5.47E+04	1.48E+05
Precombustion factors, MJ fuel extracted per MJ delivered (The excess is consumed in delivery)				1.1a	1.15b	1.15b	1.20	1.15b	1.15b	
Natural resource energy, MJ HHV fuel per MJ energy to process.				3.13	1.25	1.25	1.00	1.00	1.25	
a. half coal, half nuclear with no delivery										
b. half oil, half natural gas										

Table 2 (continued). Cradle-to-gate energies for 1000 kg glyphosate.

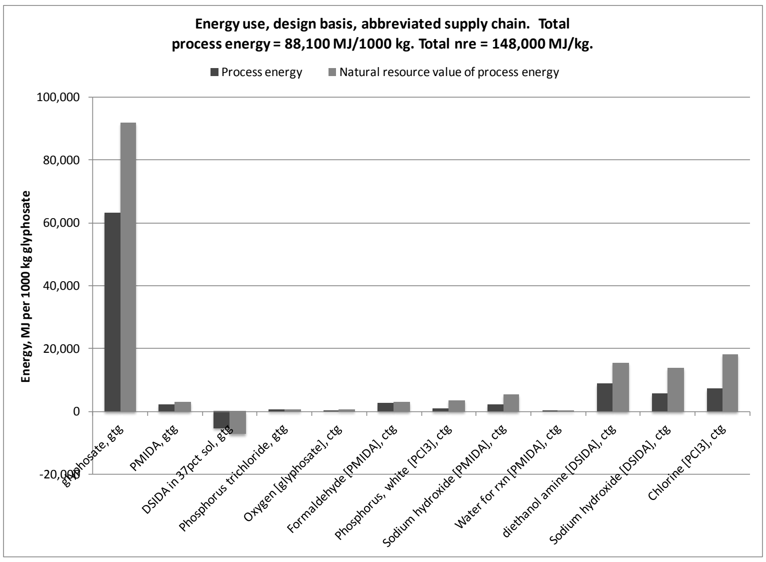


Figure 2. Cradle-to-gate energy profile for glyphosate production

Global warming potentials in particular are typically dominated by energy use. In the glyphosate supply chain, process emissions of CO_2 (no other GHG chemicals were formed from this chemical tree) are from the glyphosate gtg (oxidation of organic), ethylene oxide (over oxidation of ethylene), ammonia (coproduct), metalurgical coke production. Using a US electrical grid and excluding the ammonia coproduct CO_2, these process emissions are 0.6 kg CO_2/kg glyphosate, which corresponds to 6% of the total CO_2e emissions (10.3 kg CO_2e/kg glyphosate). Thus the transformation to gobal warming potential impact assessment had little effect, but involves more variability).

4. Discussion and Conclusions

Glyphosate production was modelled using the Monsanto route from diethanolamine. Cradle to gate energy analysis shows cumulative energy demands (all manufacturing plus energy equivalent of the fossil resources used to build the glyphosate molecule) of 181 MJ HHV/kg using mass allocation for all processes. When zero burdens were allocation to the CO_2, slag and fuel gas byproducts of ammonia and phosphorus, the cumulative energy demand was 235 MJ HHV/kg. This compares to cumulative energy demands of 406 and 226 MJ LHV/kg, for the ecoinvent models 2.1 (Green, 1987) and 2.2 (Sutter, 2010). The more recent ecoinvent model is based on a different chemistry starting from glycine. Despite the similar in cumulative energy demand result, there are some significant differences between the model presented here, and that in ecoinvent. However, the results produced by the design method for this paper are fast, inexpensive, and provide extreme transparency.

In glyphosate production, the cradle-to-gate process emissions of CO_2 were 6% of the global warming impact. A more detailed life cycle inventory and impact assessment can be obtained from the authors. The detailed analysis and documentation produced on this project represents an important change in our understanding of the life cycle of glyphosate, cyhalofop-butyl, dazomet, and chloropicrin. In the case of glyphosate, this new dataset models an important commercial production route, includes the chemical emissions from reaction by-products, and achieves a level of transparency that other LCI data suppliers should strive for.

5. References

BCPC, 2010. The Pesticide Manual, 14th edition, see www.pesticidemanual.com for details.

Bryant, R.B., 2003. Specialty Chemicals Magazine,http://www.agranova.co.uk/pdf/RJB_ChemSpec2003.pdf

Green, M. B., 1987. Energy in pesticide manufacture, distribution and use. In: Helsel Z.R. (ed.) Energy in Plant Nutrition and Pest Control, Energy in World Agriculture Vol. 2: 165-177.

Green, M., A. McCulloch, 2007. Energy considerations in the use of herbicides, J.Sci. Fd Agric, 27:95-100

Jimenez-Gonzalez, C. ,Overcash, M., 2000. Energy sub-modules applied in life cycle inventory of processes, J. Clean Products and Processes, 2, 57-66.

Overcash, M., Overcash, M. Evolving concepts in life cycle analyses, chapter in Cleaner Technologies and Cleaner Products for Sustainable Development, NATO ASI Series, Springer-Verlag, New York, NY, p 455-470, 1995.

Sutter, J., 2010. Life cycle inventories of pesticides, Swiss Centre for Life Cycle Inventories, St. Gallen

Unger, T.A., 1996. Pesticide synthesis handbook, Noyes, New Jersey.

USEPA, 2011. Pesticides Industry Sales and Usage, 2006 and 2007 Market Estimates, Prepared by Grube, A., Donaldson, D., Kiely, T., and Wu, L., United States Environmental Protection Agency.

Green, S, and Pohanish, R.P. (2005) Sittig's Handbook of Pesticides and Agricultural Chemicals, William Andrew Publishing.

R & M (2001) Research and Markets report accessed online http://www.researchandmarkets.com/reportinfo.asp?report_id=1082203&t=d&cat_id=

Woodburn, A.T. (2000) "Glyphosate: production, pricing and use worldwide," Pest Management Science 56, pp 309-312.

Yin, G. (2011). Glyphosate: There Is No Substitute, url: http://www.farmchemicalsinternational.com/cropprotection/productfocus/?storyid=2997

GHG assessment for agriculture and forestry sectors: review of landscape calculators

Vincent Colomb[1,*], Louis Bockel[2], Jean-Luc Chotte[1], Sarah Martin[3], Marianne Tinlot, Ophélie Touchemoulin, Martial Bernoux[1,*]

[1] IRD, UMR Eco&Sols (Montpellier SupAgro, CIRAD, INRA, IRD), 2 Place Viala, 34060 Montpellier Cedex 2, France.
[2] FAO, Policy and Programme Development Support Division, Policy Assistance Support Service (TCSP), via delle terme di Caracalla, 00153 Rome, Italy
[3] ADEME, Service Agriculture et Forêts, 20 avenue du Grésillé - BP 90406 - 49004 Angers cedex 1, France
[*] Corresponding authors. E-mail: *vincent.colomb@ird.fr; martial.bernoux@ird.fr;*

ABSTRACT

In parallel to IPCC work, many GHG calculators have been developed to assess agriculture and forestry practices. All these tools provide results in tonne eqCO2/ha or eqCO2/product. A review has been carried out to highlight their methodological differences, propose a typology, and discuss mains issues when working at landscape scale. Calculators were tested and questionnaires sent to the tool developers. It appears that some tools are now available for "every situation, in each part of the world", although the most suitable tool is not always easy to identify. All tools are able to identify GHG "hotspots", with exception of land use changes and soil management that are often poorly accounted. Uncertainties remain very high, limiting possibilities for economic reward/taxes for carbon actions. At last, perimeter and methodological differences impede straight comparison between studies done by different tools, and restrain managers, policy makers and non-expert users to gain reference values.

Keywords: greenhouse gas emissions, tools, landscape assessment,

Full report available on FAO-EXACT website: http://www.fao.org/tc/exact/review-of-available-ghg-tools-in-agriculture/en/

1. Introduction

Climate change has probably been the most studied impact category amongst LCA. It is also the criterion that is most likely to be adopted on short term for food labelling and implementation of green taxes. As international negotiations and state regulations on climate change goes on, policy makers and project managers are demanding for tools to move towards green economy. Land based activities, mainly agriculture and forestry, can be both sources and sinks of greenhouse gases (GHG). In most countries, they represent significant share of total GHG emissions, around 30% at global level. In parallel to IPCC work and progress on methodological issues, many GHG calculators have been developed recently to assess agriculture and forestry practices. The aim of this review is to provide users with helpful information for choosing the most suitable tool for his need, and to highlight major methodological differences between the tools. This review is complementary to other comparative studies of GHG tools for agriculture (C-AGG, 2010; Driver et al., 2010; Denef et al., 2012; Milne et al., 2012) with either a different focus (e.g. small holders) or sticking to individual tool descriptions.

2. Methods

This study focuses on calculators with a territorial/landscape approach. Generally landscape scale starts above farm scale, and implies multiple stakeholders. However, if specific landscape tools are not available, territories can always be described as a large regional/national farm. Therefore this study also includes tools working at farm scale.

A large range of calculators has been identified through internet research and cross referencing. From this extended list, only multi-activity assessment tools were selected (i.e. including at least both crop and livestock production), corresponding to 18 tools. Product specific tools, such as bioenergy tools were not included. Only tools in French, English and Spanish have been included. These farm/landscape tools have been tested and compared on several criteria regarding practical and methodological aspects. Based on this work, a pre-filled questionnaire has been sent to each tool developer for completing and validating the analysis. The analysis has been done based on the experience of GHG assessment by EX-ACT team, specialised on developing countries project assessment and ADEME team, with deep knowledge of French territorial GHG assessment.

3. Results

GHG calculation can be implemented for different reasons, depending on stakeholders and local context. For GHG landscape assessment, specific tools designed for landscape approaches should be used (Table 1). However, if these tools are not available for the study area, farm tools can be used, simulating a "regional/national" farm. This attempt to classify each tool is not strict and some tools can correspond to several categories.

- Raising awareness: set of tools usually for farmers and farming consultants. The aim is to inform them about climate change issue and the role of agriculture. The tool must be very simple (no training required); user friendly and identify hotspots. Usually there are free online tools. Tools follow typical Tiers 1 approach and have a large uncertainty. Most of them exclude soil carbon and land use change (LUC).
- Reporting: These tools are based on a landscape or farm approach, and must be able to take into account the diversity of management practices in each area. They are using Tiers 1 or Tiers 2 approach. The aim is to analyse specifically the current situation, to make comparisons between countries or farms based on a common basis and elaborate adapted policy in the future.
    - Landscape tools: Assessment of GHG emissions demanded by official institutes. Tools must avoid double counting and correspond to official standards. They have large uncertainty on results due to uncertainty on both activity data and emission factors. These tools have to use average data, they can be quite time consuming, especially for data collection.
    - Farm tools: For farmers, knowing in detail the current situation is a first step to implement reduction strategy, even if these tools are not really built to assess changes.
- Project evaluation
  Tools for project evaluation compare a baseline to a "with project" situation. They can be split in between two sub categories, depending if they are carbon market oriented
    - Focus on carbon crediting schemes: Mostly in countries where agriculture is subjected to carbon credits.
    - Not focus on carbon crediting schemes: usually account for all possible mitigation options, and especially carbon storage. However the tools must be cost efficient and user friendly. They aim at providing information for project managers, stakeholders and donors.
- Market and product oriented tools. These tools provide GHG results per product. The aim is to compare different product rather than assessing a territory. This avoids omissions of GHG emissions during leakage and indirect LUC. Usually these tools will include process and transport.

Table 3 Tool typology based on final aim

Objective of the user		**Tools and geographic zone of application**
Raising awareness		Carbon Calculator for New Zealand Agriculture and Horticulture (NZ), Cplan v0 (UK); Farming Enterprise GHG Calculator(AUS); US cropland GHG calculator (USA).
Reporting	Landscape tools	ALU (World); Climagri (FR), FullCam* (AUS)
	Farm tools	Diaterre(FR); CALM (UK); CFF Carbon Calculator (UK); IFSC (USA)
Project evaluation	Focus on ECTS schemes	Farmgas (AUS), Carbon Farming tool (NZ); Forest tools: TARAM (world), CO2 fix (world)
	Not focus on ECTS schemes:	EX-ACT (World);US AID FCC (Developing countries), CBP (World), Holos(CAN), CAR livestock tools(USA)
Market and product oriented tools		Cool farm tool (World); Diaterre (FR), LCA tools and associated database (SimaPro, ecoinvent, LCA food etc: data mainly for developed countries.)

AUS: Australia; CAN: Canada; FR: France, NZ: New Zealand; UK: United Kingdom; USA: United States of America;
FullCam: tool used by Australia for its national accounting. Only evaluate carbon and N_2O fluxes, not CH_4. High accuracy level obtained coupling extensive dataset and bio-physical process models.

Based on this classification, user can follow the following process to identify the most suitable tool for its use. An important point when choosing a tool is to select one including all major sources in the study area. For more details on the perimeter please refer to the full report (Colomb et al., 2012)

Choosing one carbon tool: a 4-step process

1.Define your aim for doing carbon evaluation and identify appropriate set of tools
2.Define geographical area and select the tool(s) that is/are available for this context
3.Check that the perimeter (forest, soil, LUC etc.) of your chosen tool is adapted to your aim, if the local tool is not adapted, you will have to choose more global tools.

4.Check your time and skill availability

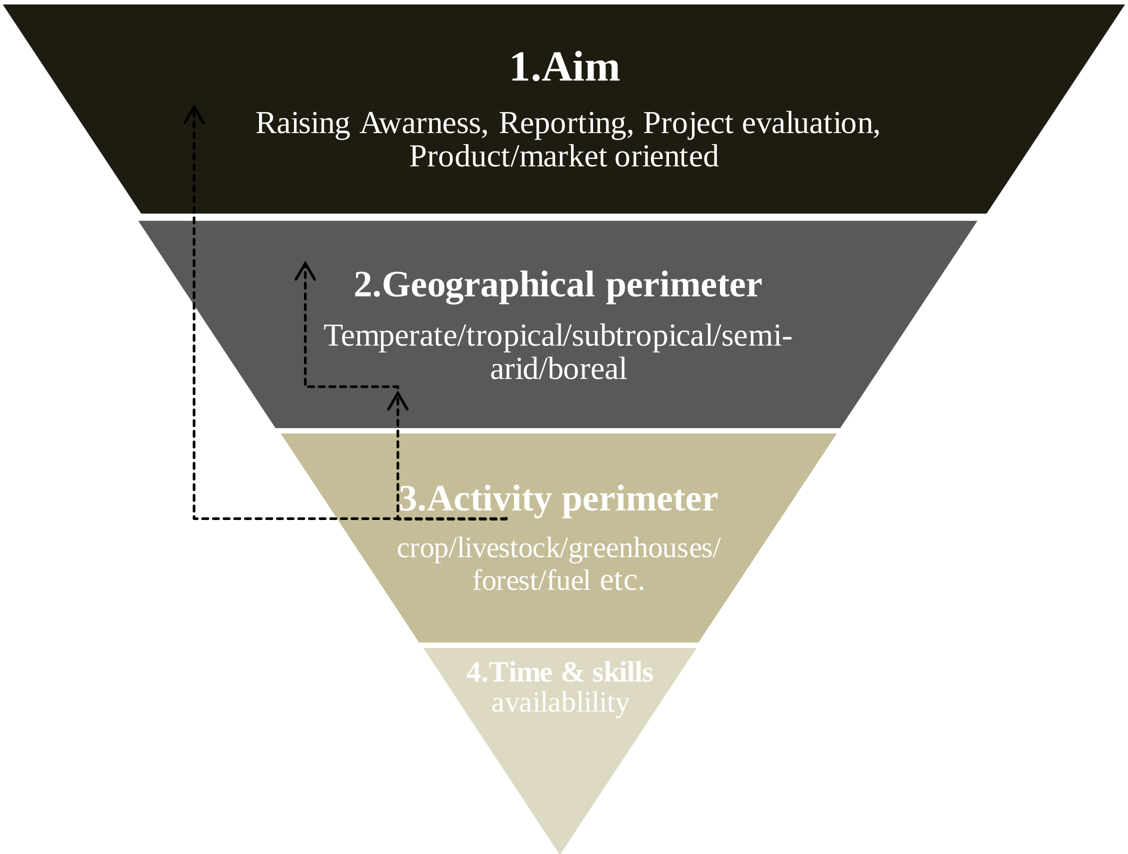


Figure 1. Tool selection process

4. Discussion

Environmental assessment at landscape scale implies several challenges (UNEP, 2012). Up scaling implies a change in data availability. At plot scale and farm scale, technical data are easily available and can be provided directly by farmers. At regional scale, data inventory often needs to be obtained from statistical data base or expert knowledge, increasing uncertainties. However uncertainty on activity data is not correlated in a linear way with scale. Indeed, it is easier to get reliable data for administrative regions such as state or counties, rather than for a watershed. Increasing scale can also reduce uncertainty with some local heterogeneity (ex: soil, management practices, micro-climate) getting balanced by working on medium rather than small scale (Post et al., 2001). In addition of these uncertainties on activity data, there are uncertainties due to year to year variability (climate and induced management practice variation) and uncertainty on emission factors themself. Only uncertainties for the emission factors are sometime mentioned by calculators.

A major challenge with landscape assessment is how to consider pedo-climatic and management heterogeneity. Indeed GHG emission and carbon stocking processes can be quite site specific, and depend strongly on mangement practices (ex: soil N_2O emissions). Moreover, good accounting of soil emission is crucial for agriculture calculators considering that N_2O account for 40% of agriculture emission on global scale, and soil carbon storage/destocking is the highest carbon sink potential, with the ability to store or release the equivalent of several years of global emissions (Baumert et al., 2005; Powlson et al., 2011). For soil related emissions, calculators either use biophysical models, such as the soil organic matter dynamic models Roth-C or Century (Cerri et al., 2004), or average emission factors provided by IPCC or national studies (IPCC 2006). The use of bio-physical models allows more accurate estimations than IPCC average factors (once the model has been properly calibrated). However, these models work at field scale and need to be linked with spatially explicit dataset ("soil maps") to work at landscape scale, these dataset not being available in most situations. In the future, proxy (NIRS) or remote sensing (satellites image analyses) technologies might enable for cheap direct measurement of the carbon stock changes or GHG emissions at large scale.

Accounting for time dynamic is also important, especially considering LUC and carbon storage. Project manager doing GHG assesment should keep in mind that landscape are under constant changes. Therefore more and more tools now suggest to evaluate a initial situation, a "business as usual" scenario and a "with project" scenario (Bernoux et al., 2010).

At landscape scale, management choices (changing, increasing or decreasing production) can induce changes on other territories (leakage), considering that food demand is not flexible. However, LUC depends not only on offer-demand balance, but also on many socio-economic parameters. Production increase can be obtained either by increase of yields (no LUC induced, but management changes induced) or by extension of cultivated land. On the ground, the drivers for LUC can be more land tenure issues, production capacities and state regulations rather than global or even local food demand. Therefore it is really difficult to establish clear consequential relationship between changes in one territory and changes in other ones. Such LUC, called indirect is calculated either by economical modelling or consequential assessment (hypothesis based on expert knowledge). Although it is clear that there are some interactions between distant territories, quantifications is really difficult and one major challenge for environmental assessment (Plevin et al., 2010; De Cara et al., 2012). So far only direct LUC is sometime accounted in the calculators.

One major point raised by this study is the lack of homogeneity concerning accounting perimeters. Indeed every GHG calculator account for different sources. Some include energy, some infrastructures and transport, some include emissions from N inputs by plant residues, some soil carbon dynamics etc. This impedes any direct comparison of results between studies done by different tools. For a better interpretation of results, users need to have references and standard in mind, seldom provided by user guides.

Results units are key criteria in the calculator structure, and strongly influence results interpretation. Results can be expressed in tonne equivalent CO_2 (Teq CO_2)/year, TeqCO_2/project (several years), TeqCO_2/year/ha, TeqCO_2/kg product. Results might also be expressed in net value (Emission – Storage); or provide both values. The most suitable unit depends on the aim of the project assessed, and the type of agriculture concerned. Indeed, industrial agriculture is clearly market oriented, has high productivity level and provides a considerable share of total food for humans. Its main challenge is to develop better efficiency and reduce the carbon footprint per kg of product, especially considering that GHG emissions are global, with no local threshold on toxicity or decontamination potential. Thus results should always be related somehow to productivity level, meaning eqCO_2/kg product, eqCO_2/kg Dry matter; eqCO_2/calorie; eqCO_2/proteins etc. Several tools are developing this approach: LCA tools; calculators with GES per kg of product, Climagri® with a "Territory Feeding potential indicator" etc. These methodologies require either allocation rules or very general productivity indicators (ex: dry matter, calories, proteins) for territories with more than one output (Schau and Fet, 2008; Cherubini et al., 2009). Not considering productivity levels in these cases induce a strong risk of leakage. On the opposite, in project oriented towards rural development, agriculture productivity is not an issue at global scale but rather a local socio-economical issue. The aim is to maximize population welfare and improve population life conditions. The eqCO_2/kg product is less suitable. Indicators should be more oriented towards socio-economy criteria, such as eqCO_2/$; eq$CO_2$/job created; eq$CO_2$/HDI point (Human development index) etc… These indicators would be a good way to promote low carbon development path for "low income countries". No such approach has been identified so far. At the moment for small holders and developing countries, calculators are more oriented toward carbon credits and possibility to get monetary benefits from reduction emissions compare to baseline.

The link between GHG assessment and economic parameter is often poor in calculators, which restrain action plan feasibility evaluation. However there has been some attempt to use carbon tools with economic tools. For instance, EX-ACT has been used with Margin Abatement Cost Curves (MACC), providing information on the cost of carbon sequestration depending on chosen options. Such studies can show that which actions are profitable for the economy, which have a reasonable cost and which are unsuitable. It also enables cross-sectorial comparison for mitigation project. Such economic approaches indicate that carbon storage and reduction of deforestation are amongst the most efficient way to fight against climate change (Smith et al., 2008). Studies indicate the potential of GHG emission for different carbon prices, showing the possible effect of a carbon tax or carbon market (Smith et al., 2008).

At last, carbon calculators are environmental assessment tools focused only on one criterion. For the analyses and solution proposed, special care for trade off must be considered (C-AGG, 2010). Some solutions that reduce carbon footprint might worsen biodiversity (ex: large biofuel plantations), increase water consumption or induce health risk (ex: growth hormone). Developing sustainable agriculture and forestry activities implies management practices that improve overall environmental footprint of products. More global methods that can be combined with carbon accounting are currently developed, such as "landscape" LCA or impact assessment analyses.

5. Conclusion

After this large review of carbon calculators, it appears that all tested calculators are accounting for main GHG sources and emissions and should be able to identify hotspots (with special care for area subjected to LUC). However there is a lack of homogeneity in methodologies, therefore it is impossible to do straight comparisons between studies done using different tools. Indeed all calculators refer to IPCC but this does not ensure homogenous approach as IPCC provide a general framework including many methodologies with different levels of details. Comparative studies are sometime available and confirm the ability of tested tool to provide coherent order of magnitudes (FAO, 2010; Soil Association Producer Support, undated). While interpreting results, it is a necessity to check for the perimeter accounted and while comparing project keep in mind uncertainties.

Some tools are now available for most activities to be assessed in every part of the world. The accuracy level is still restricted but active research is on-going and most calculator developers are frequently updating their tools. The trend is for tools to enlarge their perimeters (including more management options, more land types) and their geographical suitability. Improving accuracy implies more detailed input data, and more time demanding studies. Thus a balance must be found between efficiency and accuracy. The recent proliferation of tools testifies of this research for appropriate balance. It is not expected that one tool becomes dominant as each tool is dedicated to different situation. However there is some "competition" between tools with similar aim and geographical coverage. It might bring some confusion for non-specialist and we hope that this study will bring some clarity.

6. References

Baumert, K. A., Herzog, T., and Pershing, J. (2005). "Navigating the numbers: Greenhouse gas data and international climate policy," World Resources Inst.

Bernoux, M., Branca, G., Carro, A., Lipper, L., Smith, G., and Bockel, L. (2010). Ex-ante greenhouse gas balance of agriculture and forestry development programs. Scientia Agricola 67, 31-40.

C-AGG (2010). Carbon and Agriculture: Getting Measurable Results.http://www.c-agg.org/docs/CAGMR_complete.pdf

Cherubini, F., Bird, N. D., Cowie, A., Jungmeier, G., Schlamadinger, B., and Woess-Gallasch, S. (2009). Energy-and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. Resources, Conservation and Recycling 53, 434-447.

Colomb, V., Bernoux, M., Bispo, A., Bockel, L., Chotte, J.-l., Martin, S., Martin-phipps, C., Mousset, J., Tinlot, M., and Touchemoulin, O. (2012). "Mitigation of Climate Change by Agriculture and Forestry Sectors: Review of landscape based calculators."http://www.fao.org/tc/exact/ex-act-home/en/

De Cara, S., Goussebaïle, A., Grateau, R., Levert, F., Quemener, J., Vermont, B., Bureau, J. C., Gabrielle, B., Gohin, A., and Bispo, A. (2012). Revue critique des etudes evaluant l'effet des changements d'affectation des sols sur les bilans environnementaux des biocarburants.

Denef, K., Paustian, K., Archibeque, S., Biggar, S., and Pape, D. (2012). Report of Greenhouse Gas Accounting Tools for Agriculture and Forestry Sectors. Interim report to USDA under Contract No. GS23F8182H.

Driver, K., Haugen-Kozyra, K., and Janzen, R. (2010). Agriculture Sector Greenhouse Gas Practices and Quantification Review.

FAO (2010). Comparing results of Carbon balance appraisal using on-going Bio-Carbon fund projects. 2012.http://www.fao.org/tc/exact/validation-des-resultats/projets-du-fond-biocarbon/en/

Milne, E., Neufeldt, H., Smalligan, M., Rosenstock, T., Malin, D., Easter, M., Bernoux, M., Ogle, S., Casarim, F., Pearson, T., Bird, N., Steglich, E., Ostwald, M., Deneuf, K., and Paustian, K. (2012). "Methods for the quantification of net emissions at the landscape level for developing countries in smallholder contexts.,"

Plevin, R. J., Jones, A. D., Torn, M. S., and Gibbs, H. K. (2010). Greenhouse gas emissions from biofuels' indirect land use change are uncertain but may be much greater than previously estimated. Environmental Science & Technology.

Post, W. M., Izaurralde, R. C., Mann, L. K., and Bliss, N. (2001). Monitoring and verifying changes of organic carbon in soil. Climatic change 51, 73-99.

Powlson, D., Whitmore, A., and Goulding, K. (2011). Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. European Journal of Soil Science 62, 42-55.

Schau, E., and Fet, A. (2008). LCA studies of food products as background for environmental product declarations. The International Journal of Life Cycle Assessment 13, 255-264.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., and Rice, C. (2008). Greenhouse gas mitigation in agriculture. Philosophical Transactions of the Royal Society B: Biological Sciences 363, 789-813.

Soil Association Producer Support (undated). Carbon footprint calculators – a case study comparison review.http://www.swarmhub.co.uk/downloads/pdf/carbon_project/footprint_calculators.pdf

UNEP (2012). "UNEP Year Book: Emerging issues in our global environement. http://www.unep.org/yearbook/2012.

Management and reduction of on-farm GHG emissions using the 'Cool Farm Tool': a case study on field tomato production

Emma Keller[1,2,*], Jaqi Lee[1], Roland Clift[1] , Llorenç Milà i Canals[2], Henry King[2], Jon Hillier[3]

[1] University of Surrey, Centre of Environmental Strategy (CES), Guildford, GU2 7XH, UK
[2] Unilever, Safety and Environmental Assurance (SEAC), Colworth Science Park, Sharnbrook, MK44 1LQ, UK
[3] University of Aberdeen, Institute of Biological and Environmental Sciences, School of Biological Sciences, Cruickshank Building, St Machar Drive, Aberdeen AB24 3UU, UK
[*] Corresponding author. E-mail: *e.keller@surrey.ac.uk*

ABSTRACT

Globally, agriculture and land use change accounts for approximately 10-12% of GHG emissions (Smith et al., 2007) but also presents significant potential for mitigation (Smith et al., 2008). It has thus become a political focal point for emissions reduction. If this is to be realised, attention must be paid to the basic unit of agriculture, the farm. Effectively managing GHG emissions at the farm level can be achieved through the deployment of well documented good agricultural practices (GAPs) and resource optimisation. Quantifying and measuring farm level GHG impacts however, can provide additional insights and opportunities for reduction which farmers typically lack.

The Cool Farm Tool, developed by Hillier et al., (2011), is a multi-crop, globally applicable, farm-scale GHG calculator with high management sensitivity relevant for the farmer's specific system. It integrates a number of globally determined empirical models (Hillier et al., 2011) and requires information inputs easily accessible to farmers (e.g. fertiliser application, tillage practices), enabling them to measure their own farm GHG impact and explore different emission reduction scenarios. It therefore provides decision support to encourage low carbon farming and offers a low-cost (it's open source) and robust means of measuring on-farm GHG emissions quickly and with instant results. The tool has recently been deployed at scale through the 'Cool Farming Options' project hosted by the Sustainable Food Lab (2011). In this project a number of companies from around the world have been using the tool within a diverse range of food supply chains and in doing so, have begun to build up a multi-crop and multi-regional data repository.

This paper presents a case study of the application of the Cool Farm Tool in its first year with field tomato growers from several farms supplying into a global supply chain. It describes the emissions profiles of these farms and illustrates the variability inherent among farming systems globally as a function of variability in management practices, climate, geography, soil properties and technology used. In addition these GHG figures are compared to current published literature sources and modelled data (Nemecek et al., 2012) to demonstrate the potential of the Cool Farm Tool as a data provider at a territorial or regional level. Finally some conclusions and potential future developments for the tool methodology and use are presented.

References

Hillier, J., Walter, C., Malin, D., Garcia-Suarez, T., Milà i Canals, L. & Smith, P., 2011. A farm-focused calculator for emissions from crop and livestock production. Environmental Modelling & Software, 26, 1070-1078.

Nemecek, T, J., Weiler, K., Plassmann, K., Schnetzer, J., Gaillard, G., Jefferies, D., Garcia-Suarez, T., King, H., Milà i Canals, L., 2012. Estimation of the variability in global warming potential of a worldwide crop production using a modular extrapolation approach. Journal of Cleaner Production. (Submitted).

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, O. Sirotenko., 2007. Agriculture, in: [B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer (Eds.)], Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change., Cambridge University Press, Cambridge, United Kingdom and New York, NY, US.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, O. Sirotenko, Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M. & Smith, J. U., 2008. Greenhouse gas mitigation in agriculture, Philosophical Transactions of the Royal Society, 363, 789-813.

Sustainable Food Lab, 2011. Cool farming options. [Online] Available from: http://www.sustainablefoodlab.org/projects/climate (Accessed: 10/02/12).

PalmGHG, the RSPO greenhouse gas calculator for oil palm products

C. Bessou[1*], L. Chase[2], I. Henson[3], A.F.N. Abdul-Manan[4], L. Milà i Canals[5], F. Agus[6], M. Sharma[7]

[1] CIRAD, UPR Perennial Crops, Av. de l'Agropolis, TA B-34/02, F-34398 Montpellier Cedex 5, France
[2] Independent Consultant in Tropical Agriculture, High Trees, Martineau Drive, Dorking, Surrey RH4 2PL, UK
[3] Independent Consultant in Tropical Agriculture, 7 Richmond Dale, Clifton, Bristol BS8 2UB, UK.
[4] Shell Global Solutions Sdn. Bhd. (342714-T), Level 20, Tower 2, Petronas Twin Towers, 50088 Kuala Lumpur, Malaysia
[5] Safety and Environmental Assurance Centre, Unilever R&D, Colworth Science Park, Sharnbrook, Bedfordshire, MK44 1LQ, UK
[6] Indonesian Forestry Research Center
[7] Asian Agri Group, R&D Centre, PO Box 35, Kebun Bahilang' Tebing Tinggi Deli 20600, North Sumatera, Indonesia
[*] Corresponding author. E-mail: *cecile.bessou@cirad.fr*

ABSTRACT

The Roundtable on Sustainable Palm Oil (RSPO) is a non-profit association promoting sustainable palm oil through a voluntary certification scheme. Two successive science-based working groups on greenhouse gas (GHG) have been active in RSPO between 2009-2011, with the aim of identifying ways leading to meaningful and verifiable reduction of GHG emissions. One of the outputs is PalmGHG, a GHG calculator using the LCA approach to quantify the major sources of emission and sequestration for a mill and its supply base. A pilot study was carried out in 2011 on nine RSPO companies. Results gave an average of 1.03 t CO_2e/t crude palm oil, with a wide range of -0.07 to +2.46 t CO_2e/t CPO. Previous land use and area under peat were the main causes of the variation. Further modifications to PalmGHG are being made, notably to amend default values and upgrade it to a user-friendly software.

Keywords: palm oil, biodiesel, GHG, calculator, RSPO, PalmGHG

1. Introduction

Nowadays, palm oil is the most used vegetable oil worldwide, representing more than 30% of total produced vegetable oils by mass (Omont, 2010). About 10 to 15% of global production is certified by RSPO (USDA, 2011; RSPO, 2011). RSPO is a non-profit association registered in 2004. It promotes the production and consumption of sustainable palm oil through a voluntary certification scheme. For the growers, this scheme relies on the compliance with 39 principles and criteria (P&Cs) of sustainability that were defined by consensus in 2007. During 2009-2011, the RSPO Executive Board (EB) has commissioned a science-based working group on greenhouse gas (GHG WG) with the aim of identifying ways leading to meaningful and verifiable reduction of GHG emissions. One of the outputs is PalmGHG, a greenhouse gas calculator that allows producers calculate the GHG balances of oil palm products. PalmGHG was developed by the GHG WG as an excel spreadsheet using the LCA approach and based on a previous tool by Chase & Henson (2010). PalmGHG quantifies the major sources of emission and sequestration for a palm oil mill and its supply base, and is compatible with standard international GHG accounting methodologies. It allows for identification of principal emission sources for management purposes; regular reporting, and monitoring. This paper presents the scientific background of PalmGHG Beta version (of April 2012) calculation as well as results from a pilot study carried out in 2011 on nine RSPO companies.

2. Methods

2.1. PalmGHG approach and boundaries

The PalmGHG calculator provides an estimate of the net GHG emissions produced during the palm oil and palm biodiesel production chains. Following the IPCC guidelines (2006), the GHGs considered are CO_2, N_2O, and CH_4, with 100-year timeframe conversion factors of N_2O and CH_4 into CO_2 equivalents (CO_2e) (IPCC, 2007). The conversion factor for biogenic CH_4 is calculated from the ratio of the molecular weights of CO_2 and CH_4 to account for the released CO_2 originating from photosynthesis fixation; i.e. a global warming potential of 22.25 kg CO_2e/kg CH_4 (Wicke et al., 2008). The calculator is based on an attributional LCA approach, i.e. the impacts are those linked to the production unit without considering marginal impacts on other productions or any feedback mechanisms, and without including indirect land use changes.

The emission sources included in the calculator are: i) Land clearing; ii) Manufacture and transport of fertilisers; iii) N_2O and CO_2 resulting from the field application of fertilisers and mill by-products; iv) Fossil fuel used in the field, mainly for harvesting and collection of Fresh Fruit Bunches (FFB); v) Fossil fuel used at the mill; vi) CH_4 produced from palm oil mill effluent (POME); and vii) N_2O and CO_2 resulting from the cultivation of peat soils. In addition, the following GHG sequestration and credits are also considered: i) CO_2 fixed by oil palm trees, ground cover and plantation litter; ii) CO_2 fixed by biomass in conservation areas (methodology still under development); iii) GHG avoided by the selling of mill energy by-products (electricity sold to the grid; palm kernel shell sold to industrial furnaces; etc.). These ten elements account for the bulk of the GHG emission and sequestration occurring during the oil palm crop cycle (Chase and Henson, 2010). Items that are not included in the budget are the nursery stage, pesticide treatments, fuel used for land clearing, emissions embedded in infrastructures and machines, and the sequestration of carbon in palm products and co-products. These items are generally negligible GHG sources (Schmidt, 2007; European Commission, 2009; Choo et al., 2011). Carbon sequestered in palm products and co-products is short-lived, while the other emissions are small when annualised over the crop cycle. Changes in soil organic matter in mineral soils might be significant in the long term but were not considered due to a lack of consensual and harmonised reliable data.

In the first step, net emissions are calculated as tonnes of CO_2e per hectare. From the yield in FFB and the extraction rates in the mill, results are then calculated per tonne of Crude Palm Oil (CPO) and per tonne of Palm Kernel (PK). Allocation of the net emissions of CO_2e between CPO and PK, then subsequently between Palm Kernel Oil (PKO) and Palm Kernel Expeller (PKE), is carried out according to either the relative masses of these co-products or to their relative energy contents. Mass allocation ratios are setup as default in PalmGHG. Finally, the net emissions of CO_2e are calculated per Mega Joules (MJ) of palm biodiesel including emissions from refinery and further biofuel steps according to the methodology and default coefficients provided by the European Renewable Energy Directive (European Commission, 2009). Biodiesel results are given as GHG emission savings compared to the diesel fossil equivalent.

Provision is made for separate budgets for a mill's own crop (usually produced on estates) and an out-grower crop (such as produced by smallholders). PalmGHG uses the annualised emission and sequestration data to estimate the net GHG balance for the palm products from both own and out-grower crops at an individual mill. Emissions from the biomass cleared at the beginning of the crop cycle are averaged over the cycle. Emissions from the other sources are averaged over the three years up to and including the reporting date, thus simplifying data collection and smoothing out short-term annual fluctuations.

2.2. Land clearing and crop sequestration

The approach used to evaluate the contribution of land clearing to GHG emissions in PalmGHG is to average the emissions over a full crop cycle. The calculator estimates the total emissions occurring each year of new planting, adds them all up, and finally divides by the number of years in the average crop cycle (the default is 25 years or 20 years in the case of biodiesel calculation) to obtain an average emission per ha per year. The crop cycle length is defined by users and can differ between “own crops” and “out-growers”. It also differs between crops on mineral soils and those on peat soil, which are often shorter due to accentuated sensitivity to pest and diseases (Wetlands International, 2010).

Previous land uses and their respective carbon stocks were defined in consultation with the scientific panel of RSPO GHG WG who performed a thorough review of literature data and satellite images to identify land use changes associated with oil palm plantations in Indonesia and Malaysia. Considered carbon stocks include above- and below-ground biomass. Carbon stock values for eight previous land uses apart from oil palm stands are currently available in PalmGHG (logged forest, secondary regrowth forest, shrub, grassland, food crops, coconut, rubber, cocoa under shade). Further previous land uses should be implemented soon. However, within the framework of RSPO P&Cs, land use change after 2005 from primary forest to palm plantation will not be allowed. Emissions arising from land clearing are calculated based on measured carbon contents or in their absence an assumed carbon content of 45% in the biomass of the previous land use.

Data for carbon sequestration in the vegetation stand can be obtained from different sources. Field measurements may often be the most relevant data, should they be available and representative of a whole plantation cycle. Where the resources for obtaining these measurements are not available, modelled data may be used instead. Data from OPRODSIM and OPCABSIM models (Henson, 2005, 2009) are used as defaults in PalmGHG to calculate oil palm carbon stock depending on the crop cycle length. These models produce annual values of standing biomass for the oil palms (above and below-ground), ground cover, frond piles and other litter. Field observations revealed that biomass growth and yields are generally lower in the case of out-growers (Chase & Henson, 2010). To reflect this difference, contrasting simulation scenarios of crop sequestration are used as default estimates for mill own crops and out-growers: a ‘vigorous growth’ simulation is used for own crops, and an ‘average growth’ simulation is used for out-growers.

2.3 Emissions due to fertiliser use and field operations

Emissions due to fertilisers contribute significantly to total agricultural GHG emissions and so affect the final GHG balance of palm oil (Yusoff et Hansen, 2007; Pleanjai et al., 2009a; Arvidsson et al., 2011; Choo et al., 2011). Therefore, they have been accorded special attention in PalmGHG. Provision is given for nine widely used synthetic fertilisers and two organic ones (Empty Fruit Bunches (EFB) and POME).

For synthetic fertilisers, emissions consist of i) indirect upstream emissions due to their manufacture and transport from production sites to the mill; ii) direct field emissions linked to physical and microbial processes in the soil, and iii) indirect field emissions following re-deposition of previous direct field emissions. Emissions during fertiliser production vary with the type of product from 44 to 2,380 kg CO_2e/t fertiliser (Jensson and Kongshau, 2003). N_2O direct and indirect field emissions, as well as CO_2 emissions from urea application, are calculated according to IPCC Tier 1 (IPCC, 2006).

Emissions due to EFB and POME production are already accounted for intrinsically within the supply chain assessment. The amounts of EFB and POME are calculated using the following factors: 0.5 t POME/t FFB (Yacob et al., 2006), and 0.22 t EFB/t FFB (Gurmit, 1995). Direct and indirect field N_2O emissions are calculated according to IPCC Tier 1 based on their N content of 0.32% for EFB and 0.045% for POME (Gurmit, 1995). The amounts of EFB and POME, as well as their N contents can be substituted using on-site measurements if these are available.

Emissions due to field operations arise from fossil fuel consumed for transport and other field operations, based on the emission factor 3.13 kg CO_2e/L diesel (JEC, 2007). Total field fuel used encompasses the fuel used for the transport of workers (when managed by the mill) and materials, including the transport and spreading of fertilisers, the transport of FFB from the growing areas to the mill, and maintenance of field infrastructure. Data on fuel use is usually not disaggregated at mill level.

2.4 Emissions due to peat cultivation

Emissions from peat cultivation include CO_2 emissions due to the oxidation of organic carbon and associated N_2O emissions. Both involve enhanced microbial activity. RSPO GHG WG intensively reviewed the impacts of peat cultivation on GHG emissions and identified best management practices for oil palm cultivation on peat soils. In their findings, the authors put emphasis on the importance of managing the water table depth to limit CO_2 emissions from peat land. CO_2 emissions due to peat cultivation are hence calculated using the equation (Eq. 1) according to RSPO GHG WG (F. Agus, pers. com. 2012). Peat CO_2 emissions will vary depending on water table management and this is allowed for in PalmGHG.

$$\text{Peat } CO_2 \text{ emission (t } CO_2\text{/ha/year)} = 0.7 \times 0.91 \times \text{Drainage depth (cm)} \qquad \text{Eq. 1}$$

For N_2O emissions from peat soils, data relating emissions to drainage depth are presently inadequate. Therefore, the IPCC Tier 1 emission factor is used as a default, i.e. 16 kg N-N_2O/ha/yr (IPCC, 2006). Research is still ongoing to better determine the magnitude of peat emissions and how they are affected by and related to factors such as drainage depth, peat subsidence and plantation age.

2.5 Emissions due to oil extraction and transesterification

At the mill level, two main sources of GHG emissions are recorded, fossil fuel consumption and CH_4 emission from POME. Fuel emissions are calculated using the conversion factor of 3.13 kg CO_2e/L diesel (JEC, 2007). Diesel use is usually limited and mostly use to start the machines (Pleanjai et al., 2009a).

CH_4 emissions from POME vary according to the type of treatment. The amount of CH_4 produced per unit of POME is 12.36 kg CH_4/t POME (Yacob et al., 2005). This is the amount released by untreated POME, but options are provided for the capture of CH_4 which is then either flared or used as a fuel to generate electricity. Calculations of CH_4 production and amounts and losses during digestion, flaring, or electricity production are based on factors from Schmidt (2007) and the Environment Agency (2002). When CH_4 is flared and converted to CO_2 these emissions are not accounted for because of their biogenic origin, except for a small fraction of CH_4 that escapes conversion. When CH_4 is used to generate electricity then the amount of substituted electricity is calculated based on an energy content of 45.1 MJ/kg CH_4 (JEC, 2007). The corresponding emissions avoided by the use of the electricity are calculated using the average emission factor for Indonesia and Malaysia (RFA, 2008). A further option is given to the user in case excess palm kernel shell is sold as substitute for coal in industrial furnaces (pers. com. L. Milà i Canals, 2011).

The GHG calculation in PalmGHG was completed with excel spreadsheets from the BioGrace calculator in order to enable GHG calculation up to palm biodiesel output (BioGrace, 2010). The user does not need to provide further data apart from field and mill data.

3. Results of PalmGHG pilot

3.1. The pilot process

A pilot study was carried out in 2011 on nine RSPO companies, to determine the ease of use, and suitability of PalmGHG as a management tool. In June 2011, a preliminary questionnaire was sent to correspondents from the pilot companies. This questionnaire was the starting point of correspondences between these companies and the authors, who were responsible for guiding company correspondents with the use of PalmGHG. Mail exchanges, as well as field visit, allowed for the compilation of input data and calculation of GHG balances.

3.2. Pilot results

Results from eight mills are presented in this paper (Table 1). The average GHG balance is 1.03 t CO_2e/t CPO, with a wide range from -0.07 to +2.46t CO_2e/t CPO. Previous land use and the percentage of the area under peat were the main causes of the variation. Main emission hot spots are land clearing, peat cultivation, and CH_4 from POME. Emissions from N-fertiliser production and N-related field emissions also are an important source of GHG. For the mill C1 (Table 1), main contributors for the mill's own crop plantations are peat emissions (43%), CH_4 from POME (28%), land clearing emissions (14%) and N_2O field emissions (8%). For the same mill, main contributors for the out-grower plantations are CH_4 from POME (52%), land clearing emissions (26%), and N_2O field emissions (12%). In this case, the absence of peat area in out-grower's plantation makes a clear difference between two cropping systems supplying the same mill.

Table 1. Pilot mills, their main characteristics and GHG balances assessed with PalmGHG

Mills	**Mean yield t FFB/ha**	**Out-growers included**	**Peat soil proportions (own-growers only)**	**Previous land uses**	**t CO_2e/t CPO**
A1	23	no	0%	Shrub	0.05
A2	24	no	0%	Shrub	-0.07
B	26	no	0%	Cocoa, oil palm	0.79
C1	23	yes	25%	Grassland, shrub	0.73
C2	19	yes	80%	Grassland, shrub	2.46
F	19	no	0%	Logged forest, oil palm	1.85
G	26	yes	0%	Range from logged forest to food crops	1.15
H	17	yes	0%	Logged forest	1.35

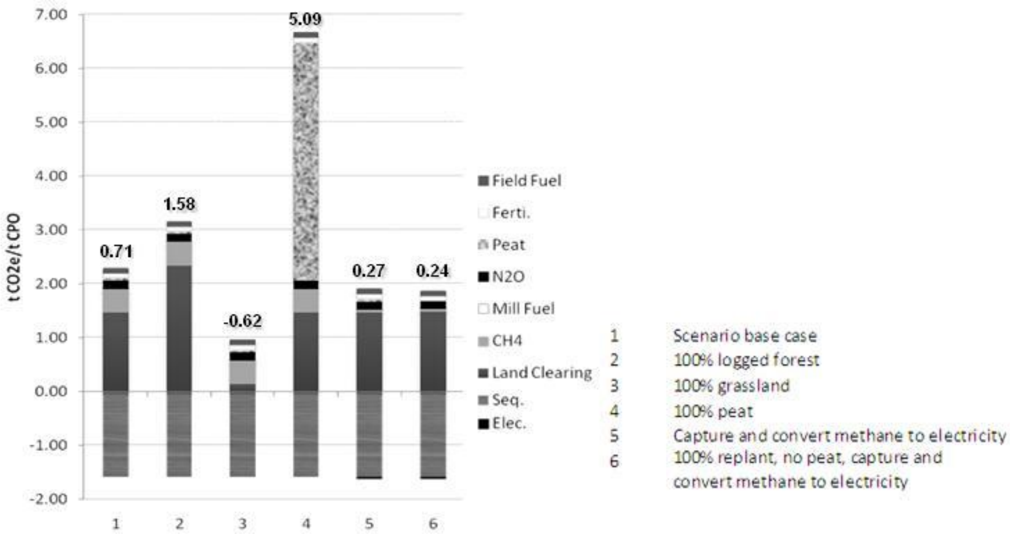


Figure 1. Scenario testing with PalmGHG: Base case (1) = mixed previous land uses, peat 3%, no POME treatment, OER 20%, mill 's own crop mean yield 20.2 t FFB/ha, out-growers' mean yield 14.2 t FFB/ha

PalmGHG readily allows manipulation of input data to test management interventions. Results of scenario testing are given for a set of dummy data for a base case (scenario 1 in Figure 1). The results show that high emissions result from clearing logged forest and peat cultivation, and conversely that very low (negative) emissions result from clearing low biomass land such as grass land. Fertiliser emissions are a non negligible contributor especially in scenario 3, where net sequestration (sequestration less land clearing emissions) is high, or in scenarios 5 and 6, where net sequestration is almost null and CH_4 is captured. The contribution of mill fuel is negligible and not visible on the graph. Net emissions below 0.5 t CO2e/t CPO can be obtained from a mature industry that is replanting palms and capturing and generating electricity from captured CH_4 (Fig. 1). This was highlighted in the recommendations to the RSPO EB.

4. Discussion

GHG balances calculated with PalmGHG are within the range of those found in the literature. However, depending on the system boundaries and particularly on assumptions regarding land clearing and peat emissions, GHG balances greatly vary around 2.3 t CO_2e/t CPO (Schmidt, 2007), 0.6-1 t CO_2e/t CPO (Siangjaeo et al., 2011), or 2.8-19.8 t CO_2e/t CPO (Reijnders et Huijbregts, 2008). Carbon stocks and peat emissions are notably very sensitive parameters. Research efforts are still needed to better quantify carbon stocks and the impacts of agricultural practices on these stocks, especially in the case of peat cultivation. PalmGHG should be updated regularly to introduce newly harmonised carbon stocks for diverse land uses with added impacts on soil organic contents, and to better model the emissions due to peat cultivation or restoration. This is of paramount importance in Southeast Asia where peat land area accounts for 57% of total tropical peat area, i.e. 10-14% of global peat carbon pool, mostly located in Indonesia and Malaysia (Page et al., 2011).

Integrating the spatial and temporal dimensions of the palm perennial crop cycle within a snapshot assessment is not immediate. In PalmGHG, this difficulty is somehow by-passed by embracing data from several plantations units from mill's own crop and out-growers at several ages. Despite large considered areas, ages of oil palms may however not be evenly distributed inducing some bias by displacing age distribution. In particular, plantation with short turn-over may displace the distribution towards young palm trees that sequester carbon more quickly.

Across the published studies, the relative importance of the diverse contributors is in agreement. Land clearing is the most important contributor together with peat emissions (Germer and Sauerborn, 2008; Reijnders and Huijbregts, 2008; Wicke et al., 2008). Some studies that do not directly address this issue still mention the primary importance of this contributor (Yusoff et Hansen, 2007; Pleanjai et al., 2009; Stichnothe et Schuchardt, 2011). In all studies also CH_4 from POME emissions and fertiliser production and use are important contributors (Choo et al., 2011; Pleanjai et al., 2009; Siangjaeo et al., 2011), although their relative total importance depends on whether land use change and peat emissions are included or not. As shown in PalmGHG scenario testing, it is often emphasised that CH_4 capture can allow for significant GHG reductions, between 30 to 50% (Vijaya et al., 2008; Chuchuoy et al., 2009).A wide range of studies focused on treatment and uses of residues and co-products (Yacob et al., 2005; Chavalparit et al., 2006; Vijaya et al., 2008; Stichnothe et Schuchardt, 2011). However, emphasis should be put on the high costs and limited options in the field to actually implement the technologies to harness the best benefits from residues, notably when grid connection is not possible. Such technologies can be implemented through clean development mechanisms provided that attention is paid to avoid double-counting of GHG savings, such as credits for coal substitution by shells both at the palm oil mill and cement factory for instance. Moreover, research effort is also needed notably to better assess fertilising efficiency of land filled residues and environmental emissions of down-stream processes related to residues treatment and transport.

The GHG balance only is one potential impact on the environment. PalmGHG is a very useful tool that can help demonstrate potentials for GHG savings at the plantation and mill levels. Together with the other RSPO P&Cs that define a broader view for sustainability criteria, it can help improve oil palm production towards sustainability. However, more complete LCA must also be considered to quantify other impacts such as eutrophication or toxicities for instance. In this case, other stages of palm oil production might also play an important role such as pesticides for ecosystem toxicity or boiler emissions for human toxicity (Schmidt, 2007; Choo et al., 2011; Bessou et al., 2012). Compared to other vegetable oils, palm oil usually performs better due to high yields (5-17 t CO_2e/t Rapeseed oil *In* Schmidt, 2007; 39-88 g CO_2e/MJ Palm Methyl Ester compared to 62 and 124-159 g CO_2e/MJ of Rapeseed Methyl Ester and *Jatropha* Methyl Ester; respectively *In* Thamsiriroj and Murphy, 2009; Achten et al., 2010a,b), but comparison on a unique criterion may induced trade-offs in environmental impacts. In particular, consideration of impacts on soil fertility and biodiversity is paramount. In this case, a more comprehensive LCA approach is needed, such as in Milà i Canals et al., (2012), to allow for a sound and harmonised comparison between agricultural products considering land transformation and land occupation compared to restored vegetation stands or other common reference land uses.

5. Conclusion

PalmGHG is a comprehensive GHG calculator representative of the state of the art in terms of available data and international methodologies for GHG accounting. Emphasis has been placed on information directly relevant to palm oil production that should be easily available at the mill level. However, default data are also provided for data which might not be available. Flexibility is also an important feature of PalmGHG, with options that allow for alternative calculations and methodology; the main example being assessment of net emissions per MJ for palm oil biodiesel.

During pilot testing it was shown that PalmGHG can identify GHG emission ‘hot spots’, and so help to define GHG reduction strategies. Feedback from the pilot companies highlighted problems in collecting data, especially those for for three consequent years. It should however, be noted that difficulties related to data recording should progressively diminish once the monitoring of GHG emissions becomes routine. On the other hand, difficulties encountered when collecting data for out-growers are not so easily resolved and indicate a need for a specific strategy to help out-growers record and collect data on a routine basis.

The results of the pilot and scenario testing provided an important information basis to design some of the recommendations to RSPO EB and communicate to a large audience on the work of RSPO GHG WG and the use of PalmGHG. Further recommendations of the GHG WG to RSPO EB refer e.g. to the characteristics that should be met by new plantations in order to ensure low GHG emissions.

Further modifications to PalmGHG are still being made, notably to amend default values. Moreover, PalmGHG needs reprogramming to make it more user-friendly. The current spreadsheet is rather complex and not easy to follow. Software would allow users to quickly generate results, but at the same time provide means to readily change default parameters and undertake tests of alternative scenarios.

6. Acknowledgements

The authors are very grateful to all who participated in the second RSPO GHG WG for their contributions and to the companies who took part in the pilot phase. We are also grateful to members of the RSPO EB for their continuing support; particular thanks are due to Dr Simon Lord for proposing the adoption of the GWAPP model and to Dr Timothy Killeen for guiding the development of PalmGHG.

7. References

Achten, W. M. J., Almeida, J., Fobelets, V., Bolle, E., Mathijs, E., Singh, V. P., Tewari, D. N., Verchot, L. V., Muys, B., 2010a. Life cycle assessment of Jatropha biodiesel as transportation fuel in rural India. Appl. Energ. 87, (12) 3652-3660.

Achten, W. M. J., Vandenbempt, P., Almeida, J., Mathijs, E., Muys, B., 2010b. Life Cycle Assessment of a Palm Oil System with Simultaneous Production of Biodiesel and Cooking Oil in Cameroon. Environ. Sci. Technol. 44, (12) 4809-4815.

Arvidsson, R., Persson, S., Fröling, M., Svanström, M., 2011. Life cycle assessment of hydrotreated vegetable oil from rape, oil palm and Jatropha. J. Cleaner Prod. 19, (2-3) 129-137.

Bessou, C., Vélu, A., Caliman, J.P. 2012. LCA of Palm Oil in Sumatra, Comparison of Cropping Systems. ICOPE Conference, 22-24 February 2012, Nusa Dua, Bali, Indonesia

BioGrace, 2010. European Project on “Harmonised Calculations of Biofuel Greenhouse Gas Emissions in Europe”; Project coordinator: John Neeft - Agentschap NL (Agency NL) (formerly SenterNovem). http://www.biograce.net./content/ghgcalculationtools/overview

Chase, L.D.C. Henson, I.E., 2010. A detailed greenhouse gas budget for palm oil production. Int. J. Agric. Sustain. 8(3), 199-214.

Chavalparit, O., Rulkens, W. H., Mol, A. P. J., Khaodhair, S., 2006. Options for Environmental Sustainability of the Crude Palm Oil Industry in Thailand through enhancement of Industrial Ecosystems. Environ. Dev. Sustain. 8, (2) 271-287.

Choo, Y. M., Muhamad, H., Hashim, Z., Subramaniam, V., Puah, C. W., Tan, Y., 2011. Determination of GHG contributions by subsystems in the oil palm supply chain using the LCA approach. Int. J. Life Cycle Assess. 16, 669-681.

Chuchuoy, K., Paengjuntuek, W., Usubharatana, P., Phungrassami, H., 2009. Preliminary Study of Thailand Carbon Reduction Label: A Case Study of Crude Palm Oil Production. Eur. J. Sci. Res. 34, (2) 252–259.

Environment Agency, 2002. Guidance on Landfill Gas Flaring. Bristol: Environment Agency.

European Commission, 2009. Directive 2009/28/EC of the European parliament and of the council of 23 April 2009 on the promotion of the use of energy from renewable sources. Official Journal of the European Union, June 5th

Germer, J., Sauerborn, J., 2008. Estimation of the impact of oil palm plantation establishment on greenhouse gas balance. Environ. Dev. Sustain. 10, (6) 697-716.

Gurmit S., 1995. Management and utilisation of oil palm by-products. The Planter. (71), 361-386.

Henson I.E., 2005. OPRODSIM, a versatile, mechanistic simulation model of oil palm dry matter production and yield. In: Proceedings of PIPOC 2005 International Palm Oil Congress, Agriculture, Biotechnology and Sustainability Conference, 801-832. Kuala Lumpur: Malaysian Palm Oil Board.

Henson I.E., 2009. Modelling carbon sequestration and greenhouse gas emissions associated with oil palm cultivation and land-use change in Malaysia. A re-evaluation and a computer model. MPOB Technology, 31, 116 pp.

IPCC, 2006. Guidelines for National Greenhouse Gas Inventories. Vol 4 Agriculture, Forestry and Other Land Use. WMO/UNEP. http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html.

IPCC, 2007. Fourth Assessment Report. Climate Change 2007 - Synthesis Report. WMO/UNEP. http://www.ipcc.ch/ipccreports/ar4-syr.htm.

JEC, 2007. Well-to-wheels analysis of future automotive fuels and powertrains in the European context. Well-to-tank Report Version 2c, Appendix 1 and 2. JRC, EUCAR, CONCAWE. http://ies.jrc.ec.europa.eu/WTW.

Milà i Canals, L., Rigarlsford, G., Sim, S., 2012. Land use impact assessment of margarine. Int. J. Life Cycle Assess.

Omont H., 2010. Contributions de la production d'huile de palme au développement durable. Problématique générale, controverses. Ol. Corps Gras Li. 17 (6), 362-367.

Page, S., Rieley, J. O., Banks, C. J., 2011. Global and regional importance of the tropical peatland carbon pool. Global Change Biology 17, (2) 798-818.

Pleanjai, S., Gheewala, S. H., Garivait, S., 2009. Greenhouse gas emissions from the production and use of palm methyl ester in Thailand. Int. J. Glob. Warming. 1, (4) 418–431.

Reijnders, L., Huijbregts, M. A. J., 2008. Palm oil and the emission of carbon-based greenhouse gases. J. Cleaner Prod. 16, (4) 477-482.

RFA (2008) Carbon and Sustainability Reporting Within the Renewable Transport Fuel Obligation. Technical Guidance Part 2 Carbon Reporting – Default Values and Fuel Chains. London: Renewable Fuels Agency. http://www.renewablefuelsagency.org/_db/_documents/RFA_ C&S_ Technical_Guidance_Part _2 _v1_200809194658.pdf

RSPO, 2011. updated data on the webpage, visited on 10/02/2012: www.rspo.org

Siangjaeo, S., Gheewala, S. H., Unnanon, K., Chidthaisong, A., 2011. Implications of land use change on the life cycle greenhouse gas emissions from palm biodiesel production in Thailand. Energ. Sust. Dev. 15, (1) 1-7.

Stichnothe, H., Schuchardt, F., 2011. Life cycle assessment of two palm oil production systems. Biomass Bioenerg. 35, (9) 3976-3984.

Thamsiriroj, T., Murphy, J. D., 2009. Is it better to import palm oil from Thailand to produce biodiesel in Ireland than to produce biodiesel from indigenous Irish rape seed? Appl. Energ. 86, (5) 595-604.

USDA, 2011. USDA report of Feb. 2011

Vijaya, S., Ma, A. N., Choo, Y. M. et Nik, M. N. S., 2008. Life cycle inventory of the production of crude palm oil - A gate to gate case study of 12 palm oil mills. J. Oil Palm Research. 20, 484-494.

Wetlands International, 2010. A quick scan of peatlands in Malaysia. Wetlands International-Malaysia: Petaling Jaya, Malaysia. 74 pp.

Wicke, B., Dornburg, V., Junginger, M., Faaij, A., 2008. Different palm oil production systems for energy purposes and their greenhouse gas implications. Biomass Bioenerg. 32, (12) 1322-1337.

Yacob, S., Hassan, M. A., Shirai, Y., Wakisaka, M., Subash, S., 2005. Baseline study of methane emission from open digesting tanks of palm oil mill effluent treatment. Chemosphere 59. (11) 1575-1581.

Yusoff, S. et Hansen, S., 2007. Feasibility Study of Performing a Life Cycle Assessment on Crude Palm Oil Production in Malaysia (9 pp). Int. J. Life Cycle Assess. 12, (1) 50-58.

Review and future perspectives in the environmental assessment of seafood production systems

Ian Vázquez-Rowe[1,2,*], Almudena Hospido[1], María Teresa Moreira[1], Gumersindo Feijoo[1]

[1] Department of Chemical Engineering, School of Engineering, University of Santiago de Compostela. 15782 – Santiago de Compostela, Spain.
[2] CRP Henri Tudor/CRTE, 66 rue de Luxembourg, Esch-sur-Alzette, Luxembourg (current adress).
[*] Corresponding author. E-mail: *ian.vazquez@tudor.lu*

ABSTRACT

A set of 29 published articles or reports were analysed with the aim of determining current trends in the use of Life Cycle Assessment (LCA) in the fishing sector in terms of novel methodological innovations and the treatment of assumptions when elaborating LCAs linked to this production sector. Results showed an increasing use of LCAs in the seafood sector in recent years, but variable levels of methodological development. Based on the evaluation, a simple guideline to set common denominators in future LCA studies are proposed in terms of goal and scope, life cycle inventory or impact category selection. Finally, innovative issues that could turn LCA into a more integrative methodology within fishing systems, in order to enhance its usefulness in policy support, are briefly discussed.

Keywords: carbon footprint, fisheries, food systems, LCA, seafood

1. Introduction

Environmental impacts linked to seafood production systems have centred numerous studies in recent decades due to increasing fragility regarding the state of world fisheries (Worm et al., 2009). In this context, Life Cycle Assessment (LCA) has arisen as a useful environmental management methodology, the only one that is standardized at an international level, to quantify resource use and emissions in a broad set of primary and industrial sectors, including seafood extraction (fishing) or production (aquaculture) and their associated industrial processes (Pelletier et al., 2007).

Nevertheless, despite the usefulness of LCA in the fishing sector, it still lacks the comprehensiveness required to evaluate certain environmental impacts linked to fishing systems, such as direct impacts on fishing stocks (Pelletier et al., 2007).

Therefore, the main aim of this study is to analyse the milestones that have been developed in fisheries LCA in the last few years, comprising discussion concerning methodological developments, functional unit (FU) choice, impact category selection, allocation or life cycle inventory (LCI). Moreover, based on the previous analysis, a simple and straightforward guideline is proposed to be used as a common denominator in future LCA studies. Finally, future perspectives regarding methodological innovations in LCA are discussed.

2. Methods

A set of 29 LCA and carbon footprint (CF) studies were obtained from scientific journals, scientific reports, and chapters from books or contributions to congresses. The selected publications were divided into the following subdivisions: (i) fishery LCA studies; (ii) seafood processing LCA studies; (iii) food meal LCAs that include products of marine origin; and (iv) seafood carbon footprint (CF) studies. For more information on the particular studies revised please check Vazquez et al., 2012c.

3. State-of-the-art of fishery LCA studies

3.1. Worldwide fishery LCA studies

These studies constitute the largest, heterogeneous and prolific group of case studies included in this review. In contrast with pioneering studies in Scandinavian countries, mainly for cod fisheries (Eyjolfsdottir et al., 2003; Ziegler et al., 2003), recent studies show deep diversification of evaluated species, gears and fisheries. However, most studies are still concentrated in fishing fleets belonging to industrialised nations with important seafood landings, such as Norway, Spain or Canada.

Concerning fishing gears, studies relating to trawls and purse seines have continued to be performed (Ramos et al., 2011; Ziegler et al., 2011), but novel gears such as long lines(Svanes et al., 2011a; Vázquez-Rowe et al., 2011) and creels have also been analysed from a life cycle perspective. Artisanal fleets and gears, however, despite their relevance on a global scale due to the number of vessels and workers involved in the subsector, have rarely been examined (Ziegler et al., 2011).

Finally, species analysis has also seen proliferation in recent years, since small pelagic fish, such as mackerel or pilchard (Ramos et al., 2011a; Vázquez-Rowe et al., 2010b), crustaceans, such as shrimps or lobster (Ziegler and Valentinsson, 2008; Ziegler et al., 2011), cephalopods (Vázquez-Rowe et al., 2012a) or gadoids other than cod, such as hake or pollock (Sund, 2009; Vázquez-Rowe et al., 2011), have been evaluated using LCA methodology.

3.2. Seafood processing

There is a limited number of research articles linked to this phase of the life cycle of seafood products. Moreover, the degree of complexity of final products, as consumed in households can be very varied. Consequently, based on these two issues, available studies can be still considered pioneering projects. In the first place, fresh fish consumption supply chains have been assessed by Vázquez-Rowe et al., (2011), showing the low relative impacts of the on land phases. Secondly, frozen seafood products were initially examined for Danish seafood products (cod, shrimps…), but recent studies also include analysis of frozen cod in Scandinavian countries and Canada (Fulton, 2010; Ziegler et al., 2003) and frozen cephalopods (Vázquez-Rowe et al., 2012a). Thirdly, the canning industry linked to the seafood sector has been assessed by Hospido et al., (2006), which highlighted the high energy demand of the canning factory, as well as the elevated environmental burdens linked to tinplate transport and production. Finally, the elaboration of complex multi-ingredient breaded seafood products has been assessed by Thrane (2004) and Fikseaunet (2007). These two studies noted the importance of catching low energy-intensive fish and on optimizing the fish content of these products in order to lower their environmental burdens.

3.3. Food meal LCAs

Despite of the reduced number of seafood LCA studies currently available in the literature, a set of publications dealing with the environmental impacts of diets and meals have recently included seafood products arriving from capture fisheries. In the first place, Muñoz et al., (2010), when analysing the environmental profile of the average Spanish diet, included available seafood products that were at the time in the literature. Nilsson and Sonesson (2010), in a study of similar characteristics for the Swedish diet, detected that 0.17 million tonnes of CO_2eq./year would be avoided if Swedes were to consume 14% fish, that is, the recommended annual amount. Finally, other studies, such as Zufía and Arana (2008) and Espinosa-Orias and Azapagic (2010) performed specific environmental impact studies of different specific meals.

3.4. Carbon footprinting

The use of CF methodology in fisheries LCA was developed initially by Winther et al., (2009), through the analysis of 14 different iconic Norwegian seafood products that are exported to other nations. Iribarren et al., (2010; 2011) evaluated the CF of the Galician fishing sector, providing extraction CF values for over 50 commonly landed fish species. While the former used mass allocation to report the results; the Galician study selected an economic perspective following the PAS2050 specifications. The relevance of these CF studies, however, must be understood in a wide framework, in which GHG emissions may not constitute the main environmental burden. Nevertheless, the use of CF as a single indicator to reach consumers and stakeholders has already been proved in other production sectors. Finally, it is important to highlight the fact that certain fishery management decisions have been proved to cause important changes in terms of GHG emissions in fisheries (Driscoll and Tyedmers, 2010).

4. Key methodological issues in fisheries LCAs

An overwhelming majority of the evaluated publications assume the general ISO guidelines to compute LCA in fisheries and food products (ISO, 2006). Nevertheless, important differences regarding the LCA approach can be seen depending on the nature of the analysed fisheries or the consumption of seafood trends between nations.

4.1. Methodological assumptions in LCA

A wide range of variance can be seen when assessing fishery and seafood case studies, depending on the assumptions that are taken into consideration:

- Attributional or consequential LCA perspective. All, but one of the assessed studies have adopted a descriptive, and therefore, current state-of-the-art approach, using the attributional perspective. The use of consequential assessment by Thrane (2004) was aimed to detect the environmental changes linked to certain predicted decisions. While the minimal use of consequential LCA in fishing systems may be linked to the lack of data availability regarding marginal production systems, its use combined with stock prediction techniques may constitute an important milestone in policy making.
- Functional unit and system boundaries. The selection of the functional unit (FU) shows great differences between studies, suggesting that the nature of the project is the main factor that influences the reference unit. Nevertheless, two different tendencies were observed. On the one hand, those studies that limited their system boundaries to the fishing stage used FUs that refer to bulk landings at port (Hospido and Tyedmers, 2005) or intermediate supply chain packaging units (Ziegler et al., 2011). On the other hand, studies that focused on the entire supply chain presented highly specific FUs, linked to final package presentations (Zufía and Arana, 2008) or to standard consumption portions (Vázquez-Rowe et al., 2011).
- Allocation procedure. Allocation constitutes a key feature in fisheries LCA, affecting mainly the fishing stage due to the multispecies characteristics of many fisheries. Furthermore, debate around allocation has gradually increased given its strong influence on final results (Pelletier and Tyedmers, 2011). Despite the extended use of either mass or economic allocation in the majority of evaluated studies, recent studies show increasingly critical visions regarding these two options. For instance, several studies have shown their disagreement with the fact that using an economic allocation creates a situation in which low value species show reduced environmental burdens with respect to other species in a unique biophysical system (Pelletier and Tyedmers, 2011). As a result, some recent studies propose new allocation perspectives, such as energy content (Svanes et al., 2011b). Nevertheless, the use of multiple allocation approaches in different studies may lead to atomisation; therefore, the need for well-discussed allocation explanations is needed to guarantee reproducibility and transparency (Ayer et al., 2007).
- Assessment methods and impact categories selection. The greater proportion of evaluated case studies used the CML Baseline 2000 assessment method (Frischknecht et al., 2007), constituting a midpoint approach to the results. Concerning the selection of impact categories, an increase in the number of utilised categories has been identified, including ozone layer depletion or toxicity categories and newly developed fishery-specific categories (non-standardized).
- Result interpretation. The majority of the publications evaluated limit the result reporting to the inventory data and the characterisation phase (Sund, 2009; Vázquez-Rowe et al., 2012a; Ziegler et al., 2011). Regarding sensitivity analysis, its use has not been an extended practice in most publications.

4.2. Methodological advances

A selection of outstanding methodological innovations in the field of fisheries LCA is listed below:

- New impact categories. Innovations regarding the introduction of novel fishery-specific impact categories have been limited. More specifically, three different biological issues linked to fisheries have been developed. In the first place, the computation of seafloor damage was incorporated by Ziegler et al., (2003) and has gained acceptance ever since. Secondly, the calculation of net primary productivity in LCA studies was introduced in aquaculture studies. However, recent studies have introduced the so called biotic resource use (BRU) indicator (Fulton, 2010; Parker, 2011). Finally, discards, which were initially included in terms of total mass discarded per FU have recently been computed in a global discard index – GDI (Vázquez-Rowe et al., 2012b).
- LCA+DEA. The joint use of LCA with data envelopment analysis (DEA), as proposed by Vázquez-Rowe et al., (2010a) aims at reducing the effect of increased standard deviations, as well as providing additional information for result interpretation. Moreover, the main innovative issue linked to the LCA+DEA method is the inclusion of an operational dimension in environmental management. In fact, LCA+DEA was computed for a set of different fishing fleets to quantify the environmental burdens concerning operational inefficiency in order to define target performance values for inefficient vessels.

- Timeline analysis. This approach in fishery LCA acknowledges the existence of strong fluctuations in the environmental burdens in pelagic fisheries. In fact, Ramos et al., (2011) proved the existence of sharp variations during an 8-year period for the Cantabrian Atlantic mackerel fishery.

5. Best practices in fishing production systems

A set of common ground fundamentals can be seen found in fisheries LCA, regardless of the methodological and non-methodological assumption of each specific case study. Therefore, a best practices protocol is proposed in order to provide a basic inventory guideline for future studies, as well as a mechanism to guarantee data completeness when assessing fisheries, transparency and reproducibility. Nevertheless, the suggested protocol only analyses the fish extraction stage due to the complex supply chains existing once fish is landed, the varied processing techniques that may be used or the limited amount of life cycle studies concerning fish processing.

5.1. Recommendations for goal and scope

Goal and scope decisions are highly dependent on the approach the authors selected based on the specific context of the study. However, it should be highlighted that when an allocation method is needed, the introduction of new biophysical allocation methods, other than mass allocation, may constitute interesting milestones in the near future (Pelletier and Tyedmers, 2011).

5.2. Recommendations for LCI

In order to carry out the environmental assessment a given fishery, the following life cycle inventory (LCI) items, based on data retrieved from the evaluated studies, should allow LCA practitioners to develop a comprehensive LCA study:

Table 1. Minimum required inventory items to perform an integrated LCA study for extractive fisheries.

Item	Characteristics
Diesel production and consumption	Further research on fuel breakdown needed to understand consumption patterns in vessels.
Gear production and use	More research should be put into gear loss and ghost fishing.
Anti-fouling and boat paint	Future research should determine new sources of potential burdens in this subsystem.
Cooling agents	Recently highlighted as an important source of ozone layer and climate change emissions.
Ice production	Deeper analysis linked to fuel use and ice production.
On-board processing	Packaging resources, offal waste and fuel use linked to processing must be explored.
Vessel construction	To date, very simplistic approach. Shi dock system could be analysed in future studies.
Seafloor damage	Currently, based on land covered. Future perspectives: direct impact and quality.
Bait	Further analysis must be done concerning the link between fleets regarding bait.
Captures and landings	Comprehensive data regarding catches, including the landed, discarded, offal and slipped fractions would help improve data quality.

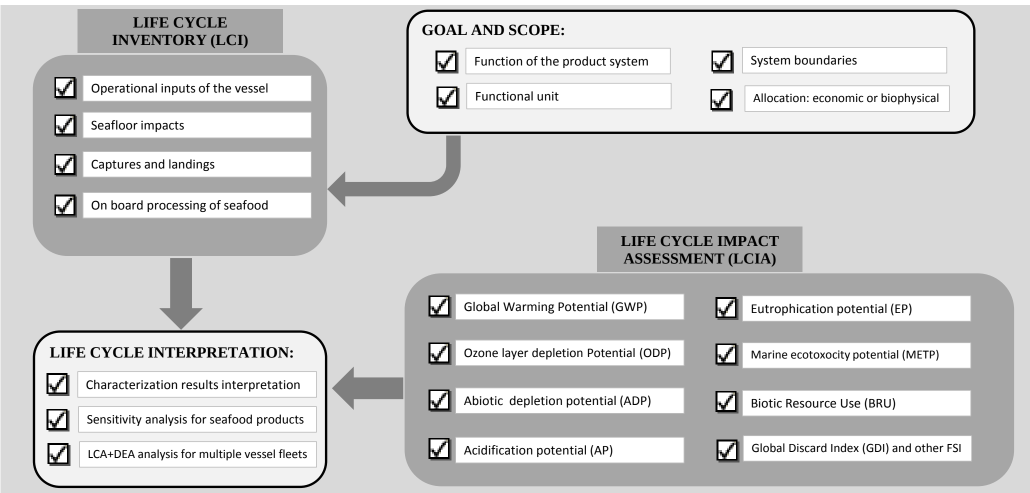


Figure 1. Minimum requirements linked to LCA implementation in fishing systems. Source: Vázquez-Rowe et al., 2012c.

5.3. Recommendations for LCIA

The full use of the inventory data items included in Figure 1 permits a comprehensive assessment of all major impact categories used to date in the available bibliography. Nevertheless, it must be noted that in recent years two different perspectives have developed in fisheries LCA: those publications that focus entirely on the reporting of the CF value of seafood and those that adopt an integrated perspective, combining the commonly used LCA impact categories in fisheries LCA with newly developed fishery-specific impact categories. In fact, certain recommendations suggested by prior fishery LCA reviews (Pelletier et al., 2007) have been accomplished, but others remain largely unanswered. For instance, the use of BRU, as recommended in this publication, has had a limited implementation, and the introduction of new indicators linked to the biological implications of fisheries has been discrete. Moreover, to date no studies in the field of seafood LCA have contributed to the expansion of LCA to Life Cycle Sustainability Assessment (LCSA), and only a small set of studies has considered the introduction of operational and economic dimension in combination with other methodologies (Vázquez-Rowe et al., 2010a).

Regarding the selection of impact categories, these can be included or removed from studies based on the initial objectives of the study, without the need of performing any assumptions. Hence, Figure 1, in an attempt to integrate the different patterns observed in impact category selection, includes a series of recommended impact categories and indicators that should be included in fishery LCA studies, with the aim of guaranteeing quality and transparency in future studies, as well as creating some minimum foundations for future development in this specific field.

6. Conclusions

The race to include fishery-specific impacts on the ecosystem in LCA has involved a sharp increase in seafood LCA publications. However, the level of pioneering in this sector in terms of LCA still remains high, impeding a clear analysis in terms of current trends. However, based on the inventoried publications, the main highlights and developments in this field have been specified and discussed. Finally, the implementation of the best practices suggested in this paper should help the definition of more concrete developments in the field of fisheries LCA in the near future.

7. References

Ayer, N., Tyedmers, P., et al., 2007. Co-product allocation in life cycle assessments of seafood production systems: Review of problems and strategies. Int. J. Life Cycle Ass. 12, 480-487.

Driscoll, J., Tyedmers, P., 2010. Fuel use and greenhouse gas emission implications of fisheries management: the case of the New England Atlantic herring fishery. Mar. Policy 34, 353-359.

Espinoza-Orias, N., Azapagic, A., 2010. Carbon footprint of school meals. VII International Conference on Life Cycle Assessment in the agri-food sector. Bari, Italy, September 22nd-24th 2010.

Eyjolfsdottir, H.R., Jonsdottir, H., et al., 2003. Environmental effects of fish on the consumers dish. Life cycle assessment of Icelandic frozen cod products. Icelandic Fisheries Laboratories. Report 06-03.

Fikseaunet, T.K., 2007. Alternative fish finger production chains – life cycle analysis. Marine Systems M. Sc. Thesis, Norwegian University of Science and Technology.

Frichknecht, R., Jungbluth, N., et al., 2007. Implementation of Life Cycle Impact Assessment Methods: Data v2.0 ecoinvent report No. 3. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Fulton, S., 2010. Fish and fuel: life cycle greenhouse gas emissions associated with Icelandic cod., Alaskan pollock and Alaskan pink salmon fillets delivered to the UK. M. Sc. Thesis Dalhousie University.

Hospido, A., Tyedmers, P., 2005. Life cycle environmental impacts of tuna fisheries. Fish. Res. 76, 174-186.

Hospido, A., Vázquez, M.E., et al., 2006. Environmental assessment of canned tuna manufacture with a life-cycle perspective. Resour. Conserv. Recy. 47, 56-72.

Iribarren, D., Vázquez-Rowe, I., et al., 2010. Estimation of the carbon footprint of the Galician fishing activity (NW Spain). Sci. Total Environ. 408, 5265-5272.

Iribarren, D., Vázquez-Rowe, I., et al., 2011. Updating the carbon footprint of the Galician fishing activity (NW Spain). Sci Total Environ. 409, 1609-1611.

ISO, 2006. ISO 14040. Environmental management – Life Cycle Assessment – Principles and Framework. International Organization for Standardization.

Muñoz, I., Milá i Canals, Ll., et al., 2010. Life cycle assessment of the average Spanish diet including human excretion. Int. J. Life Cycle Ass. 15, 794-805.

Nilsson, K., Sonesson, U., 2010. Changing diets – What is the influence on greenhouse gas (GHG) emissions of different consumption patterns? VII International Conference on Life Cycle Assessment in the agri-food sector. Bari, Italy, September 22nd-24th 2010.

Parker, R., 2011. Measuring and characterizing the ecological footprint and life cycle environmental costa of Antartic krill (*Euphasia superba*) products. M. Sc thesis, Dalhousie Univeristy.

Pelletier, N., Ayer, N., et al., 2007. Impact categories for Life Cycle Assessment research of seafood production systems: review and prospectus. Int. J. Life Cycle Ass. 12, 414-421.

Pelletier, N., Tyedmers, P., 2011. An ecological critique of the use of market information in life cycle assessment research. J. Ind. Ecol. 14, 342-354.

Ramos, S., Vázquez-Rowe, et al., 2011. Environmental assessment of the Atlantic mackerel (*Scomber scombrus*) season in the Basque Country. Increasing the timeline delimitation in fishery LCA studies. Int. J. Life Cycle Ass. 16, 599-610.

Sund, V., 2009. Environmental assessment of Northeast arctic co (*Gadus morhua*) caught by long-lines and Alaska pollock (*Theragra chalcogramma*) caught by pelagic trawls. Carbon footprinting and analysis of biological sustainability of two frozen seafood products

Svanes, E., Vold, M., et al., 2011a. Environmental assessment of cod (*Gadus morhua*) from autoline fisheries. Int. J. Life Cycle Ass. 16, 611-624.

Svanes, E., Vold, M., et al., 2011b. Effect of different allocation methods on LCA results of products from wild-caught fish and on the use of such results. Int. J. Life Cycle Ass. 16, 512-521.

Thrane, M., 2004. Environmental impacts from Danish fish products. Hot spots

Vázquez-Rowe, I., Iribarren, et al., 2010a. Combined application of life cycle assessment and data envelopment analysis as a methodological approach for the assessment of fisheries. Int. J. Life Cycle Ass. 15, 272-283.

Vázquez-Rowe, I., Moreira, M.T., et al., 2010b. Life cycle assessment of horse mackerel fisheries in Galicia (NW Spain): Comparative analysis of two major fishing methods. Fish. Res. 106, 517-527.

Vázquez-Rowe, I., Moreira, M.T., et al., 2011. Life cycle assessment of fresh hake fillets captured by the Galician fleet in the Northern Stock. Fish. Res. 110,128-135.

Vázquez-Rowe, I., Moreira, M.T., 2012a. Environmental assessment of frozen common octopus (*Octopus vulgaris*) captured by Spanish fishing vessels in the Mauritanian EEZ. Mar. Policy 36, 180-188

Vázquez-Rowe, I., Moreira, M.T., et al., 2012b. Inclusion of discard assessment indicators in fisheries LCA studies. Expanding the use of fishery-specific impact categories. Int. J. Life Cycle Ass. 17, 535-549.

Vázquez-Rowe, I., Hospido, A., et al., 2012c. Review: Best practices in Life Cycle Assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems Technol., doi= 10.1016/j.tifs.2012.07.003.

Winther, U., Ziegler, F., et al., 2009. Carbon footprint and energy use of Norwegian seafood products. SINTEF Fisheries and Aquaculture Report SFH80 A096068, Trondheim, Norway.

Worm, B., Hilborn, R., et al., 2009. Rebuiding global fisheries. Science 325, 578-585.

Ziegler, F., Nilsson, P., et al., 2003. Life cycle assessment of frozen cod fillets including fishery-specific environmental impacts. Int. J. Life Cycle Ass. 8, 39-47.

Ziegler, F., Valentinsson, D., 2008. Environmental life cycle assessment of Norway lobster (*Nephrops norvegicus*) caught along the Swedish west coast by creels and conventional trawls – LCA methodology with case study. Int. J. Life Cycle Ass. 13, 487-497.

Ziegler, F., Emanuelsson, A., et al., 2011. Extended life cycle assessment of Southern pink shrimp products originating in Senegalese artisanal and industrial fisheries for export to Europe. J. Ind. Ecol. 15, 527-538.

Zufía, J., Arana, L., 2008. Life cycle assessment to eco-design food products: industrial cooked dish case study. J. Clean. Prod. 16, 1915-1921.

Overfishing, overfishedness and wasted potential yield: new impact categories for biotic resources in LCA

Andreas Emanuelsson[1,2,*], Friederike Ziegler[1], Leif Pihl[2], Mattias Sköld[3], Ulf Sonesson[1]

1SIK – The Swedish Institute for Food and Biotechnology, Department of Sustainable Food Production, 402 29 Gothenburg, Sweden
2University of Gothenburg, Department of Biological and Environmental Science, 451 78 Fiskebäckskil, Sweden
3Swedish University of Agricultural Sciences, Department of Aquatic Resources, Institute of Marine Research 453 30 Lysekil, Sweden
[*] Corresponding author. E-mail: *ae@sik.se*

ABSTRACT

Overfishing is one of the largest environmental challenges that mankind face, since it's the largest driver for marine biodiversity loss on planet that to two thirds consist of oceans and directly limits a biotic resource of high nutritional and cultural value. Yet it has not been directly incorporated in LCA methodology which restricts a holistic scope of any Seafood LCA. We propose Wasted Potential Yield as a midpoint impact category to fill this gap, complemented with two sub impact categories explaining the main contributing mechanisms: F-Overfishing and B-Overfishedness. Characterisation methods relate to the Maximum Sustainable Yield concept that has been reinstated as a management goal for the European Union with full implantation deadline to 2015, after the ratification of Johannesburg Plan of Implementations and the United Nations Convention on the Law of the Sea. Characterisation factors were obtained for 43 European commercial stocks regarding 13 species between 2000-2010, which covered approximately half of European catches and 7% of the global catches, i.e. most of the commercially important stocks in the North East Atlantic. Due to typically high variation in fishery production system, we stress the need for both database aggregated characterisation factors and routines for continuous data collection by the LCA practitioner to minimize spatial and temporal error of representativeness.

Keywords: fisheries, impact assessment, seafood LCA, maximum sustainable yield, overfishing

1. Background

Global marine fish catches have stabilised around 80 million tons per year during the last decade (FAO 2012), although the global effort spent to catch fish has steadily increased since the seventies (Anticamara et al., 2011). Fishing fleets have expanded towards deeper and more remote fishing locations (Swartz et al., 2010) and the margins of profitability have steadily decreased (FAO 2008). This has been widely interpreted as the result of overfishing fish stocks, that are spatially or temporally separated in their reproduction and depend on their own stock size and structure for and growth (Pauly et al., 2002; Myers and Worm 2003; Mullon et al., 2005; Worm et al., 2009; Froese and Proelß 2010; FAO 2012). Contrary to earlier beliefs many fish stocks do not quickly recover when fishing pressure decreased once depleted or degraded (Hutchings and Reynolds 2004). One of the reasons is that high fishing pressure typically alters the age structure within the stock leaving younger individuals that are less efficient in reproducing (Jennings et al., 2001). As fisheries exploit the top predators of the ecosystem, the entire ecosystem will be impacted by trophic cascade effects (Frank et al., 2005). Both coastal and offshore fishing pressure has been shown to induce trophical shifts, from which a restoration to previous state is unlikely even if fishing pressure decreased (Scheffer et al., 2001; Jackson et al., 2001). The present extinction rate and loss of biodiversity has been considered the worst exceeding of planetary boundaries by humans (Rockstrom et al., 2009) and the Millennium Ecosystem Assessment established overfishing as the dominant direct driver for losses in marine ecosystems (in contrast to habitat change for most terrestrial system) (MEA 2005), thus the pursuit of a few commercial stocks indirectly effect the rest of the ecosystem. But overfishing is also directly limiting a biotic resources that today accounts for 17% of the animal protein intake worldwide, with high nutritional and economical values that are crucial for many low income and food deficient countries (FAO 2012). Fish as a product can also replace potential market shares of other environmentally costly food supplies such as beef (Winther et al., 2009). In economic terms the global fishery systems are sub-optimized, leaving many fisheries with low profitability due to low stock sizes and overcapacity. If the stocks restored to larger biomasses and then exploited with equaling catches the global profits has been estimated to increase with 50 billion $ annually equaling more than half of the value of existing landings (FAO 2008). However, such global generalisations are crude, but probably also underestimated rather than overestimated (Holt 2009).

Life Cycle Assessment (LCA) is an acknowledged and standardized method to assess potential impacts or damage related to a product or process (ISO 2006a, 2006b). The European Union has concluded that it provides the best framework for assessing the potential environmental impacts of products currently available (EC 2003). One of the benefits is the ability to compare and relate products with either potential impacts in terms of midpoint impacts or endpoint damage categories. Endpoint categories have higher model uncertainty but also higher explanatory value, and they target defined safeguard objects of Natural ecosystems, Natural resources or Human health, (Finnveden et al., 2009). The theoretical impact pathways implies that overfishing is presently damaging two out of three safeguard objects which are mandatory to check and address interpretation of according to the new ILCD standard (ILCD 2010). Clearly, overfishing is a scientifically underpinned and relevant environmental aspect of fishery systems from a natural ecosystem and resource perspective, which is stressed by an increasing international focus. Yet, no seafood LCA has accounted for overfishing as a quantitative impact category with resolution sufficient to match fishery management needs, seafood guide criteria's or existing sustainable seafood labelling frameworks, which all require (as minimum) single stock exploitation boundaries to address overfishing.

2. Theory

The Maximum Sustainable Yield (MSY) concept has been reinstated as a management goal for the European Union (EC 2006) and the ratification of the Johannesburg summit agenda has given a deadline until 2015 to restore all stocks to levels capable of producing maximum sustainable yield (JPOI 2002). Transition towards an MSY based management has also been implemented in the advice given by ICES, the International Council for Exploration of the Seas, to the European Commission and ultimately Council of Fisheries ministers, that annually set the total allowable fishing quotas for all major European fisheries. Stock summary outputs from these assessments are easily accessible on an annual basis, which are based on models using year class tracking time series of commercial landing and multinational stock surveys. Total biomass (TB) and spawning stock biomass (SSB), typically the mature part of the stock, are two of the metadata given by the assessments. Aggregated fishing mortality (F) is another, and the primary regulating indicator that measures the annual fraction of harvested spawning stock biomass. When F is set to a calculated FMSY (i.e. varying quotas but constant mortality from fisheries), the fish stocks will over time lead to MSY, the maximum long-term sustainable catches, as the biomass oscillates around BMSY (Fig. 1). In European assessments the F and B refers to SSB, and hereafter in this work, but other assessments could also refer to TB.

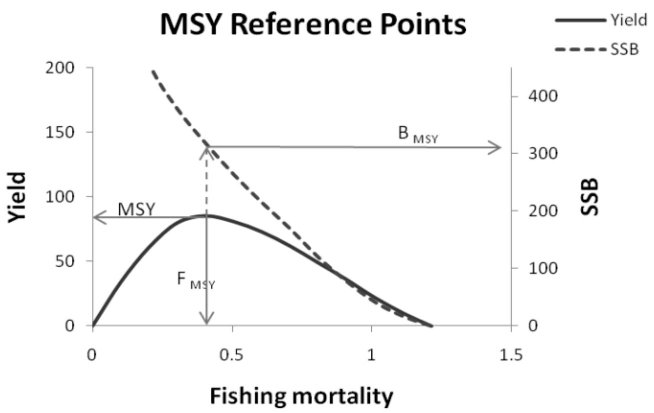


Figure 1. With increased fishing pressure F, the spawning stock biomass will decline (dotted line) rendering increased long term yields (filled line) until F = F_{MSY} after which the long term yields will start to decrease. The corresponding biomass at F_{MSY} will fluctuate around B_{MSY} implying landings at MSY. Used with kind permission from International Council for Exploration of the Seas.

However, ICES only provides $B_{TRIGGER}$ a safeguard limit above which recruitment (successful reproduction) should not be impaired. But first when the theoretical B_{MSY} values have been computed can a recovery time until full capacity is restored be established. This could motivate even higher decreases in F in some cases, as for example stopping all fishing until stocks have recovered to their full potential of providing MSY catches. When B_{MSY} values was assessed for all European stocks in 2010, the conclusion was that with current fishing pressure would have missed to restore 91% of the stocks for the management plan by 2015 (Froese and Proelß 2010).

3. Methods

To quantify pathways towards potential damage targeting natural ecosystem and natural resources we propose three mid-point impact categories related to the MSY approach. Primarily we defined the impact category of Wasted Potential Yield (WPY), which represent the long-term damage to the biotic resource and indirectly the surrounding ecosystem, in units of mass that could have been used if an optimal harvest strategy was chosen. Note that even at WPY=0 will any fishery impose theoretical damage to the ecosystem. For management purpose and minimisation of model uncertainty, we subdivided WPY into two "sub impact categories": F-Overfishing (current *rate* of on-going overexploitation, for clarity added with an "F" for fishing mortality) and B-Overfishedness (current *state* of stock, for clarity added with a "B" for biomass). Both complementing categories are expressed in relation to their corresponding optimal value in a MSY framework (Table 1).

Table 1: Proposed impact category definitions (bold), characterisation model definition and units. Strictly the true unit is kg for all categories, but a complementing qualifier has been noted in brackets.

IC Type	Impact Category	Characterisation model	Unit (qualifier)
Midpoint	**F-Overfishing (FO)**	F/F_{MSY}-1	kg (excess F_{MSY} equiv.)
Midpoint	**B-Overfishedness (BO)**	B_{MSY}/Baverage5y-1	kg (lost B_{MSY} equiv.)
Midpoint	**Wasted Potential Yield (WPY)**	Schaeffer, Baverage5y, $F_{OPTIMAL}$ (0 while B<B_{MSY} ELSE F_{MSY})	kg (wasted potential yield equiv.)

The F-Overfishing (FO) category is based on an F/F_{MSY} ratio, but has been expressed for LCA purpose as F/F_{MSY}-1 to adjust the scales so that zero, not one, corresponds to the point of optimal F, i.e. "zero emission". This strictly represents the catch taken in excess of what an F_{MSY} approach would lead to per kg catch. B-Overfishedness (BO) describes the present average biomass state in relation to BMSY (BMSY/Baverage5y-1), which means inverted compared with FO to increases with increased environmental harm and likewise scale adjusted. By this B-Overfishedness represents how many excess kg that potentially could have been caught per kg if the biomass was at optimal B_{MSY}, which is a theoretical target for fluctuating biomasses, therefore a moving average which also is used for WPY. However, an unsustainable F applied to a B_{MSY} biomass (regarding BO) will only initially generate high catches, thus the WPY better captures causal effect of biotic resource damage since it uses iteratively projections of current exploitation scenario, as if "what would be the consequence if current exploitation rate was sustained for T years forward". This is compared with if an optimal MSY strategy was used over T years. Just comparing landings with MSY would not attribute the consequences of each kg fished this year, nor capture the mechanism of present overfishing rate. To compute WPY, F, B, F_{MSY}, and B_{MSY} are related to each other in a discrete Schaeffer surplus production function. It projects next year biomass based on current added by a growth term and subtracted by the landings. The intrinsic growth rate is substituted by $2F_{MSY}$ and carrying capacity with $2B_{MSY}$, which follows from the assumption of logistic growth (Schaefer 1954) (Eq. 1).

$$B_{t+1} = B_t + 2F_{MSY}B_t\left(1-\frac{B_t}{2B_{MSY}}\right) - F_tB_t$$

Equation 1. Year discrete Schaefer biomass function, projecting next year's (t+1) biomass as current biomass B_t added with a F_{MSY} and B_{MSY} based growth term and subtracted with the removals F_t*B_t.

An optimal long term scenario was defined by setting F=0 until B reaches B_{MSY} and then harvest at F_{MSY}. The discrepancy between the projection sums of optimal yield and current yield scenarios are divided by the sum of current yield scenario. This technically represents wasted average future yields due to current harvest practise and will be dependent on the observed time period T (Eq. 2).

$$CF_{WAFY} = \sum_{T}^{1}\frac{Y_{opt}-Y}{Y}$$

Equation 2. Characterisation is defined as the differences between optimal (Y_{opt}) and current yield (Y) per unit of catch during a period of *n* years as a representation of the wasted average future yields due to current fishing practise.

We choose the time-scale of 20 years as the main scenario for the projections based on trade-offs between increased uncertainty of longer time periods (<50years) and a buffered distance to the longest break-even times (where the gain of stopped fishing until stocks are restored exceed the initial losses) (>15years). Also a relatable time period for fishermen and fishery managers were sought for (i.e lesser than active working years). Stock input parameters were retrieved from ICES publicly available Stock Summary Database (ICES

2011). Meta data was selected for stock assessments between 1995-2010 regarding 42 out of 54 major European stocks for which complementing B_{MSY} values could be found (Froese and Proelß 2010). The F_{MSY} and B_{MSY} reference values are consensus values, i.e. averages values based on three modelling approaches; 1) a demographic yield per recruit analysis 2) a surplus production analysis and 3) a stock recruitment analysis with corresponding confidence intervals (Froese and Proelß 2010). They are intended by the authors to be updated every five years with the original assessment based on time series data up until 2008 (Froese et al., 2011). Projection calculations were performed in the free statistical computing software R (www.r-project.org).

4. Results

All characterisation factors showed varied between stocks and years although 2010 are considered the main dataset. For an overview the stocks during 2010 could be aggregated into species generalised factors, se fig. 1. However, the variation within species groups is considerable: with a coefficient of variation (CV) of 1.2 for WPY and 1.0 for both F-Overfishing and B-Overfishedness, regarding all species groups with 3 or more stocks represented.

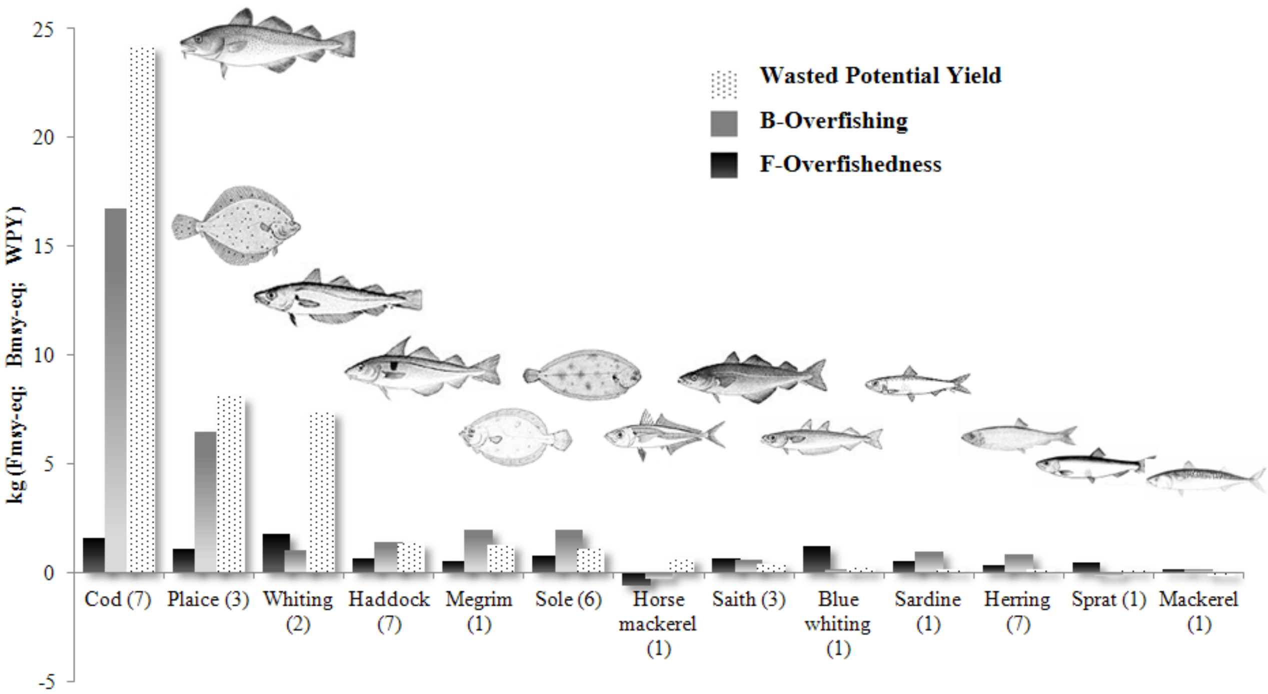


Figure 1. Characterisation of 2010 values of F-Overfishing (black), B-Overfishedness (grey) and Wasted Potential Yield (dotted) ranked according to their common mass based indicator. Brackets indicate number of stocks included. Y-axis measures lost catch in kg. Fish illustrations: FAO

Cod stocks were in worst shape in terms of wasted potential yield followed by plaice and whiting stocks, which are all demersal (ground dwelling) species relatively far up in the trophic chain. The five best placed species in relation to biotic resource use were all pelagic species of typically smaller body mass and mean trophic level. Horse mackerel (one stock) placed midrange but negative F-Overfishing and B-Overfishedness suggested that this score relates to under exploitation, all other species groups are subjects of missed yield due to overexploitations in various extents.

5. Discussion

A Schaeffer projection of an aggregated stock biomass is not the state of the art projection of fish stock used today to set exploitation quotas, even though it's still frequently used to describe the main principals of exploitation dynamics (Jennings et al., 2001). But it has been shown a useful tool for characterising the importance of metadata retrieved from state of the art stock assessments, combining measurements of present exploitation rate and the present state of biomass with MSY limits. We judge that it is sufficiently accurate for general comparison between stocks and years in an LCA's context for broad picture management. A central limitation is that only the largest and most commercially important stocks will have sufficient data documentation, rendering it practically useless for bycatches; however other completing methods for assessing bycatch exist and could in future be completed with vulnerability indices.

In a sensitivity analysis the time perspective was tested and characterisation factors (CF) generated up to 500 projection years. With longer time span (T) used for projections, the higher were badly managed stocks are relatively contributing in WPY and the higher the average WPY became for all stocks. Most stocks stabilise asymptotically with T approaching large values. Exceptions are heavily exploited stocks which instead increase rapidly but asymptotically approached a constant increase rate. However with a longer projections (typically >100years) it is unrealistic that a constant fishing mortality could be sustained, unlikely that the reference values would still be updated, and cause large error propagations. With a too short T, restoration of overexploited stocks will not be the optimal scenario and instead favoured by the (unwanted) scenario of "killing all fish one year, leaving none to the next". This forces a subjectively chosen T within the projection range where CFs has not stabilised, which indirectly becomes a matter of weighing the importance of heavily exploited stocks risk of depletion. For good and bad, the time dependency actually mimics the dilemma of fishery management; balancing the uncertainty of future catches against short time yields which may jeopardize the long term revenue. One solution could be to formulate Hierachist, Individualist and Egalitarian view perspectives by a convenient set of projections.

Decreased model- and input data uncertainty can be achieved by using the sub impact categories of F-overfishing and B-overfishedness. F/F_{MSY} based F-Overfishing is however the best choice based on input parameter uncertainty since the B_{MSY} assessment comes with a broader confidence interval in the original assessments, and the temporal variation is smallest for F. Furthermore, sustainability of fishery or biotic resource depletion potential will independent of measurement method always have high temporal variation due to the fluctuating nature of the stocks, random environmental factors and political decisions influencing the exploitation rates. Spatial variation is also substantial within each species group. For database purpose the stock entries could be aggregated into optimally sized groups that minimize the representation bias, a practical reason for this is that insufficient traceability legislations often renders it impossible to known the origin of a fish product down to the relevant stock level. However, the undisputable best resolution would be achieved if the LCA practitioner collected stock summary data in the same way as other inventory data (F, SSB and limit values) for the actual stock of concern.

By this methodology, overfishing can be described as midpoint categories, but relating to damage of natural biotic resources lays closest in term of future endpoint assessments, since Wasted Potential Catch is a mass unit that could be translated into economical terms and compared with abiotic resource depletion. Natural ecosystem damage could be compared in terms of pristine state for similar stocks, since a Schaeffer model also indicates the carrying capacity (the unfished population) as 2Bmsy corresponding to natural ecosystem. However relative importance in the ecosystem should then also be included for inclusion of both marine and terrestrial fauna.

6. Conclusion

With the inclusion of biological impact categories, LCA´s are concluded to be a useful complementary tool for fisheries managers, seafood industry or seafood labelling/consumer guides where quantitative overfishing indices has been the missing part of the toolbox. We suggest Wasted Potential Yield and/or F-Overfishing B-Overfishedness as impact categories depending on the scope of the study, and conclude them to meet demands of applicability, relevancy and scientific soundness. Without directly addressing and quantifying the biological effects of target stock, internationally acknowledged broadly as overfishing, any future seafood LCA could be misinterpreted or even deliberately misused as a biased proxy for total "environmental" damage.

7. References

Anticamara, J.A., Watson R., Gelchu A., Pauly D. 2011. Global fishing effort (1950–2010): Trends, gaps, and implications. Fisheries Research 107 (1–3):131-136.

EC 2003. COM(2003) 302 Integrated Product Policy - Building on Environmental Life-Cycle Thinking. Comisssion of the European Communities, Brussels.

EC 2006. Implementing sustainability in EU fisheries through maximum sustainable yield Communication from the Commission to the Council and the European Parliament. Brussels.

FAO 2008. The sunken billions the economic justification for fisheries reform. World Bank and Food and Agriculture Organization, Washington DC.

FAO 2012. The state of world fisheries and aquaculture (SOFIA). Rome.

Finnveden, G., Hauschild M.Z., Ekvall T., Guinée J., Heijungs R., Hellweg S., Koehler A., Pennington D., Suh S. 2009. Recent developments in Life Cycle Assessment. Journal of Environmental Management 91 (1):1-21.

Frank, K.T., Petrie B., Choi J.S., Leggett W.C. 2005. Trophic Cascades in a Formerly Cod-Dominated Ecosystem. Science 308 (5728):1621-1623.

Froese, R., Branch T.A., Proelß A., Quaas M., Sainsbury K., Zimmermann C. 2011. Generic harvest control rules for European fisheries. Fish and Fisheries 12 (3):340-351.

Froese, R., Proelß A. 2010. Rebuilding fish stocks no later than 2015: will Europe meet the deadline? Fish and Fisheries 11 (2):194-202.

Holt, S. 2009. Sunken Billions - But how many? Fisheries Research 97 (1-2):3-10.

Hutchings, J.A., Reynolds J.D. 2004. Marine fish population collapses: Consequences for recovery and extinction risk. Bioscience 54 (4):297-309.

ICES 2011. ICES Stock Summary database. ICES - The International Council of exporation of the Sea. http://ices.dk/datacentre/StdGraphDB.asp. 2011

ILCD 2010. International Reference Life Cycle Data System (ICLD) Handbook - General guide to Life Cycle Assessments - Detailed guidance. First edition edn. Joint Research Centre of the European Commission.

ISO 2006a. ISO14040 - Environmental management - Life cycle assessment - Principals and Framework. International Organization for Standardization, Geneva, Switzerland.

ISO 2006b. ISO14044 - Environmental management - Life cycle assessment - Requirements and Guidelines. International Organization for Standardization, Geneva, Switzerland.

Jackson, J.B.C., Kirby M.X., Berger W.H., Bjorndal K.A., Botsford L.W., Bourque B.J., Bradbury R.H., Cooke R., Erlandson J., Estes J.A., Hughes T.P., Kidwell S., Lange C.B., Lenihan H.S., Pandolfi J.M., Peterson C.H., Steneck R.S., Tegner M.J., Warner R.R. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293 (5530):629-638.

Jennings, S., Kaiser M., Reynolds J.D. 2001. Marine Fisheries Ecology. Wiley-Blackwell.

JPOI 2002. Johannesburg Plan of Implementation: adopted by the United Nations World Summit on Sustainable Development in Johannesburg, South Africa, on 4 September 2002. Directorate: Sustainable Development Coordination, Department of Environmental Affairs and Tourism.

MEA 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. Millennium Ecosystem Assessment (MEA), Washington DC.

Mullon, C., Freon P., Cury P. 2005. The dynamics of collapse in world fisheries. Fish and Fisheries 6 (2):111-120.

Myers, R.A., Worm B. 2003. Rapid worldwide depletion of predatory fish communities. Nature 423 (6937):280-283.

Pauly, D., Christensen V., Guenette S., Pitcher T.J., Sumaila U.R., Walters C.J., Watson R., Zeller D. 2002. Towards sustainability in world fisheries. Nature 418 (6898):689-695.

Rockstrom, J., Steffen W., Noone K., Persson A., Chapin F.S., Lambin E.F., Lenton T.M., Scheffer M., Folke C., Schellnhuber H.J., Nykvist B., de Wit C.A., Hughes T., van der Leeuw S., Rodhe H., Sorlin S., Snyder P.K., Costanza R., Svedin U., Falkenmark M., Karlberg L., Corell R.W., Fabry V.J., Hansen J., Walker B., Liverman D., Richardson K., Crutzen P., Foley J.A. 2009. A safe operating space for humanity. Nature 461 (7263):472-475.

Schaefer, M. 1954. Some aspects of the dynamics of populations important to the management of the commercial Marine fisheries. Bulletin of the Inter-American Tropical Tuna Commission 1(2): 27-56.

Scheffer, M., Carpenter S., Foley J.A., Folke C., Walker B. 2001. Catastrophic shifts in ecosystems. Nature 413 (6856):591-596.

Swartz, W., Sala E., Tracey S., Watson R., Pauly D. 2010. The Spatial Expansion and Ecological Footprint of Fisheries (1950 to Present). PLoS One 5 (12):e15143.

Winther, U., Ziegler F., Skontorp Hognes E., Emanuelsson A., Sund V., Ellingsen H. 2009. Carbon footprint and energy use of Norwegian seafood products. .

Worm, B., Hilborn R., Baum J.K., Branch T.A., Collie J.S., Costello C., Fogarty M.J., Fulton E.A., Hutchings J.A., Jennings S., Jensen O.P., Lotze H.K., Mace P.M., McClanahan T.R., Minto C., Palumbi S.R., Parma A.M., Ricard D., Rosenberg A.A., Watson R., Zeller D. 2009. Rebuilding Global Fisheries. Science 325 (5940):578-585.

Biotic resources extraction impact assessment in LCA of fisheries

Juliette Langlois[1,2*], Pierre Fréon[3], Jean-Philippe Delgenes[2], Jean-Philippe Steyer[2], Arnaud Hélias[1,2]

[1] Montpellier SupAgro, 2 place Pierre Viala, F-34060 Montpellier cedex 2, France
[2] INRA, UR50, LBE, Laboratoire de Biotechnologie et de l'Environnement, Avenue des Etangs, F-11000 Narbonne, France
[3] IRD EME R212, Av. J. Monnet, F-34203, Sète, France
[*] Corresponding author. E-mail: *langlois@supagro.inra.fr*

ABSTRACT

Because direct environmental impacts of fisheries can hardly be assessed using conventional methods of Life Cycle Assessment (LCA), we suggest building a new methodological framework to account for most of them. We propose a regionalized method of calculation for characterisation factors dedicated to an uptake of biomass through fishing activities (biotic resources extraction impact assessment). These characterisation factors are proposed for the assessment of impacts on biotic resources depletion and on life support functions of marine ecosystems. The method is applied on two examples of fisheries, to demonstrate that it is relevant for comparisons between different fisheries, exploiting different fish species. A discussion on the compatibility of this method with other frameworks is then performed.

Keywords: biotic resources extraction, fisheries, net primary production, maximum sustainable yield

1. Introduction

Life Cycle Assessment (LCA) tends to be exhaustive for the impacts it assesses, but as identified by Pelletier et al., (2007), there is a need of improvement to assess impacts of seafood products. In seafood LCA case studies, most authors deemed necessary to add non-conventional indicators (1) to take into account fish removal from their ecosystem and allow comparisons between terrestrial and aquatic food products, (2) to assess depletion of fish stocks and perturbation of the ecosystem by imbalanced exploitation between trophic levels, (3) to assess seafloor damage. To this aim, they used respectively (1) indicators of net primary production use, (2) small-size ratio of target catch, discard ratio, by-catch ratio and fishing-in-balance index, (3) area of seafloor trawled. In order to harmonize these different proposals, Langlois et al., (2011) suggested the creation of a new impact category, called "sea use" by analogy with "land use", which could allow the assessment of marine ecosystems transformation and occupation impacts. They suggested keeping the most consensual framework of terrestrial land use (Mila i Canals et al., 2007), i.e. defining a quality index whose values could be compared from a use to another and varying according to time to reach a new steady state after a certain time of restoration. They quoted the possibility to use an indicator expressing the life support capability of marine ecosystems.

In the case of biomass removal through fishing activities, impacts are especially strong. First, one or more specific stocks of wild species can be depleted by direct biomass removal and their future use by human as a natural resource can be altered (impacts on Biotic Natural Resources (BNR)). Secondly, the total biomass available for the ecosystem functioning is also decreased by this removal as well as the functioning of the whole ecosystem (impacts on Life Support Functions (LSF)). The biodiversity loss due to fishing is also severe, especially the alpha biodiversity for benthic species due to trawls dredging the seabed, with about 75% of the shelf areas trawled worldwide every year (Kaiser et al., 2002), as well as for commercial species and by-catches, due to a high intensity of direct capture (FAO, 2010).

In marine ecosystems, ecosystem production and biodiversity tend to display correlations (Libralato et al., 2008) and assessing LSF constitute a challenging issue in the present context of worldwide overfishing. Thus, the present study focuses only on the impact assessment of BNR extractions and ecosystem LSF alteration due to fishing activity; the impacts of fishing on biodiversity loss were not considered here. As underlined by Udo de Haes et al., (2002), both BNR and LSF have to be assessed. These authors explain in detail that it does not consist in double counting because two different areas of protection are considered (natural resources and ecosystem quality respectively). This work details and discusses methods for characterisation factors calculations for these two impact pathways. The method is presented in the section 2 and illustrated with an example of fishery in the section 3. Section 4 opens the way to a discussion on the relevance of the proposed methods and on their compatibility with other existing assessment methods.

2. Methods

Two methods of impact assessment are proposed and detailed for BNR and LSF in part 2.1 and 2.2 respectively. One of the constraints considered in this study was to provide some results in comparable units.

2.1. Fishing activities and biotic resources extraction impact assessment

The goal of biotic resources extraction impact assessment is to characterise to what extent the current biotic extractions worsen the possibilities for human society to cover future needs, due to stock reductions as stated by Udo de Haes et al., (2002). One commonly used reference for fish stock status assessment is the maximum sustainable yield (MSY). This is the highest yield in fish production that can be sustained in the long term (Graham, 1935; Schaefer, 1954). It results from the assumptions that current catches at time t (C_t) can be increased up to a certain level by increasing the fishing effort (E) because they are compensated by an equivalent fish production. Above the MSY level and its corresponding E_{MSY} level, the renewal of the resource (reproduction and body growth) cannot keep pace with the removal caused by fishing. In this case further increases in exploitation leads to a reduction in yields (Fig. 1). The MSY can either be calculated through different stock assessment methods or can be estimated empirically (Hilborn and Walters, 1992). Rough stock assessments are performed by FAO but the most interesting database is the RAM Legacy Stock Assessment Database, including biological reference points for over 200 stocks (Ricard et al., 2011).

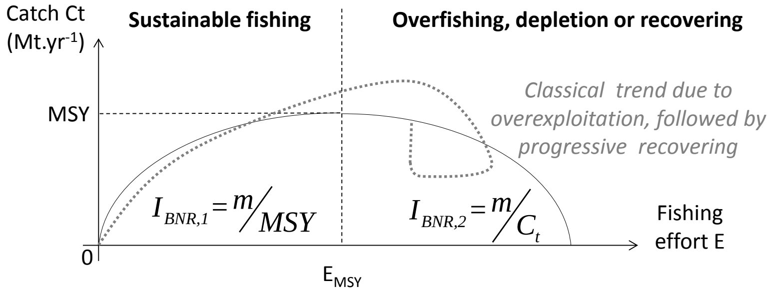


Figure 1. Trends in catches evolution according to fishing effort (in cases of equilibrium states).

We suggest an impact assessment of BNR depletion corresponding to the uptake of a mass (m) of a given marine species using the MSY related points. This allows a differentiation between different fish species, in relation with the size of their stock and the proportion that can be sustainably removed. The environmental impact on biotic natural resources ($I_{BNR,1}$) is thus calculated using the following formula:

$$I_{BNR,1} = \frac{m}{MSY} \qquad \text{Eq. 1}$$

Thus, impacts of biotic extraction resources are here expressed in potential time of regeneration, i.e. in time required to restore an uptake of a particular species assuming equilibrium conditions. This equation is valid to assess impacts of biotic extractions as long as stocks are not overexploited (underexploited, moderately exploited or fully exploited, following the typology provided by FAO), i.e. that their catch never exceeded the MSY value. These cases appear on the left side of the graph in Figure 1. Nevertheless, FAO estimates that 32% of the stocks are not in this case, being either (1) overexploited, (2) depleted or (3) recovering from depletion (FAO, 2010). This corresponds to cases where C_t is respectively (1) higher than the MSY value, (2) smaller but decreasing because of previous overexploitation or (3) smaller and increasing. These cases appear on the right side of Fig. 1.

I_{BNR} should express that the uptake of one functional unit from an overexploited stock is worse than the uptake of the same unit from a stock species having the same MSY value and being sustainably exploited. Thus it appears important to multiply $I_{BNR,1}$ by a factor depending on the gap between current catches and MSY in the case of overexploited or recovering stocks. This factor should vary from 1 to infinite for values of C_t varying from MSY to zero (when the stock is severely depleted). One of the easiest possibilities for this factor is the ratio MSY over C_t. Thus $I_{BNR,2}$ would become:

$$I_{BNR,2} = \frac{m}{MSY} \times \frac{MSY}{C_t} = \frac{m}{C_t} \qquad \text{Eq. 2}$$

In the particular case of a recent and unsteady overexploitation, where C_t is higher than MSY (Fig. 1; dashed line), we estimated that the impacts should be kept at $I_{BNR,1}$, to avoid minimizing I_{BNR} and to avoid the assessment of a transient state.

2.2. Fishing activities and life support functions assessment

The consensual framework of land use (Mila i Canals et al., 2007) has been developed in a context of intense agricultural and urban occupation as well as habitat transformation. Thus, parameters as time occupation or restoration and area used or transformed were particularly important for this impact assessment. In the case of marine activities were there is seldom continuous occupation and often slow habitat transformation, one of the major issues is to assess the quantity of biomass the ecosystem is deprived of (for fishing activities as well as for other uses, see in the discussion section). A quality index related to the alteration of biomass production capability of the ecosystem could be expressed in free Net Production (fNP). The fNP is the amount of biomass produced remaining in the ecosystem and usable for its own functioning after humans have removed a part of it from the ocean. To account for the trophic level of the biomass removed, we can use equivalence with the corresponding quantity of primary production that was necessary to produce it. Thus the quality index could be expressed in free Net Primary Production equivalent ($fNPP_{eq}$), being the Net Primary Production equivalent (NPP_{eq}) produced by the ecosystem minus the Human Appropriation of Net Primary Production equivalent ($HANPP_{eq}$). Both of them are expressed in kilogram of organic carbon per m² and per year. To fit the framework of (Mila i Canals et al., 2007), the impacts on LSF in marine ecosystems (I_{LSF}) would be the volume defined on Fig.2, expressed in kg of carbon (equivalent to primary carbon which was necessary for its production). For fishing activities, this quantity of carbon the ecosystem is deprived of, directly corresponds to the NPP_{use}, indicator (in kg C_{eq}) used in some LCA studies to quantify the impacts of seafood products, as described by Papatryphon et al., (2004).

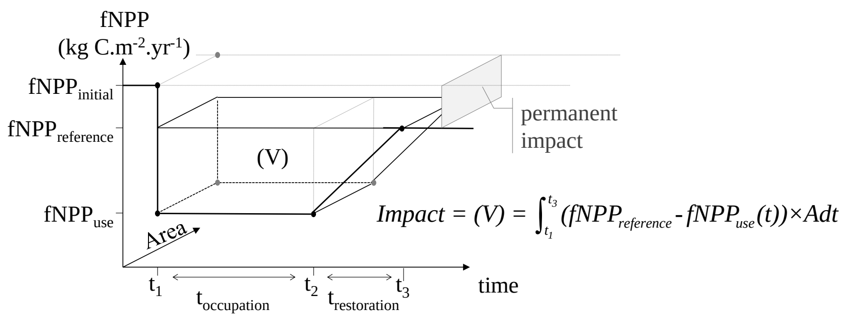


Figure 2. Graphical presentation of sea use impacts on LSF, inspired from (Mila i Canals et al., 2007).

The equivalences between fish masses and primary carbon required to sustain its production can be calculated, considering trophic levels (TL) of the uptake and the transfer efficiency between two trophic levels (TE). Updated values of TL are available per species in the fishbase database (Froese and Pauly, 2012) and updated TE values provided by Libralato et al., (2008) according to the types of ecosystems (i.e. oceanic systems, upwelling systems, tropical shelves, non-tropical shelves, coastal and coral systems). Based on these two parameters and a conservative 1:9 ratio of carbon to wet weight, NPP_{use} for a biomass uptake (m) in kg of wet weight can be calculated in kg of carbon as proposed by Pauly and Christensen (1995):

$$NPP_{use} = \frac{m}{9} \times TE^{TL-1} \qquad \text{Eq. 3}$$

This assessment has to be regionalised beyond the regionalisation of TE because the impacts are highly depending on the area where it takes place. Moreover, the value of NPP_{use} allows quantifying how much carbon the ecosystem is deprived of, but it does not provide any information about the relative importance of this uptake relative to the total value of free biomass remaining within the ecosystem. Thus, this “classical” way to assess occupation and transformation impacts can be improved, by adding a factor expressing the scarcity of the biotic resource in the ecosystem. This was suggested by Weidema and Lindeijer (2001) and used by Michelsen (2007) for land use impact assessment. The goal of the factor is to express that for the same amount of biomass removed from the sea, if it is fished in an ecosystem where biomass is scarce, the impacts on ecosystem are worse than if biomass is fished in a fertile one. Two parameters play a role to determine the scarcity of the resource: the ecosystem size ($A_{ecozone}$) and its productivity ($NPP_{mean,ecozone}$). We defined $NPP_{ecozone}$ as the total amount of NPP produced in a given ecozone for a year:

$$NPP_{ecozone} = A_{ecozone} \times NPP_{mean,ecozone} \quad \text{Eq. 4}$$

Apart from LCA, this parameter was also introduced for fishing activities impact assessment by Halpern et al., (2008) and Libralato et al., (2008). For the calculation of the impacts due to sea use on life support functions (I_{LSF}), we suggest the introduction of this factor.

$$I_{LSF} = \frac{NPP_{use}}{NPP_{ecozone}} \quad \text{Eq. 5}$$

Thus I_{LSF} expressed the time required to regenerate the amount of biomass removed from the sea. The classification of the zones is based on the Marine Ecoregions Of the World, developed by Spalding et al., (2007) and recommended for land use impact assessment by Koellner et al., (2012). World maps of NPP values for year 2010 are also available (Oregon State University, 2010). These two types of data were merged in a Geographical Information System software to compute $NPP_{ecozone}$, also using the 200m-isobath (British Oceanographic Data Centre, 2003) and the coastlines (Wessel, 2012).

3. Results

The methods developed in the previous section were applied to two simple case-studies of fisheries. The first one is the fishing of 1 kg of Atlantic cod and the second one of 1 kg of herring. They are both fished along the coastal area of the USA (Gulf of Maine). Data used for this assessment as well as the resulting Characterisation Factors (CF) and impacts are detailed in Table 1.

Table 1. Data used for characterisation factors calculation and results obtained.

	Type of data *[unit]*	**Fishery 1**	**Fishery 2**
Inventory data	m *[kg ww]* [a]	1	1
	Ecozone	Gulf of Maine	Gulf of Maine
	Species	Atlantic herring	Atlantic cod
Data used for Characterisation Factors (CF) calculation	Stock status (2004)	Recovering from depletion	Depleted
	Catch *[kg ww.yr⁻¹]* [a]	114 090[b]	4 950 [c]
	MSY[d] *[kg ww.yr⁻¹]* [a]	194 000	31 159
	TL	3	3.8
	TE (%)	14	14
	NPP_{use} *[kg C_{eq}]*	22	180
	$A_{ecozone}$ *[m²]*	136 E9	136 E9
	$NPP_{ecozone}$ *[kg $C.yr^{-1}$]*	6.8 E10	6.8 E10
CF	CF_{BNR} *[yr.kg ww^{-1}]* [a]	8.8 E-15	2.0 E-13
	CF_{LSF} *[yr.kg C^{-1}]*	3.2 E-10	2.6 E-9
Impact	I_{BNR} *[yr]*	8.8 E-15	2.0 E-13
	I_{LSF} *[yr]*	3.2 E-10	2.6 E-9

[a] ww: wet weight
[b] Average values from 2001 to 2005
[c] Average values from 2003 to 2007
[d] Informative data (not used for these particular assessments).

Both in the case of biotic natural resources extraction impacts and of life support functions, impacts of Atlantic cod fishing are higher than for Atlantic herring. This is due to a previous severe depletion of the cod stock, a relatively small value of its MSY and its higher trophic compared to herring.

4. Discussion

The MSY-related biological reference points have been widely debated, first because they are based on equilibrium conditions or steady states periods not always observed and on the assumption that production in the ecosystem can reach a stable and unique maximum (Larkin, 1977). Furthermore, single species stock assessment methods do not seem accurate for a sustainable management of marine resources and an ecosystem-based management is preferred (Botsford et al., 1997). However, these reference points are still the most commonly used to compare multiple stocks, even if not used by all management agencies (Ricard et al., 2011). The biomass reference point B_{MSY} is the internationally agreed and legally binding reference point for managed fisheries in the United Nations Convention on the Law of the Sea and the United Nations Fish Stock Agreement and provides a useful basis for comparing stocks (Ricard et al., 2011). The expression of $I_{BNR,2}$ as the inverse of current yearly catches can appear as a loss of the information due to the exclusion of MSY. Nonetheless, since this assessment is applied for the interval MSY-extinction of the stock, and C_t is bounded to MSY, MSY is still indirectly taken into account in this assessment. Furthermore it would be difficult to provide a more precise and simple assessment because the impacts induced by fishing on overexploited stocks are hardly predictable. Thus, it is hard to assess these impacts using any simple indicator, except for stocks where information about the current stock biomass (B_t) and the stock biomass at MSY (B_{MSY}) would be available. In these cases, the gap between B_t and B_{MSY} could provide a relevant information on the severity of the impact.

NPP_{use} allows the assessment of impacts due to biomass removal for the biomass landed as well as for the discards, within the same impact category. It should be noted that the calculation of oceanic NPP at a global scale using remote sensing and global models is not very accurate: a factor two exists for resulting NPP values, depending on the methods used for the calculation (Carr et al., 2006). It is mainly due to the integration of the vertical dimension of the sea. This assessment is especially uncertain in coastal areas, due to a high level of sediments in the water column, and in some deep oceanic waters where a chlorophyll deep maximum layer is observed. Moreover, the indicator NPP_{use} also presents some limits: it does not allow the recognition of an imbalance induced by fishing activities. The new impact category we propose encourages the catches in lower trophic level. This could be detrimental if this practice would become excessive.

To allow a good consistency between the different impact categories, BNR and LSF impact assessment must fit existing frameworks. For BNR, the framework is neither well defined nor consensual yet, as no operational methods has been developed in LCA. Udo de Haes et al., (2002) reviewed some suggestions for the operationalisation of BNR assessment, using the balance of exploited biomass for every species, according to its worldwide use and natural replenishment (in kg per year). This balance of overexploited biomass is bounded on zero if the use is smaller than the replenishment. It is then divided by the worldwide stock of this species or its squared value according to the authors. The resulting ratio (Q) is the inverse of the time required to destroy the stock for this species. Udo de Haes et al., (2002) suggested the use of the Red list database edited by the International Union for Conservation of Nature. It provides a level of endangerment of the species, which can be converted to coarse values of (Q), but this method of calculation does not allow a precise differentiation between species (especially for those used below their rate of replenishment). One of the major advantages of our method is that it sidesteps these limitations.

Regarding the applicability of the framework developed for LSF impact assessment to other marine activities**,** fNPP appears particularly relevant as quality index: in the case of shading impacts due to constructions, or in the case of seafloor destruction due to constructions or destructive fishing, fNPP is also decreased. Thus, both indicator and methodology would be relevant (Langlois et al., in preparation). Moreover, I_{LSF} is compatible with terrestrial land use impact assessment, as the same types of data are also available for terrestrial ecosystems (availability for values of NPP_{use} or $\Delta fNPP_{eq}$ by type of use, biogeographical classifications and maps of NPP).

5. Conclusion

Thanks to these two new impact categories, both impacts on production capability (I_{LSF}) and stock status (I_{BNR}) can be assessed using the same unit (time), which could quite easily be extended to other impact pathways linked with land or sea use. Data required for the I_{BNR} calculation were easily available, and this would be the case for most exploited stocks. The same advantage can be underlined for I_{LSF}. Thus, the methodology proposed for biomass removal from the ocean seems promissing.

Alterations of habitat by biodiversity damage have been excluded, as well as damage of benthic habitats due to trawls. This should constitute the next step of methodological improvement for this impact assessment.

6. Acknowledgements

This work benefited from the support of the French National Research Agency, (WinSeaFuel project). J. Langlois and A. Hélias are members of the ELSA research group (Environmental Life Cycle and Sustainability Assessment, http://www.elsa- lca.org/); they thank all the members of ELSA for their precious advice. The authors also thank N. Devaux, E. Crochelet and B. Mallavan for their advice in GIS data treatment.

7. References

Botsford, L. W., Castilla, J. C. and Peterson, C. H., 1997. The Management of Fisheries and Marine Ecosystems. Science 277 (5325), 509–515.

British Oceanographic Data Centre, 2003. GEBCO Digital Atlas, (consulted May 2012 http://www.bodc.ac.uk/projects/international/gebco/)

Carr, M.-E., Friedrichs, M. A. M., Schmeltz, M., Noguchi Aita, M., Antoine, D., Arrigo, K. R., Asanuma, I., et al., 2006. A Comparison of Global Estimates of Marine Primary Production from Ocean Color. Deep-Sea Res Pt II 53 (5-7), 741–770.

FAO, 2010. Part 1: World review of fisheries and aquaculture , in: FAO (Eds), State of World Fisheries and Aquaculture, 2010. Rome, pp. 3–89.
Froese, R. and Pauly, D., 2012. Fishbase. World Wide Web electronic publication
Graham, M., 1935. Modern Theory of Exploiting a Fishery, and Application to North Sea Trawling. ICES Journal of Marine Science 10 (3), 264–274.
Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., et al., 2008. A Global Map of Human Impact on Marine Ecosystems. Science 319 (5865), 948–952.
Hilborn, R. and Walters, C. J., 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Rev Fish Biol Fisher 2 (2), 177–178.
Kaiser, M. J., Collie, J. S., Hall, S. J., Jennings, S. and Poiner, I. R., 2002. Modification of Marine Habitats by Trawling Activities: Prognosis and Solutions. Fish and Fisheries 3 (2), 114–136.
Koellner, T., de Baan, L., Beck, T., Brandao, M., Civit, B., Goedkoop, M., Margni, M., i Canals, L., Müller-Wenk, R., Weidema, B. and Wittstock, B., 2012. Principles for Life Cycle Inventories of Land Use on a Global Scale. Int J Life Cycle Assess 1–13.
Langlois, J., Hélias, A., Delgenes, J. P. and Steyer, J. P., 2011. Review on Land Use Considerations in Life Cycle Assessment: Methodological Perspectives for Marine Ecosystems, in: Finkbeiner, M. (Eds), Towards Life Cycle Sustainability Management. Springer, Berlin, pp. 85–96.
Larkin, P. A., 1977. An Epitaph for the Concept of Maximum Sustained Yield. T Am Fish Soc 106 (1), 1–11.
Libralato, Coll, M., Tudela, S., Palomera, I. and Pranovi, F., 2008. Novel Index for Quantification of Ecosystem Effects of Fishing as Removal of Secondary Production. Marine Ecology Progress Series 355, 107–129.
Michelsen, O., 2007. Assessment of Land Use Impact on Biodiversity. Int J Life Cycle Assess 13 (1), 22–31.
Mila i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R. and Rydgren, B., 2007. Key Elements in a Framework for Land Use Impact Assessment Within LCA. Int J Life Cycle Assess 12, (1) 5–15.
Oregon State University, 2010. Ocean Productivity, (consulted February 2012 http://www.science.oregonstate.edu/ocean.productivity/custom.php)
Papatryphon, E., Petit, J., Kaushik, S. J. and van der Werf, H. M. G., 2004. Environmental Impact Assessment of Salmonid Feeds Using Life Cycle Assessment (LCA). AMBIO: A Journal of the Human Environment 33 (6), 316–323.
Pauly, D. and Christensen, V., 1995. Primary Production Required to Sustain Global Fisheries. Nature 374 (6519), 255–257.
Pelletier, N. L., Ayer, N. W., Tyedmers, P. H., Kruse, S. A., Flysjo, A., Robillard, G., Ziegler, F., Scholz, A. J. and Sonesson, U., 2007. Impact Categories for Life Cycle Assessment Research of Seafood Production Systems: Review and Prospectus. Int J Life Cycle Assess 12, 414–421.
Ricard, D., Minto, C., Jensen, O. P. and Baum, J. K., 2011. Examining the Knowledge Base and Status of Commercially Exploited Marine Species with the RAM Legacy Stock Assessment Database. Fish and Fisheries (in press).
Schaefer, M. B., 1954. Some Aspects of the Dynamics of Populations Important to the Management of the Commercial Marine Fisheries. Bull. IATTC 1 (2), 27–56.
Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdana, Z. A., Finlayson, M., Halpern, B. S., et al., 2007. Marine Ecoregions of the World: a Bioregionalization of Coastal and Shelf Areas. BioScience 57 (7), 573–583.
Udo de Haes, H. A., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E., Hofstetter, P., Jolliet, O., et al., 2002. Life cycle impact assessment: Striving towards best practice, SETAC Press, Pensacola.
Weidema, B. P. and Lindeijer, E., 2001. Physical Impacts of Land Use in Product Life Cycle Assessment, Technical University of Denmark.
Wessel, P., 2012. GSHHS Coastlines, SOEST Univesity of Hawai'i, Honolulu (consulted February 2012, http://www.soest.hawaii.edu/pwessel/gshhs/index.html).

Regionalised method to assess soil erosion in LCA

Montse Núñez[1,2,*], Assumpció Antón[1,3], Pere Muñoz[1], Joan Rieradevall[4]

[1] IRTA, Sostenipra. Centre de Cabrils 08348 Cabrils, Barcelona, Spain
[2] INRA-LBE, Avenue des Etangs 11000 Narbonne, France
[3] Departament d'Enginyeria Química, Universitat Rovira i Virgili 43003 Tarragona, Spain
[4] ICTA, Sostenipra. Universitat Autònoma de Barcelona 08193, Bellaterra, Barcelona, Spain
[*] Corresponding author. E-mail: *montse.nunez@supagro.inra.fr*

ABSTRACT

One of the most relevant and endangered ecosystem services worldwide due to land use change and inappropriate land use practices is the capacity of ecosystems to stabilize soils by preventing soil erosion. This research was meant to develop a method to include the assessment of soil erosion in LCA on a global scale. The method deals with land occupation impacts. As soil erosion depends on local conditions, characterisation factors were regionalised on a 5 arc-minutes grid-cell level resolution. Two endpoint indicators covering the areas of protection damage to resources and damage to ecosystem quality were proposed. The method was applied to the agricultural stage of five agricultural crop rotations. Further research efforts should aim at applying the method beyond the agricultural stage and to identify a feasible and relevant spatial scale at which to aggregate characterisation factors to cope with data gaps on location of processes.

Keywords: ecosystem services, land use, soil erosion, spatial differentiation, USLE

1. Introduction

Ecosystem services (ES) are resources and processes that are supplied by natural ecosystems. Despite their fundamental role in sustaining ecosystem functioning and human activities, they have been traditionally disregarded in life cycle oriented methods such as life cycle assessment (LCA), a method that is meant to encourage environmental sustainability. This oversight can lead to improper decisions. One of the most relevant and endangered ES worldwide due to land use change and unsustainable land use management practices is the capacity of ecosystems to stabilise soils and to prevent soil sediment accumulation downstream. Soil erosion leads to a reduction of soil quality, as usually a large amount of nutrients is lost together with the topsoil. Nutrient impoverished soils are less capable to provide ecosystem services. Detached soil nutrients, such as phosphorous, organic matter and heavy metals lead to pollution of water courses and lost particles to contamination of the air. Annually, humans cause de loss of 50 to 75 billion metrics of soil (Harvey and Pimentel, 1996). Agricultural lands account for 75% of the soil erosion worldwide, though it also occurs in other human-modified ecosystems, such as during the construction of roads and buildings. Several methods deal with the issue of erosion in LCA. Cowell and Clift (2000) proposed a non-spatially-explicit indicator, thus one generic characterisation factor (CF), for assessment of soil depletion, the static reserve life (years). The LANCA® calculation tool (Beck et al., 2010), based on Baitz (2002)'s method, allows assessing soil loss for specific cases of land use that can be used within life cycle inventory (LCI) databases. Saad et al., (2011) adopted the LANCA® modelling approach to derive CFs of land occupation and land transformation for Canada at different spatial scales. Van Zelm et al., (2011) proposed CFs for erosion due to agricultural land occupation for the world at the country level. The methods by Saad et al., (2011) and Van Zelm et al., (2011) are spatially-resolved, that is, CFs include site-specific geospatial information, because land use impacts are highly influenced by local conditions regarding climate, soil properties and landscape. So far, even though some proposals already exist for LCA to integrate soil erosion, approaches are still in its infancy and the discussion on its characterisation is far from being settled.

The objective of this research was to go one step further towards the integration of ES in LCA by developing a regionalised method for the world to include land occupation impacts of any type of human activity on the erosion regulation ES. To show the applicability of the method, erosion and environmental impacts from growing agricultural rotations with food and energy crops in Spain were assessed.

2. Method

Two endpoint indicators covering the areas of protection (AoP) damage to resources and damage to ecosystem quality were defined. CFs for the two modelled impact pathways were regionalised on a 5 arc-minutes (approximately 10*10 km^2) resolution grid, without further aggregation on broader scales.

2.1. Resource-depletion impact pathway

The modelled cause-effect chain follows this pathway: land occupation leads to soil erosion and this leads to loss of topsoil reserves, which eventually leads to soil resource depletion. Current soil losses reduce soil availability as a future resource.

We proposed to calculate soil losses in the LCI with the universal soil loss equation (USLE, Wischmeier and Smith, 1978), although other estimation models may also be applied. The endpoint life cycle impact assessment (LCIA) model informs of the decrease in local soil stock as response to soil loss mass due to land occupation. We differentiated twenty-one soil-depth classes following the FAO's soil-depth map of the world (FAO/UNESCO, 2007), from very shallow soils (0.05 m deep) to very deep soils (2.25 m deep). For every soil-depth class we derived a CF. Larger CFs have been assigned to thinner soils as, from a resource perspective, they are more vulnerable than thicker soils. Final damage is expressed as surplus (solar) energy needed to rebuild the stock of soil loss on the used area during the time of occupation. A global (solar) energy quality value for all soil types and locations was used (23.9 MJ g^{-1} soil loss, Odum, 1996), as more specific values are not presently available. Further details on the proposed indicator are reported in Núñez et al., (2012a).

2.2 Ecosystem-quality impact pathway

The modelled cause-effect chain follows this pathway: land occupation leads to soil erosion and to the loss of soil organic carbon (SOC), this reduces biomass productivity, which eventually affects overall biodiversity and ecosystem quality.

The endpoint indicator for damage to ecosystem quality can be divided in a fate step, linking SOC loss to soil loss ($SOC_{loss} = soil_{loss}$ *%SOC), and an effect step, linking ecosystem biomass productivity drop to SOC loss ($\%NPP_{loss} = aSOC_{loss} + b$). Previous LCA methods proposed soil organic matter content (Milà i Canals et al., 2007) and ecosystem biomass productivity (Cowell, 1998; Lindeijer, 2000; Pfister et al., 2011) as indicators for life support functions of land. The LANCA® operational tool (Beck et al., 2011) also includes the biotic production of the ecosystem as an indicator for land use impacts. In our approach, we combined both indicators: ecosystem biomass production is modeled as a function of soil quality, which is indicated by the SOC of the soil lost. We assessed ecosystem biomass productivity as a function of the net primary production of the potential natural vegetation (NPP_0). Final damage expresses the net primary production depletion (NPPD) as a response to soil loss mass due to the occupation of an area during a time period. For an occupation of 1 m^2 and 1 year, NPPD ranges from 0 to 1(percentage expressed as a decimal). Larger CFs have been assigned to the most productive soils, as NPP is a scarce resource on earth. Further details on the proposed indicator are reported in Núñez et al., (2012a).

2.3 Case study on food and energy crops

The method was applied to the agricultural stage of five three-year crop rotation systems with food and energy crops in Spain. Spain was selected as a representative case study because it is a highly diverse country in climate, soil and topography, where water erosion is one of the main causes of land degradation. Of the analysed rotations, three were traditional rainfed rotations of annual crops grown in the Mediterranean region: i) winter barley-winter wheat-rye, ii) winter barley-winter wheat-pea, and iii) winter barley-winter wheat-unseeded fallow. Another was a rainfed rotation where a bioenergy crop was introduced: iv) winter barley-winter wheat-oilseed rape; and finally, a deficit-irrigated short rotation coppice of a perennial crop, poplar, cultivated in three years rotation cycles during twelve-fifteen years: v) poplar-poplar-poplar. Results of the assessment are useful to validate the applicability of the method and to compare the environmental costs of food and energy-crop rotations on the soil erosion regulation ES. The analysis includes 120 agricultural plots located throughout the country. The impact assessment was performed for the occupation of one square metre during one year ($1m^2y$), taking the annual average value of soil and SOC loss during the three years of the rotation.

3. Results

3.1 Characterisation factors

Fig. 1 shows the regionalised CFs for (a) the resource-depletion impact pathway and (b) the ecosystem-quality impact pathway. As stated previously, for resource depletion, larger CFs were for thinner soils, and for ecosystem quality, larger CFs were for soils with higher biomass productivities.

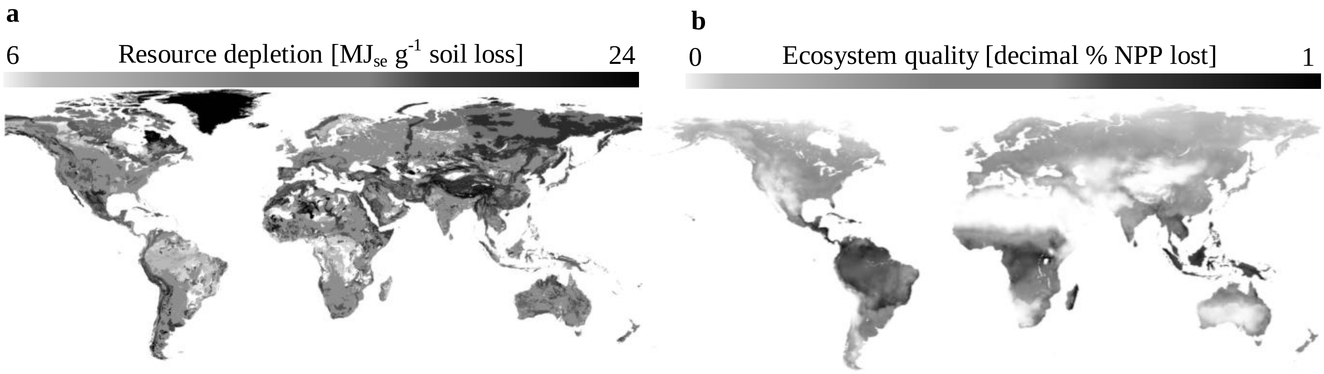


Figure 1. Characterisation factors for (a) resource depletion and (b) ecosystem quality.

3.2 Case study on food and energy crops

At the LCI level, soil erosion of three of the crops grown in the evaluated rotations as well as of the unseeded fallow reference system are depicted in Fig. 2. Results are shown grouping the analysed plots on the ten main water-basins in Spain and on the country scale. It can be seen how soil loss varies as a function of location for the same crop, hence the importance of including geospatial information as accurately as possible in the inventory step.

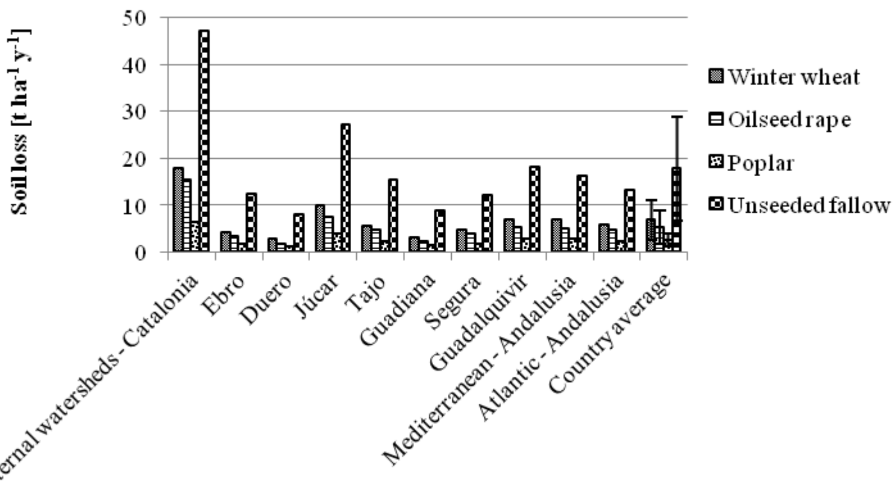


Figure 2. Soil erosion (t ha^{-1} y^{-1}) on the 120 studied plots grouped on the water-basin and arithmetic average ±standard deviation on the country level. The reported soil losses were calculated for a whole year (crop + crop residues + rough fallow before next seeding).

Results at the LCIA stage for the five crop rotations evaluated are shown in table 1. As in the inventory, damage to resources and to ecosystem quality varies greatly between water-basins for one and the same crop rotation, with a coefficient of variation higher than 60% in almost all cases (except for the poplar rotation, CV is 35%). According to the presented results, the most appropriate combination of rotation and location was the poplar energy-crop short rotation forestry grown in the North of the country (Duero water basin in table 1). Other water basins, such as the Guadiana (Central Spain, results not shown here) scored very close to the Duero water basin.

Table 1. Damage to resources and to ecosystem quality (LCIA) of the five crop rotation systems on five important watersheds in Spain and arithmetic country average. Results are per m^2 and year of land occupation.

	Ebro	Duero	Tajo	Segura	Guadalquivir	Country average ± SD
Resources [MJ_{se}]						
B-W-R[a]	6366	4125	9028	5778	9943	10410 ± 6290
B-W-P[b]	6612	4394	9293	5777	9841	10585 ± 6444
B-W-F[c]	10285	6978	14450	8948	15608	16198 ± 9652
B-W-OR(*)[d]	6000	3606	8691	5576	9156	9729 ± 5941
PP(*)-PP(*)-PP(*)[e]	1334	746	1602	975	1585	1776 ± 1142
Ecosystem quality [NPPD, decimal% of NPP lost]						
B-W-R[a]	0.03	0.02	0.04	0.02	0.05	0.047 ± 0.028
B-W-P[b]	0.04	0.02	0.04	0.02	0.05	0.048 ± 0.028
B-W-F[c]	0.05	0.03	0.06	0.03	0.07	0.070 ± 0.043
B-W-OR(*)[d]	0.03	0.02	0.04	0.02	0.05	0.045 ± 0.027
PP(*)-PP(*)-PP(*)[e]	0.02	0.01	0.01	0.01	0.02	0.017 ± 0.006

[a] winter barley – winter wheat – rye
[b] winter barley – winter wheat – pea
[c] winter barley – winter wheat – unseeded fallow
[d] winter barley – winter wheat – oilseed rape
[e] poplar – poplar – poplar
Asterisks indicate crops for energy use.

4. Discussion

4.1 Soil-erosion impact assessment model

Two endpoint indicators for the AoP soil resources and ecosystem quality were proposed in order to include in LCA impacts of land occupation on the soil erosion regulation ES. Compared to previous methods, the added value of this research is that we provide CFs for the world which can be applied to assess land occupation impacts of any type of human activity. Land transformation impacts were not addressed. Due to the complex link between soil loss and human health damage, this impact pathway was excluded from the assessment. CFs were regionalised on the grid-cell level without aggregating them on broader administrative or ecological scales. Yet, a relevant and feasible scale of aggregation should be found in order to deal with data gaps on location of processes, especially in the background system. However, this is a complex issue, due to the huge variability of soils even at the landscape scale. Variability and uncertainty of location of processes (e.g., plastic and fertiliser production of the background system) should also be assessed.

4.2 Case study on food and energy crops

A great variability of soil losses was recorded in the inventory step, depending on climatic and edaphic conditions. The same trend was found in the impact assessment step, as the regionalised model developed includes information on the sensitivity of the receiving ecosystem to impacts of land use. It is therefore very important that land-use impact models include geospatial information in both the LCI and LCIA phases. The results of the case study show that the poplar energy-crop rotation system in Spain can potentially reduce erosion rates and environmental impacts per area-time unit compared to traditional cereal and legume crop rotations. The use of other functional units (e.g., kg, MJ) might have led to different results. However, the selection of non-area-based functional units when crops and rotations with different functions are being compared is an unresolved issue in LCA (Núñez et al., 2012b). In relation to the recommended locations to cultivate the analysed crop rotations to reduce soil erosion impacts, results indicate a trend. However, these results should be interpreted with caution. They are valid for the analysed plots, but each specific case should be studied separately, as soil erosion depends on plot level factors. Only the agricultural stage was included in the assessment, thus the soil erosion model here developed should be applied to the overall life cycle (i.e., from the production of seed to the use of the crop or to the final waste management) to perform a proper LCA study. The extension of the system boundary beyond the agricultural stage would be facilitated if unit processes of LCI datasets had information on the amount of soil loss. To this end, the LANCA® tool can be used, as already implemented in inventory flows of the GaBi software. However, we may consider the obtained results for the agricultural stage to be representative of the total soil losses during the complete life cycles of crops, as agriculture is by far the land use activity with higher soil erosion rates.

5. Conclusion

We developed a globally applicable, spatially differentiated method to account for land occupation impacts in LCA, focusing on the aspect of soil erosion. LCI data needed was set up, CFs developed on a grid-cell level resolution, and LCIA models for the AoP soil resources and ecosystem quality proposed. The method aims to contribute to a better land use impact assessment of agricultural and forestry production systems. It was successfully applied to the agricultural stage of producing different food and energy crops. Further research should focus on testing the applicability of the method across the overall life cycle of a product and to determine the most relevant and feasible scale at which to aggregate CFs to deal with data gaps on location of processes. A site-generic CF, which is useful for processes of unknown location, may be derived aggregating all CFs on the grid-cell level. The freshwater ecoregions regionalisation approach, as suggested in Koellner et al., (2012), might be an option of aggregation to be studied in further work.

6. References

Baitz, M., 2002. Die Bedeutung der funktionsbasierten Charakterisierung von Flächen-Inanspruchnahmen in industriellen Prozesskettenanalysen: Ein Beitrag zur ganzheitlichen Bilanzierung. Dissertation. Berichte aus der Umwelttechnik. Aachen: Institut für Kunststoffprüfung und Kunststoffkunde, Universität Stuttgart: Shaker Verlag.

Beck, T., Bos, U., Wittstock, B., Baitz, M., Fischer, M., Sedlbauer, K., 2010. LANCA® Land use indicator value calculation in life cycle assessment – method report. University of Stuttgart.

Cowell, S.J., 1998. Environmental life cycle assessment of agricultural systems: integration into decision-making. PhD thesis. University of Surrey, Guildford, Surrey, UK.

Cowell, S.J., Clift, R., 2000. A methodology for assessing soil quantity and quality in life cycle assessment. J. Ind. Ecol. 8, 321-331.

FAO/UNESCO, 2007. Effective soil depth raster map. http://www.fao.org/geonetwork/srv/en/main.home.

Harvey, C.A., Pimentel, D., 1996. Effects of soil and wood depletion on biodiversity. Biodivers. Conserv. 5, 1121-1130.

Koëllner, T., Baan, L., Beck, T., Brandao, M., Civit, B., Goedkoop, M., Margni, M., Milà i Canals, L., Müller-Wenk, R., Weidema, B., Wittstock, B., 2012. Principles for life cycle inventories of land use on a global scale. Int. J. Life Cycle Assess. doi:10.1007/s11367-012-0392-0.

Lindeijer, E., 2000. Review of land use impact methodologies. J. Cleaner Prod. 8, 273-281.

Milà i Canals, L., Romanyà, J., Cowell, S.J., 2007. Method for assessing impacts on life support functions (LSF) related to the use of fertile land in life cycle assessment (LCA). J. Cleaner Prod. 15, 1426-1440.

Núñez, M., Antón, A., Muñoz, P., Rieradevall, J., 2012a. Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain. Int. J. Life Cycle Assess. Accepted for publication.

Núñez, M., Pfister, S., Antón, A., Muñoz, P., Hellweg, S., Koehler, A., Rieradevall, J., 2012b. Assessing the environmental impacts of wáter consumption by energy crops grown in Spain. J. Ind. Ecol. doi: 10.1111/j.1530-9290.2011.0049.x.

Odum, H.T., 1996. Environmental accounting: emergy and environmental decision making. John Wiley and Sons, New York, United States.

Pfister, S., Bayer, P., Koehler, A., Hellweg, S., 2011. Environmental impacts of water use in global crop production: hotspots and trade-offs with land use. Environ. Sci. Technol. 45, 5761-5768.

Saad, R., Margni, M., Koellner, T., Wittstock, B., Deschênes, L., 2011. Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. Int. J. Life Cycle Assess. 16, 198-211.

Van Zelm, R., van der Velde, M., Núñez, M., Huijbregts, M.A.J., 2011. Spatially explicit characterization factors for impacts of agriculture on ecosystems services. 21st SETAC Europe Annual Meeting, 15-19 May 2011, Milano, Italy.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses – A guide to conservation planning, Agricultural Handbook no. 537, Department of Agriculture, United States.

The effect of crop management on soil organic matter in the carbon footprint of agricultural products

Simona Bosco*, Claudia Di Bene, Enrico Bonari

Institute of Life Sciences – Scuola Superiore S. Anna, Piazza Martiri della Libertà, 33 - 56127 Pisa, Italy
* Corresponding author. E-mail: *s.bosco@sssup.it*

ABSTRACT

The agricultural sector has an important role in greenhouse gas (GHG) mitigation, since cropland soils can also act as sinks. Attention has already been drawn to the value of soil quality and soil organic matter (SOM) in the life-cycle of agricultural products. Nevertheless, many life-cycle assessment studies of food and non-food products do not take into account changes in SOM in arable land. A simpler one-compartment SOM model is needed to be integrated into the LCA study. Recently, many authors have adopted the Hénin-Dupuis SOM model for cropping system and orchard studies under different climate conditions, and it has proved useful for less-detailed modelling at sites where input requirements for running the more complex models are not readily available. The model was applied to wine production considering four scenarios with decreasing levels of OM inputs. This SOM balance method was sensitive to changes in management practices.

Keywords: crop management, Hénin-Dupuis model, soil organic matter, organic matter input, tillage intensity.

1. Introduction

The role of agriculture in climate mitigation centres on a conservative soil management designed to both protect soil quality and guarantee GHG emissions reduction (Smith et al., 2008). It is well-established that deep tillage, over-fertilisation, excessive use of pesticides and irrigation, and the removal of crop residues can dramatically affect soil quality (Lal 2004).

As Nemecek and Erzinger (2005) have indicated, farms are characterised by high variability in both natural and management factors, and by difficult-to-measure emissions. Attention has already been drawn to the crucial role of soil quality and soil organic matter (SOM) in the life-cycle of agricultural products (Cowell and Clift 1997, 2000; Milà i Canals et al., 2007). Nevertheless, many life-cycle assessment studies of food and non-food products do not take into account changes in SOM in arable cropland (Brentrup et al., 2004; Milà i Canals et al., 2006; Mourad et al., 2006; Hillier et al., 2009; Roy et al., 2009; Gazulla et al., 2010; Muñoz et al., 2010; Bosco et al., 2011). This remains something of an unresolved issue in LCA, due to the spatial and temporal variations, and local environmental uniqueness of soil quality (Reap et al., 2008). Neither PAS2050 or GHG Protocol Standard consider until today soil emissions except in instances of land-use changes (BSI, 2011; WRI/WBCSD, 2004). To date, few studies have incorporated soil into an LCA study (Beck et al., 2010; Mila i Canals et al., 2007; Meisterling et al., 2009; Brandão et al., 2011; Nemecek et al., 2011; Ponsioen and Blonk 2011; Saad et al., 2011) and no common methodology for its estimation and LCA inclusion exists.

Soil quality and SOM changes are entirely site-specific, as they are strongly influenced by management practices and soil and climate conditions. Many process-oriented SOM models are available for charting its evolution on a daily, monthly or annual basis and they have already been incorporated into LCA analyses (Milà i Canals et al., 2007; Hillier et al., 2009). However, the huge amount of data required (meteorological data, crop phenological data, and chemical and physical soil characteristics) to run these models and to establish the life cycle inventory (LCI) could limit widespread use of this approach.

A simpler one-compartment SOM model would be more easily integrated into the LCA study. Recently, many authors have adopted the Hénin-Dupuis SOM model for cropping system and orchard studies under different climate conditions, and it has proved useful for less-detailed modelling at sites where input requirements for running the more complex models are not readily available (Sofo et al., 2005; Bayer et al., 2006; Bockstaller et al., 2008; Bechini et al., 2011; Di Bene et al., 2011).

In this paper we proposed a new method for soil inclusion in agriculture LCA, integrating the Hénin-Dupuis SOM model into a carbon footprint (CF) analysis. Sample results were reported in a case study on wine to test the impact of different management practices on the overall result. Wine was chosen as a case study because it is a widely-studied food product and because of its importance within international and Italian food markets.

2. Methods

2.1. Soil organic matter model

The soil organic matter model, based on the first-order kinetics model developed by Hénin and Dupuis (1945), evaluates the effect of agricultural practices on the evolution of the SOM pool and it is described by the following equation:

$$SOM_t = SOM_0 e^{-k_2 t} + k_1 OM_I / k_2 \, (1 - e^{-k_2 t}) \qquad \text{Eq. 1}$$

where SOM_t is the SOM pool (Mg ha^{-1}) at time t; SOM_0 is the initial SOM pool (Mg ha^{-1}) at time t = 0; k_2 is the mineralisation coefficient corresponding to the annual rate of SOM loss by mineralisation; k_1 is the humification coefficient and refers to the annual rate of OM inputs incorporated in SOM; and OM_I is the annual OM inputs (Mg ha^{-1}). A detailed view of the model structure is given in Fig. 1.
The first component of equation 1, $SOM_0 e^{-k_2 t}$, represents the fraction of SOM_0 still in the soil at time t. The second component, $k_1 \, OM_I / k_2 \, (1 - e^{-k_2 t})$, is the fraction of the SOM pool deriving from the humification of organic material inputs.
The soil organic matter pool was calculated according to the following equation:

$$\text{SOM pool} = (SOM_c \times BD \times d \times A)/100 \qquad \text{Eq. 2}$$

where SOM pool is organic matter stock (Mg ha^{-1}); SOM_c is soil organic matter concentration (g kg^{-1}), as determined using the modified Walkley–Black wet combustion method (Nelson and Sommers, 1982); BD is the soil bulk density (g cm^{-3}), measured by the Culley method (1993); d is the soil sampling depth (0.30 m); and A is the area being considered (1 ha = 10,000 m^2).
The mineralisation coefficient (k_2) is affected by climate conditions (air temperature) and soil characteristics (texture and lime content). As in Boiffin et al., (1986) and Bockstaller and Girardin (2003), k_2 was calculated as follows:

$$k_2 = 1200 \times f_\theta \, [(c + 200) \times (l + 200)] \qquad \text{Eq. 3}$$

where f_θ is a temperature factor given by $f_\theta = 0.2\,(T-5)$, where T is the average annual air temperature (°C), c is clay content (g kg^{-1}), and l is limestone content (g kg^{-1}). As proposed by Mary and Guérif (1994) and Bechini et al., (2011), a dimensionless correction factor of the mineralisation coefficient (P) was used for the inclusion of farm soil management. P was calculated as:

$$P = p_r \times f_r \times I \times T_s \qquad \text{Eq. 4}$$

where p_r refers to maximum crop plough depth (D), where $p_r = 0.0333 * D$; f_r is a coefficient considering crop management (for example, frequency of ploughing, frequency of residue incorporation, manure) as proposed by Mary and Guérif (1994); I is the mineralisation weight factor (1.25 and 1.00 for irrigated and non-irrigated crops, respectively); and T_s is the tillage factor (1 where the soil is ploughed at least once every four years, 0.5 for non-tillage management, and 0.8 for intermediate cases).

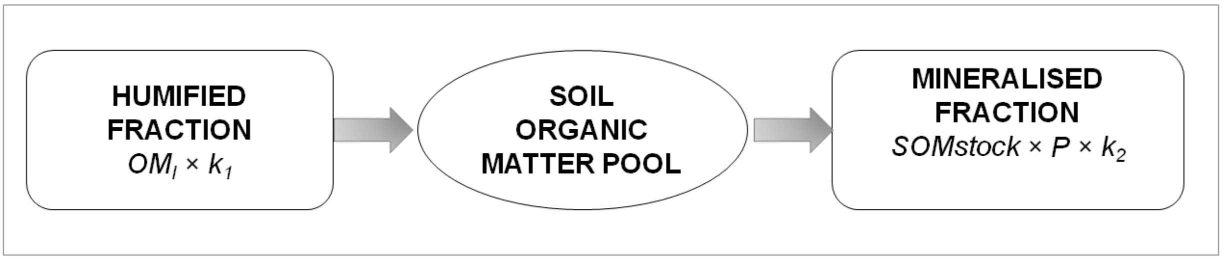


Figure 1. The Hénin-Dupuis soil organic matter balance.

The Hénin-Dupuis model was integrated into and run using the GaBi5 software by creating a parameterised process for each vineyard sub-phase (planting, pre-production and production) (GaBi, 2012). Thus, SOM changes were evaluated for each sub-phase, and the results were allocated on the basis of the temporal length of each sub-phase in years in relation to overall vineyard lifespan. The results obtained, expressed in

Mg SOM ha^{-1}, were converted into Mg CO_2 ha^{-1} using a SOM/SOC ratio of 1.724 (van Bemmelen factor, in Nelson and Sommers, 1982) and a C/CO_2 molecular weight ratio of 3.66 (44 CO_2/12 C g $mole^{-1}$).

Case study

The study area was the hilly inland region in Southern Tuscany, Italy. Climate is typically Mediterranean, characterised by two main rainy seasons in the autumn and the spring, a total annual rainfall of around 800 mm, and an average temperature of 14.5° C. Soils are quite variable with a texture ranging from silt loam to clay. The main physical and chemical characteristics of the soil on the case study vineyard are reported in Table 1.

Table 1. Soil parameter on the case study vineyard needed to run the model.

Soil parameter	Unit	Value
Sand	(g/kg)	200
Silt	(g/kg)	485
Clay	(g/kg)	315
Limestone	(g/kg)	0.00
Bulk density	(g/cm^3)	1.30
Soil organic matter	(g/kg)	1.19

The functional unit (FU) used for the study was one 0.75 L bottle of wine, and all data refers to the year 2009. The system boundary was divided into two main phases, the vineyard and the winery, and seven sub-phases, including vineyard planting (1 year), pre-production sub-phase (3 years), production sub-phase (27 years), vinification, bottling, packaging and distribution.

Data on vineyard soil management, such as manure distribution, use of pruning residues and inter-row vineyard grass cover or grassing was collected using specific questionnaires, while data relating to the physical and chemical characteristics of the soil was provided by farmer in the form of soil samples collected at the vineyard planting stage. CO_2 emissions/removal caused by carbon stock changes in vine biomass were not included, since its carbon pool is considerably smaller than that of soil, less than 1% (Keightley 2011), and the corresponding vine biomass C pool is removed at the end of the vineyard production period. Direct and indirect N_2O soil emissions from synthetic and organic fertilisers were calculated using the IPCC methodology and emissions factors (IPCC, 2006).

For this study, GWP impact was evaluated by considering the CO_2, CH_4 and N_2O emissions associated with energy and material inputs during each sub-phase of the production chain. Biogenic emissions were not considered.

The baseline situation and four scenarios have been elaborated to assess the effect of crop management, considering a decreasing levels of organic matter inputs, as follow:

- S1: manure distribution at vineyard planting; inter-row grassing with cover crops; incorporation of pruning residues into the soil;
- S2: no manure distribution; inter-row grassing with cover crops; incorporation of pruning residues into the soil;
- S3: manure at vineyard planting; tillage for weed control; pruning residues removed;
- S4: no manure distribution; tillage for weed control; pruning residues removed.

The values used in the scenarios are reported in Table 2.

Table 2. Values used in the baseline and in the scenarios to simulate the effect of vineyard management.

Treatment	Unit	Baseline	S1	S2	S3	S4
Manure at planting	Mg f.m. ha^{-1}	20	65	0	65	0
Incorporation of pruning residues	Mg d.m. ha^{-1}	0	3.02	3.02	0	0
Inter-row grassing with cover crops	Mg d.m. ha^{-1}	0	4.5	4.5	0	0

3. Results

Table 3 shows the SOM pool evolution during the vineyard's lifespan, from the vineyard planting to the last productive year. For each sub-phase, the organic inputs and the organic outputs are reported, as well as the intermediate SOM balance.

Table 3. SOM pool evolution (Mg ha^{-1} year^{-1}) in the baseline scenario during the vineyard lifespan (SOM_{vp}: SOM the end of vineyard planting; SOM_{pp}: SOM at the end of pre-production period; SOM_p: at the end of production period)

SOM before planting	Vineyard planting			Pre-production			Production		
	OM_I	**OM_o**	**SOM_{vp}**	**OM_I**	**OM_o**	**SOM_{pp}**	**OM_I**	**OM_o**	**SOM_p**
46.41	1.56	2.59	45.38	0.40	0.58	45.20	0.48	0.58	45.10

The scenarios were established to evaluate the effects of different vineyard management techniques, using manure distribution, residue management and inter-row grassing as variables. The scenario results, showed in Fig. 2, have been expressed in kg CO_2eq for FU and presented in two sections, with the vineyard and winery phases being treated as separate.

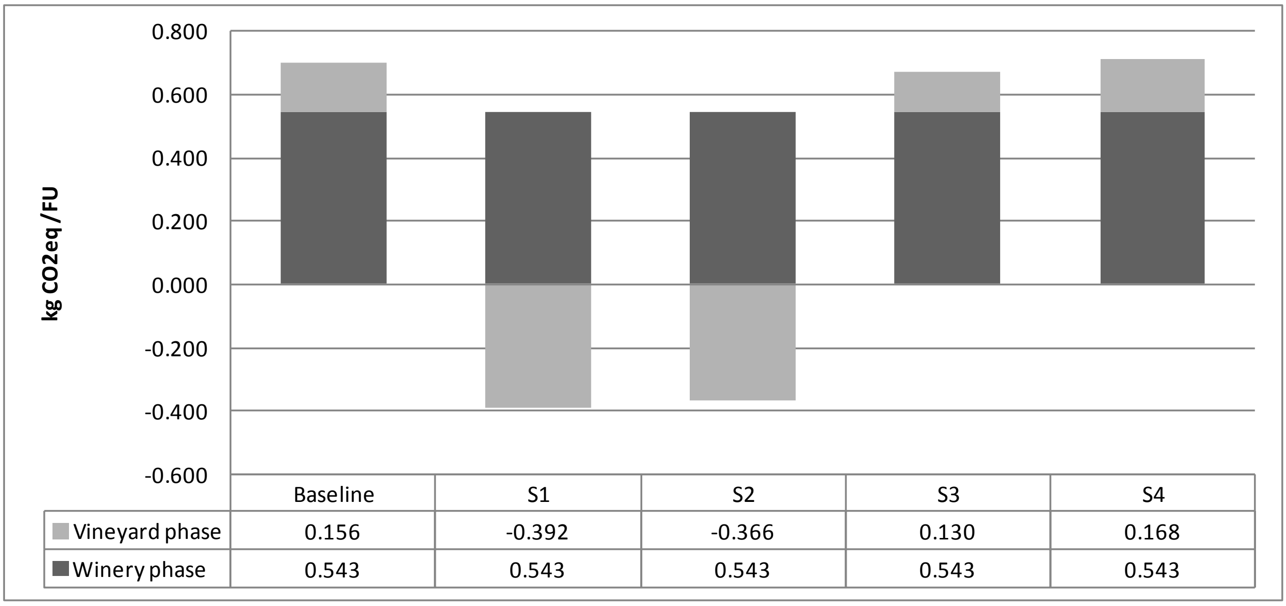


Figure 2. Soil effect in the selected scenarios on total carbon footprint, showed in vineyard phase and winery phase.

4. Discussion

The GWP of a 0.75 L bottle of red wine, was equal to 0.699 kg CO_2eq, which is comparable to results obtained in previous studies on wine CF (Notarnicola et al., 2003; Point 2009; Gazulla et al., 2010).
Here, the full SOM balance allowed us to evaluate vineyard management factors, such as fertilisation, grassing, tillage intensity and residue management and, consequently, to understand the importance of soil accounting in the CF of agricultural systems. The model used in this study revealed the positive impact of cover crops on SOM content, as an alternative to the use of inter-row harrowing for weed control, as reported by Parat et al., (2002) and Steenwerth and Belina (2008). Our results, ranging from -160 to 18 g C m^{-2} yr^{-1} for the S1 and S4, respectively, were comparable to the value of -15 g C m^{-2} yr^{-1} observed in a long-term study (30 years) conducted in a Californian-Mediterranean climate (Kroodsma and Field 2006).

Furthermore, this model for SOM change estimation needs a Minimum Data Set (MDS) for agricultural product CFs that accounts for soil carbon, thus establishing a standard for inventory. This MDS includes the following: physical and chemical soil characteristics (clay, SOM and limestone contents), climate parameters (average annual temperature), organic matter inputs and management practices.

5. Conclusion

In the context of the scientific debate on soil quality and SOM accounting, the approach proposed in this study can be regarded as a simplified methodology among other existing methodologies. Our results highlighted the need to consider soil in agricultural product CFs, indeed soil constitutes a major C pool in cropland, meaning there is scope for large amounts of C to be gained or lost from soils as a consequence of management practices. The method outlined in this study, based on the SOM balance, considered both OM input into the soil and organic output lost in the form of natural and agriculture-induced mineralisation. This simple and robust model was sensitive to changes in management practices.

Organic and conservation agriculture focus their attention to soil fertility and SOM content maintenance. Nevertheless, the application of the carbon footprint to these systems does not take into account all of the co-benefits that comes from having such systems. In this way, incorporating soil into the analysis can lead to the improved comparison of organic and conventional systems, highlighting the positive role of organic farming in the conservation of soil quality.

6. References

Bayer, C., Lovato, T., Dieckow, J., Zanatta, J., Mielniczuk, J., 2006. A method for estimating coefficients of soil organic matter dynamics based on long-term experiments. Soil Till. Res. 91, 217-226.

Bechini, L., Castoldi, N., Stein, A. 2011. Sensitivity to information upscaling of agro-ecological assessments: Application to soil organic carbon management. Agr. Syst. 104, 480-490.

Beck T., Bos U., Wittstock B., Baitz M., Fischer M., Sedlbauer K., 2010. LANCA Land Use Indicator Value Calculation in Life Cycle Assessment, p. 73, Fraunhofer IRB Verlag.

Bockstaller, C., Girardin, P. 2003. Mode de Calcul des Indicateurs Agrienvironmentaux de la Methode INDIGO. Version 1.61 du Logiciel. Unpublished INRA Internal Technical Report. Colmar, France.

Bockstaller, C., Guichard, L., Makowski, D., Aveline, A., Girardin, P., Plantureux, S. 2008. Agri-environmental indicators to assess cropping and farming systems. A review. Agron. Sustain. Dev. 28, 139-149.

Boiffin, J., Keli Zagbahi, J., Sebillotte, M. 1986. Systèmes de culture et statut organique des sols dans le Noyonnais: application du modèle de Hénin–Dupuis. Agronomie. 6, 437–446.

Bosco, S., Di Bene, C., Galli, M., Remorini, D., Massai, R., Bonari, E. 2011. Greenhouse gas emissions in the agricultural phase of wine production in the Maremma rural district in Tuscany, Italy. Ita. J. Agron. 6, 93-100.

Brandão, M., Milà i Canals, L., Clift, R., 2011. Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. Biomass Bioenerg. 35, 2323-2336.

Brentrup, F., Kusters, J., Kuhlmann, H., Lammel, J. 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology: I. Theoretical concept of a LCA method tailored to crop production. Eur. J. Agron. 20, 265-279.

BSI (2011) PAS 2050:2011 - Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. BSI British Standards, London.

Cowell, S.J., Clift, R., 2000. A methodology for assessing soil quantity and quality in life cycle assessment. J. Clean. Prod. 8, 321-331.

Cowell, S.J., Clift, R. 1997. Impact assessment for LCAs involving agricultural production Int. J. Life Cycle. Ass. 2, 99-103.

Culley, J.L.B. 1993. Density and compressibility. In: M.R. Carter (Ed.), Soil Sampling and Methods of Analysis, Louis Publisher, CRC, Boca Raton, FL, pp. 529–539.

Di Bene, C., Tavarini, S., Mazzoncini, M., Angelini, L.G., 2011. Changes in soil chemical parameters and organic matter balance after 13 years of ramie [Boehmeria nivea (L.) Gaud.] cultivation in the Mediterranean region. Eur. J. Agron. 35, 154-163.

GaBi5 (2012). GaBi 5 software. http://gabi-software.com

Gazulla C., Raugei M., Fullana-i-Palmer P., 2010. Taking a life cycle look at crianza wine production in Spain: where are the bottle-necks? Int. J. LCA 15, 330-337.

Hénin, S., Dupuis, M., 1945. Essai de bilan de la matière organique du sol. Ann. Agron. 15, 17–19.

Hillier, J., Whittaker, C., Dailey, G., Aylott, M., Casella, E., Richter, G., Riche, A., Murphy, R., Taylor, G., Smith, P. 2009. Greenhouse gas emissions from four bioenergy crops in England and Wales: Integrating spatial estimates of yield and soil carbon balance in life cycle analyses. GCB Bioenergy 1, 267-281.

IPCC, 2006. Prepared by the National Greenhouse Gas Inventories Programme. In: Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K. (eds): IPCC guidelines for national greenhouse gas inventories. IGES, Japan.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623-1627.

Mary, B., Guérif, J. 1994. Intérêts et limites des modèles de prevision de l'évolution des matières organiques et de l'azote dans le sol. Cah. Agric. 3, 247–257.

Meisterling, K., Samaras, C., Schweizer, V. 2009. Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat. J. Clean. Prod. 17, 222-230.

Milà I Canals, L., Burnip, G.M., Cowell, S.J. 2006. Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): Case study in New Zealand. Agr. Ecosyst. Environ. 114, 226-238.

Milà i Canals, L., Romanya, J., Cowell, S., 2007. Method for assessing impacts on life support functions (LSF) related to the use of "fertile land" in Life Cycle Assessment (LCA). J. Clean. Prod. 15, 1426-1440.

Mourad, A.L., Coltro, L., Oliveira, P.A.P.L.V., Kletecke, R.M., Baddini, J.P.O. 2007. A simple methodology for elaborating the life cycle inventory of agricultural products. Int. J. LCA 12, 408-413.

Muñoz, I., Campra, P., Fernández-Alba, A.R. 2010. Including CO_2-emission equivalence of changes in land surface albedo in life cycle assessment. Methodology and case study on greenhouse agriculture. Int. J LCA 15, 672-681.

Nelson, D.W., Sommers. L.E. 1982. Total carbon, organic carbon and organic matter, in: Page, A.L., Miller, R.H., Keeney, D.R. (Eds.), Methods of Soil Analysis, Part 2, Chemical and microbiological properties, Second edition. Agronomy Monograph 9, American Society of Agronomy, Madison, WI, pp. 539-579.

Nemecek, T., Erzinger, S. 2005. Modelling Representative Life Cycle Inventories for Swiss Arable Crops. Int. J. LCA 10, 1-9.

Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G. 2011. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agr. Syst. 104, 217-232.

Notarnicola B, Tassielli G, Nicoletti M (2003). LCA of wine production. In: Mattsonn B., Sonesson U. (eds): Environmentally-friendly food processing, pp. 306-326, Woodhead-Publishing and CRC Press, Cambridge-England, Boca Raton-USA.

Parat, C., Chaussod, R., Leveque, J., Dousset, S., Andreux, F. 2002. The relationship between copper accumulation in vineyard calcareous soils and soil organic matter and iron. Eur. J. Soil Sci. 53, 663-669.

Point E.V. 2008. Life cycle environmental impacts of wine production and consumption in Nova Scotia. MSc Thesis, Dalhousie University Halifax, Nova Scotia, Canada.

Ponsioen, T.C., Blonk, T.J. 2012. Calculating land use change in carbon footprints of agricultural products as an impact of current land use. J. Clean. Prod. 28, 120-126.

Reap, J., Roman, F., Duncan, S., Bras, B. 2008. A survey of unresolved problems in life cycle assessment. Int. J. LCA 13, 374-388.

Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T. 2009. A review of life cycle assessment (LCA) on some food products. J. Food Eng. 90, 1-10.

Saad R., Margni M., Koellner T., Wittstock B. and Deschênes L. 2011. Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. Int. J. LCA, 16, 198-211.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J. 2008. Greenhouse gas mitigation in agriculture. Philos. T. Roy. Soc. B. 363, 789-813.

Sofo, A., Nuzzo, V., Palese, A.M., Xiloyannis, C., Celano, G., Zukowskyj, P., Dichio, B. 2005. Net CO_2 storage in Mediterranean olive and peach orchards. Sci. Hortic. 107, 17-24.

Steenwerth, K., Belina, K.M. 2008. Cover crops enhance soil organic matter, carbon dynamics and microbiological function in a vineyard agroecosystem. Appl. Soil Ecol. 40, 359-369.

WRI/WBCSD, 2004. The Greenhouse Gas Protocol; a Reporting and Accounting Standard. World Resources Institute and World Business Council for Sustainable Development.

Life cycle assessment and eMergy application to the comparison between organic and conventional productions

Elena Neri[1,*], Francesca Rossetti[1], Benedetto Rugani[2], Federico M. Pulselli[3], Valentina Niccolucci[3], Simone Bastianoni[1], Nadia Marchettini[3]

[1] Ecodynamics Group, Dipartimento di Chimica, Università degli Studi di Siena, via A. Moro, 2 – 53100 Siena - Italy
[2] Public Research Centre Henri Tudor (CRPHT) / Resource Centre for Environmental Technologies (CRTE) - 66, rue de Luxembourg, P.B. 144, L-4002, Esch-sur-Alzette, Luxembourg
[3] Ecodynamics Group, Dipartimento di Chimica, Università degli Studi di Siena, via della Diana, 2A – 53100 Siena, Italy
[*] Corresponding author. E-mail: *elena.neri@unisi.it*

ABSTRACT

The present study proposes the parallel application of LCA and EMergy Evaluation (EME) to compare wine and olive oil production systems according to different agricultural management practices: conventional vs. organic. The purpose is to contribute to the research question on whether and how organic products constitute effectively a better choice both for consumers and producers in terms of environmental loadings. The parallel application of EME and LCA represents a strategic tool for a comprehensive interpretation of the issue under discussion. LCA outlines that major impacts, for both wine managements, are related to the packaging phase (average 50%), while the most detrimental phase in both olive oil productions is represented by the agricultural phase (average 60%). In both case conventional productions show higher impact values. EME outlines much larger intensive use of resources to produce a bottle of conventional wine and olive oil, highlighting the importance of local resources valorization and renewability of different production managements

Keywords: LCA, eMergy, conventional agriculture, organic agriculture, olive oil, wine.

1. Introduction

Organic productions aim at providing consumers with fresh and genuine products obtained through traditional agricultural managements, which usually avoid the use of synthetic fertilisers and rely on a wise consumption of resources and on the respect of natural cycles and pest controls, on crop rotations and green manure applications. However, discussions upon the different level of environmental sustainability between organic and conventional farms are still open. Can the management of a farm, by means of organic practices, actually guarantee its sustainability? The parallel application of two environmental accounting methodologies, i.e. LCA and EMergy Evaluation (EME), may represent a valid approach to obtain comprehensive outcomes and answers upon that issue, since LCA and EME have complementary features when assessing the overall environmental impacts of the whole production chain (Rugani et al., 2009). The aim of this paper is to apply LCA and EME to wine and olive oil from Tuscany (Italy), which are typical products of this region. Two types of agro-industrial productive managements, i.e. organic and conventional, are compared for these products in order to evaluate the differences in terms of environmental performance. Indeed, the proposed approach fits arguably in the current growing interest towards the improvement of environmental performances of agri-food products such as wine and olive oil, as recent studies demonstrate. Concerning olive oil productions several works approach the energetic and resource-oriented question (Guzman et al., 2008; Avraamides et al., 2008; De Gennaro et al., 2012). Otherwise Salomone et al., (2012) highlighted that, in order to understand and properly manage local food-supply chains in a sustainable manner, more specific chain-focused and regional-focused LCA studies in the olive oil industry are needed. Concerning wine productions, the introduction of the life cycle thinking is becoming more and more common, as the growing number of works shows (e.g. Notarnicola et al., 2003; Ardente et al., 2006; Petti et al., 2006; Point et al., 2012, Rugani et al., 2009; Vázquez-Rowe, 2012).

2. Methods

2.1 LCA and eMergy

For the two farms of wine and olive oil production, the LCA was conducted *from cradle to gate* (where gate is the bottling phase). The functional units were 1 L of bottled wine and 1 kg of bottled oil. LCI data were elaborated by means of the software SimaPro 7.3 and using the Ecoinvent database as background dataset, while the LCIA characterisation phase was determined using the CML 2 Baseline Method 2001 (Guinée et al., 2002). In order to be consistent with the aim of the present work, only impact categories strictly connected with the evaluation of environment impacts of the two productive processes were selected: Acidification (AP), Eutrophication (EP), Global Warming (GWP_{100}), and Photochemical Oxidation (POP) potentials. The other method used for the compared analysis was eMergy, that is "*...the quantity of solar energy neces-*

sary directly or indirectly to obtain a certain good or service." (Odum, 1996). In eMergy, the term Unit Emergy Value (UEV) (i.e. usually named *trasformity*,in seJ/J, or *specific eMergy*, in seJ/g), defines the eMergy necessary to obtain one unit of product or service (Odum, 1996), representing the coefficient through which different types of energy are converted to solar energy. Every biotic and abiotic resource in the geo-biosphere can be evaluated in eMergy terms and can be accounted for within the system's output. Emergy can be considered as an "energy memory", i.e. a memory of all solar energy necessary to support a defined system, considering the environmental work required previously for the production. In eMergy every life cycle input to the system is multiplied by the corresponding UEV and the resulting eMergy flows are added to obtain the total eMergy flow of the output. This last value is divided by the product of the considered process (i.e. annual quantity of olive oil and wine per hectare here) to obtain its UEV, which in the classical eMergy method is considered as an environmental performance's index for comparison of products: the higher the UEV, the higher is the equivalent solar energy demand per unit of product and thus lower is the life-cycle resource consumption's efficiency to generate that product. This study refers to the 9.26E+24 sej/J baseline (Campbell, 2000).

2.3 Case studies: wine and olive oil productions

The selected farms are all located in the centre of Tuscany. The organic wine farm (hereafter OW) had a vineyard of 10 ha and an average production of 3500L/ha per year (Chianti Colli Senesi wine). The conventional winery (hereafter CW) presented a semi-industrial management that covered an area of 120 ha and produced about 2200 L/ha of wine per year (Nobile di Montepulciano CGOD wine). CW presented a lower yield in wine production because of a rigorous selection of grapes; in fact, in order to improve the quality of the final product, only half of all the grapes harvested by CW are generally suitable for wine production. The two farms were selected because of the same price of the wine bottle at the market. For both farms the production chain can be divided in three phases: planting and production (Phase 1), wine-making and storage (Phase 2) and bottling (Phase 3). Concerning olive oil production, the organic farm (hereafter OO) extended for 4 ha, while the annual average yield of olives was 1500 kg/ha with annual production of 250 kg of oil. With regard to the conventional farm (hereafter CO), olive cultivation extended for 20 ha, whereas the conduction was directed towards an intensive production management, characterised by hard mechanisation and chemicals use. The annual mean production was 3820 kg/ha of olives, which are processed in 483 kg of oil per year (Protected Geographical Indication, PGI, quality). Similarly to wine, these two farms were also selected because of the same price of one olive oil bottle at market. However, the olive oil production was divided into two phases: agricultural phase (Phase 1) and oil mill phase, including bottling phase (Phase 2). Inventory data (the same used for LCA and EME) for both wine and olive oil productions are showed in the third and fourth columns, respectively, of the corresponding eMergy tables (see Tables 2 and 4).

3. Results

3.1. Wine LCA results

The comparison of the impacts characterisation of the two wine life cycles (Table 1) shows meaningful differences, highlighting that major values are connected to CW. The major departures were found for the packaging phase for both managements (average 50% for the 4 impact categories), with the higher values recorded by the CW. Concerning phase 1 (planting and production) and 2 (storage), the higher values are related to OW.

Table 1. LCIA of conventional (CW) and organic (OW) wine productions. Data reported for functional unit (1 L bottled wine).

Impact category	unit	conventional wine (CW)	organic wine (OW)
AP	kg SO_2 eq.	6,00E-03	4,00E-03
EP	kg PO_4 eq.	9,80E-04	6,40E-04
GWP_{100}	kg CO_2 eq.	7,30E-01	4,80E-01
POP	kg C_2H_4 eq.	2,40E-04	1,50E-04

3.2 Wine eMergy results

The EME output of the annual production was 9.25E+15 seJ/ha and 9.64E+15 seJ/ha for CW and OW, respectively (Table 2). However, the UEV (per bottle of wine) of the CW is almost twice the UEV of the OW (i.e. 4.22E+12 seJ/bottle and 2.75E+12 seJ/bottle, respectively). In Table 2 inventory data (columns 3, 4) and final calculation results are shown, divided for local resources, agricultural phase, processing phase and human labour.

Table 2. Emergy flows and UEVs for conventional (CW) and organic (OW) wine productions. Square bracket=references for UEV; Type of input: R= local renewable resources, N= local non-renewable resources, F=non-local purchased inputs.

						Emergy Fluxes (sej/ha/yr)	
Input	Units	Quantity/yr (CW)	Quantity/yr (OW)	Ref. UEV	TYPE OF INPUT	CONVENTIONAL WINE (CW)	ORGANIC WINE (OW)
LOCAL RESOURCES							
sunlight	J	8,49E+11	8,49E+11	[1]	R	8,49E+11	8,49E+11
rain	g	1,67E+08	1,67E+08	[2]	R	1,40E+13	1,40E+13
geothermal heat	J	7,16E+08	7,16E+08	[2]	R	4,98E+12	4,98E+12
loss of topsoil	J	1,03E+08	1,03E+08	[3]	N	7,45E+12	7,45E+12
PHASE 1							
fertilizer	g	1.21E+05		[3]	F	1,98E+15	
diesel	J	9,92E+09	1,75E+10	[5]	F	6,53E+14	1,15E+15
machinery	g	7,74E+03	1,68E+04	[7]	F	5,08E+13	1,03E+14
wood	g	6,02E+05	3,73E+05	[7]	50%R; 50%F	2,06E+10	1,28E+10
pesticides	g	1,04E+04	5,74E+04	[3]	F	1,51E+14	8,32E+14
concrete	g		3,57E+04	[12]	F		6,35E+13
PHASE 2							
water	g	3,26E+05	9,72E+07	[8]	F	5,70E+11	1,70E+14
electricity	J	8,67E+07	2,67E+08	[4]	F	1,03E+13	3,17E+13
chemicals	g	5,71E+02	3,50E+01	[3]	F	8,28E+12	1,30E+13
machinery	g	3,07E+04	3,89E+04	[7]	F	2,01E+14	2,30E+14
PHASE 3							
glass	g	1,78E+06	1,87E+06	[10]	F	5,49E+15	5,79E+15
cork	g	3,02E+04	4,83E+04	[10]	F	4,49E+13	7,19E+13
paper	g	8,58E+03	1,37E+04	[11]	F	4,25E+13	6,78E+13
glue	g	8,92E+02	1,43E+03	[10]	F	8,83E+12	1,42E+13
aluminium	g	4,49E+03	7,19E+03	[10]	F	2,65E+13	4,25E+13
machinery	g	2,10E+03	6,13E+03	[7]	F	1,38E+13	3,62E+13
electricity	J	1,89E+08		[4]	F	2,25E+13	
diesel	J		1,01E+09	[5]	F		6,65E+13
HUMAN LABOUR	h	1,73E+02	3,16E+02	[6]	10%R; 90%F	5,10E+14	9,33E+14
TOTAL						**9,25E+15**	**9,64E+15**
WINE (sej/g)	g	2,19E+06	3,50E+06			**4,22E+09**	**2,75E+09**
WINE (sej/BOTTLE OF WINE)						**4,22E+12**	**2,75E+12**

Literature references for UEV
[1] Odum H.T., 1996.
[2] Odum H.T.,et al. 2000.
[3] Brandt-Williams S. L., 2002.
[4] Tiezzi E. et al., 2001.
[5] Bastianoni S. et al., 2009
[6] Pulselli, R.M., et al., 2008.
[7] Brown M.T., Bardi E., 2001.
[8] Pulselli F. M.,et al., 2011.
[9] Campbell D.E. et al. 2002.
[10] Buranakarn, V. 1998.
[11] Tilley, D.R., 1999.
[12] Pulselli, R.M., et al., 2007.

3.3 Olive oil LCA results

Table 3 shows the LCIA comparison between the two production systems: likewise for wine, the CO presents the higher impacts, while regarding single phases, the agricultural phase (phase 1) presents higher impact values both for OO and CO.

Table 3. LCIA of organic and conventional olive oil productions. Data reported for functional unit (1 kg bottled olive oil).

Impact category	unit	conventional oil (CO)	organic oil (OO)
AP	kg SO_2 eq.	2,10E-01	6,00E-02
EP	kg PO_4 eq.	7,00E-01	1,00E-02
GWP_{100}	kg CO_2 eq.	2,64E+01	9,64E+00
POP	kg C_2H_4 eq.	1,20E-02	2,00E-03

3.4 Olive oil eMergy results

EME output is 2.50E+16 sej/ha/yr and 9.35E+15 sej/ha/yr as total eMergy flows and 5.18E+13 seJ/bottle and 3.74E+13 seJ/bottle as UEVs for CO and OO, respectively. In Table 4 inventory data (columns 3, 4) and final calculation results are shown, divided by local resources, agricultural phase, processing phase and human labour.

Table 4. Emergy flows and UEVs for conventional and organic oil productions. Square bracket=references for UEV; Type of input: R= local renewable resources, N= local non-renewable resources, F= non-local purchased inputs.

						Emergy Fluxes (sej/ha/yr)	
Input	Units	Quantity/yr (CO)	Quantity/yr (OO)	Ref. UEV	TYPE OF INPUT	CONVENTIONAL OIL (CO)	ORGANIC OIL (OO)
LOCAL RESOURCES							
sunlight	J	5,25E+13	5,25E+13	[1]	R	5,25E+13	5,25E+13
rain	g	7,90E+09	7,90E+09	[2]	R	6,64E+14	6,64E+14
geothermal heat	J	3,15E+10	3,06E+10	[2]	R	2,19E+14	2,13E+14
loss of topsoil	J	2,77E+09	2,77E+09	[3]	N	2,00E+14	2,00E+14
PHASE 1							
diesel	J	3,07E+09	3,74E+10	[5]	F	2,02E+14	2,46E+15
fertilizer	g	3,80E+05	7,40E+02	[3]	F	6,23E+15	1,75E+13
machinery	g	5,23E+04	8,85E+04	[7]	F	3,43E+14	5,81E+14
water	g	1,65E+08	1,27E+08	[8]	F	2,89E+14	2,22E+14
plastc, tyre	g	2,43E+04	8,97E+04	[9]	F	6,59E+13	2,43E+14
pesticides	g	9,17E+05	2,00E+04	[3]	F	1,33E+16	2,90E+14
electricity	J	8,25E+09	6,00E+09	[4]	F	9,81E+14	7,13E+14
PHASE 2							
machinery	g	1,54E+04	3,00E+04	[7]	F	8,98E+13	1,91E+14
diesel	J	4,78E+08	4,78E+08	[5]	F	3,14E+13	3,14E+13
water	g	2,27E+06	1,18E+06	[8]	F	3,97E+12	2,07E+12
electricity	J	9,35E+08	4,84E+08	[4]	F	1,11E+14	5,75E+13
plastc, tyre	g	4,88E+03	2,38E+04	[9]	F	1,32E+13	6,45E+13
glass	g	6,53E+05	3,37E+05	[10]	F	2,02E+15	1,04E+15
HUMAN LABOUR	h	8,50E+01	7,95E+02	[6]	10%R 90%F	2,51E+14	2,35E+15
TOTAL						**2,50E+16**	**9,35E+15**
OIL (sej/g)	g	4,83E+05	2,50E+05			**5,18E+10**	**3,74E+10**
OIL (sej/BOTTLE OF OIL)						**5,18E+13**	**3,74E+13**

Literature references for UEV
[1] Odum H.T., 1996.
[2] Odum H.T.,et al. 2000.
[3] Brandt-Williams S. L., 2002.
[4] Tiezzi E. et al., 2001.
[5] Bastianoni S. et al., 2009
[6] Pulselli, R.M., et al., 2008.
[7] Brown M.T., Bardi E., 2001.
[8] Pulselli F. M.,et al., 2011.
[9] Campbell D.E. et al. 2002.
[10] Buranakarn, V. 1998.
[11] Tilley, D.R., 1999.
[12] Pulselli, R.M., et al., 2007.

4. Discussion

4.1 LCA-EME of wine productions

In wine productions, the packaging phase (phase 3) presents major impacts both for the production systems analysed, due to the use of glass. Impact values related to CW are higher than those of the OW (Table 1) principally because the conventional system considered in this work used heavy and non-recycled glass. In contrast, the organic farm, thanks to the use of a lighter type of glass for bottles, as imposed by the European guidelines for organic productions, can reduce impacts related to this phase. These results are in accordance with Kavargiris et al., (2009) on the organic vs. conventional wine. Inputs linked to phase 1 and 2 are higher in the organic farm. CW uses chemical pesticides, effective at lower doses but with higher impact values. On the contrary, OW uses conventional sulphur and copper based pesticides, which are less effective and have to be used in higher quantities. Moreover, the CW in this study uses modern machineries, while in the OW case most of the work is done by hand, but the mechanisation, even if highly reduced, is quite old and less efficient.

EME results highlight highest values for OW's total eMergy flow, while CW presents highest value of UEV (Table 2). It means that CW has much larger intensive use of resources per bottle of product. This difference is due to the lower wine productivity of CW because of the rigorous selection of grapes. OW utilises more resources than CW: less efficient machineries, non-synthetic chemicals and all what concerns storage and bottling phase (more materials are necessary to have a higher quantity of wine). Furthermore, the total eMergy flow is higher for the organic production principally due to the higher annual human labour and consumption of diesel per hectare. Results related to eMergy flows grouped into macro-categories (Fig. 1) highlight how inputs are split and their weight on total eMergy flow (see the caption of Fig.1 for characteristics of each group). The category "other materials" (materials for the bottling phase) represents the highest percent of total eMergy flow for CW and OW respectively. While CW presents "chemical and fertilisers" as the second higher macro-category, OW presents energy contributions (diesel and electricity consumption). The slightly higher value of "human labour" in OW is because most of the work is done by hand. "Natural resources" represent less than 1% of total flows for both the two farms (not shown in the figure below).

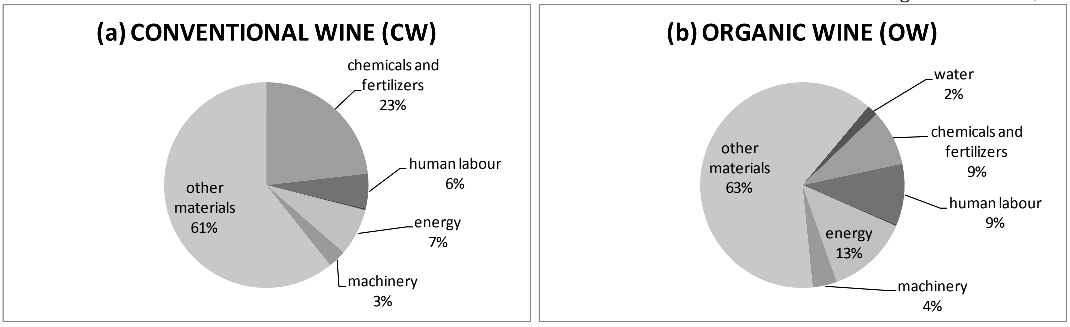


Figure 1. Emergy flows of conventional (a) and organic (b) wine production grouped for macro-categories: chemicals and fertilisers= pesticides, fertilisers and chemicals for fermentation; human labour; natural resources= rain and geo-heat; loss of topsoil; energy= diesel and electricity; other materials= glass, cork, paper, glue; water; machinery= steel, iron, aluminium involved in farm operations, transports and processing.

By summing different type of resources (R, N and F) (Table 2), both conventional and organic farms show the use of 1% renewable resources. Emergy flows are divided in four parts (Table 2). Results highlight highest values for “phase 3” both in CW (61.11%) and OW (63.13%), followed by “phase 1” (30.70% for CW and 22.30% for OW), “human labour”, “phase 2” and “local resources”. The “packaging phase” (phase 3) has bigger relevance because of the higher materials use.

4.2 LCA-EME of olive oil productions

Concerning LCA results of the olive oil productions, data demonstrate that the major impacts are associated to the conventional system (Table 3). Comparing the systems by phases, both productions are characterised by higher values related to the agricultural phase. In OO, impacts connected to phase 1 are basically due to the higher fuel consumptions, which depend from old and less efficient machineries, while in CO the use of chemicals represents the major impact of phase 1. These data agree with literature results derived from olive oil LCA studies (e.g. Fiore et al., 2009). In the packaging phase the glass is once more the main impact for both the systems; moreover, in our case studies, all farms use heavy glass bottles. Regarding EME, the higher environmental performances are explicitly obtained by applying organic practices because of the lower use of chemicals, which extremely reduces the indirect contribution of solar energy provision (Table 4). While CO presents an intensive use of resources (e.g. fertiliser, pesticides, electricity). The difference between CO and OO values is reduced in the UEVs because of the less oil yield (productivity) in organic farm (CO=483 kg/ha/yr vs. OO=250 kg/ha/yr). Fig. 2 shows all eMergy flows grouped into macro-categories (see the caption of Fig.2 for characteristics of each group). In CO the higher contribution is related to “chemicals and fertilisers”, followed by “other materials” and “energy”. Other macro-categories have values lower than 5%. Instead, OO presents a less homogeneous distribution of macro-categories: “energy” and “human labour” represent main flows, followed by “machinery” and “other materials”. “Natural resources” consists of 10% of flows, a high percent in respect to CO. It is important to remark that machinery used in organic farm are aged (and less efficient).

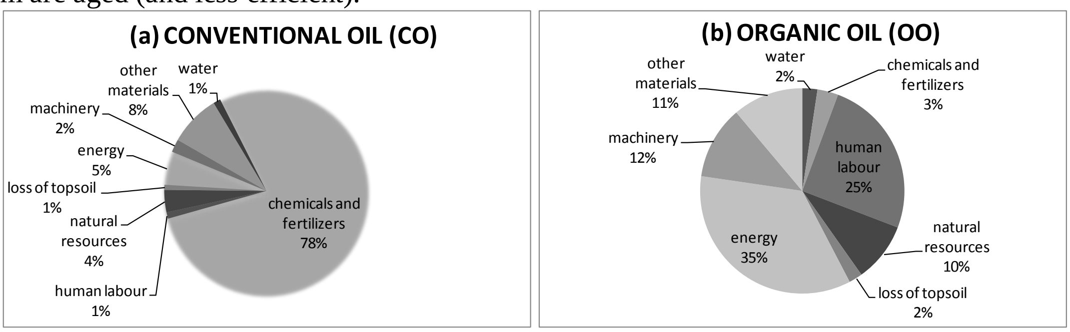


Figure 2. Emergy flows of conventional (a) and organic (b) oil production grouped for macro-categories: chemicals and fertilisers= pesticides and fertilisers; human labour; natural resources= rain and geo-heat; loss of topsoil; energy= diesel and electricity; other materials= glass; water; machinery= steel, iron, aluminium, cast iron involved in farm operations, transports and olive processing.

By summing different types of resource (R, N and F) (Table 4), the CO highlights the use of 4% renewable resources, while OO 12%. By gathering eMergy flows in four parts (Table 4), the agricultural phase

(phase 1) becomes the highest in both the two case studies. Indeed, it represented 85.59% of CO total flow (due to a large use of fertilisers and pesticides) and 48.49% of OO total flow (due to the use of obsolete and less efficient machineries that leaded to high consumption of e.g. diesel, electricity). "Phase 2", "local resources" and "human labour" have minor importance on total CO eMergy flow (9.07%, 4.33% and 1% respectively). On the contrary, "human labour" represents the 25.10% of OO total eMergy, followed by "phase 2" (14.88%) and "local resources" (11.53%).

5. Conclusion

The approach introduced in the present study fits in the current growing interest towards the improvement of environmental performances of agri-food products. Currently, the promotion of sustainability is gaining further significance and companies are becoming more and more sensitive to the environment safeguarding (e.g. developing eco-friendly products). This trend calls for integrated approaches concerning assessments more concentrated on the overall production systems instead of assessments focused on the single processes of the life cycle. Thus, the minimisation of environmental impacts by the mere use of cleaner production technologies is no more sufficient. This study, through the parallel application of LCA and EME, provides a large overview about the environmental sustainability of farm management choices. Outcomes from the analysis agree with current literature findings and the combination of these two different approaches can lead to more comprehensive and significant results, suitable for policy recommendations and improvement decisions. The added value of this study consisted in identifying, through LCA results, weak points in environmental terms and which process phases were improvable for the eco-profile of the final product, in addition to the importance of local resources valorisation and renewability of different production managements, as highlighted by EME. This approach points out that not always organic conductions mean best environmental performances, efficiency and higher sustainability. Obviously, a deeper analysis is necessary to better understand the organic vs. conventional topic, including socio-economic aspects in which EME can constitute a valid framework.

6. References

Ardente, F., Beccali G., Cellura, M. and Marvuglia, A., 2006. POEMS: A Case Study of an Italian Wine-Producing Firm. Environ. Manag., 38 (3), 350–364.

Avraamides M., Fatta D., 2008. Resource consumption and emissions from olive oil production: a life cycle inventory case study in Cyprus. J Clean Prod 16, 809-821.

Campbell, D.E., 2000. A revised solar trasformity for tidal energy received by the earth and dissipated globally: implications for emergy analysis. In: Brown, M.T. (Ed.), Emergy Sinthesis; Theory and Application of the Emergy Methodology. The Center for Environmental policy, Gainsville, pp. 255-263.

De Gennaro B., Notarnicola B., Rosselli L.,Tassielli G., 2012. Innovative olive-growing models: an environmental and economic assessment. J Clean Prod 28, 70-80.

Fiore M., Breedveld L., Arrivas Bajardi C., Giaimo L., Notaro A., 2009 Certificazione ambientale di prodotti agroalimentari. LCA dell'olio d'oliva. ARS n. 122 - luglio / settembre 2009.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., Koning, A. de, Oers, L. van, Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., Bruijn, H. de, Duin, R. van, Huijbregts, M.A.J., 2002. Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background. Kluwer Academic Publishers, Dordrecht.

Guzman G.I., Alonso A.M., 2008. A comparison of energy use in conventional and organic olive oil production in Spain. Agricultural System 98, 167-176.

Kavargiris, S.E., Mamolos, A.P., Tsatsarelisb, C.A., Nikolaidou, A.E., Kalburtji, K.L., 2009. Energy resources utilization in organic and conventional vineyards: Energy flow, greenhouse gas emissions and biofuel production. Biomass and Bioenergy 33, 1239–1250.

Notarnicola B., Tassielli G., Nicoletti G.M., 2003. LCA of wine production. In B. Mattsonn, U. Sonesson (editors): Environmentally-friendly food production, Cap. XVII, pg. 306-326. Woodhead-Publishing, Cambridge, England and CRC Press Boca Raton, USA, ISBN 1 85573 677 2.

Odum, H.T., 1996. Environmental Accounting: Emergy and Environmental Decision making. Wiley & Sons, new York, USA.

Petti, L., Raggi, A., De Camillis, C., Matteucci, P., Sára, B., Pagliuca, G., 2006. Life cycle approach in an organic wine-making firm: an Italian case-study. Proceedings Fifth Australian Conference on Life Cycle Assessment, Melbourne, Australia.

Point, E., Tyedmers, P., Naugler, C., 2012. Life cycle environmental impacts of wine production and consumption in Nova Scotia, Canada. J. Clean. Prod. 27, 11-20.

Rugani, B., Niccolucci, V., Pulselli, R. M., Tiezzi, E., 2009. A cradle-to-gate Life Cycle Assessment integrated with EME: sustainability analysis of an organic wine production. In: Proceedings of the SETAC Europe 19th Annual Meeting, 31st May-4th June 2009, Göteborg, Sweden, p. 274.

Salomone R., Ioppolo G. 2012. Environmental Impacts Of Olive Oil Production: A Life Cycle Assessment Case Study In The Province Of Messina (Sicily). J. Clean. Prod. 28, 88-100.

Vázquez-Rowe, I., 2012. Joint life cycle assessment and data envelopment analysis of grape production for vinification in the Rías Baixas appellation (NW-Spain). J. Clean. Prod. 27, 92-102.

Assessment of digestibility improving enzymes potential to reduce greenhouse gas emissions in broiler production

Anja Marie Bundgaard[1,*], Randi Dalgaard[2], Mikkel Thrane[3], Trakarn Prapaspongsa[1,4]

[1] Aalborg University, Department of development and planning, Vestre Havnepromenade 9, 9000 Aalborg, Denmark
[2] 2.-0 LCA consultants, Skibbrogade 5,1, 9000 Aalborg, Denmark
[3] Dupont Nutrition and Health, Danisco, Edwin Rahrs Vej 38, DK-8220 Brabrand, Denmark.
[4] Mahidol University, Department of civil and environmental engineering, 25/25 Puttamonthon, Nakorn Pathom, 73170, Thailand
Corresponding author. E-mail: *anjamarieb@gmail.com*

ABSTRACT

The objective of this study was to examine the potential of digestibility improving enzymes to reduce greenhouse gas (GHG) emissions in broiler production. The product examined was a new enzyme called Axtra XAP, developed by DuPont, Danisco Animal Nutrition. Two scenarios were compared: one where Axtra XAP was not included in the diet and one where Axtra XAP was included in the diet. Axtra XAP facilitated higher inclusion rates of cheaper (and possibly more environmentally friendly) feed ingredients that have a lower nutritional value in the diet. Axtra XAP's environmental improvement potential was documented through a Life Cycle Assessment (LCA) by applying a consequential approach including indirect land use changes (ILUC). The findings showed that Axtra XAP could reduce GHG emissions from broiler production by 5%. A sensitivity analysis was conducted to assess the robustness of the results and it showed that the result varied substantially. The most important parameters were the inclusion or exclusion of ILUC and changes in the feed formulation.

Keywords: broilers, enzymes, consequential LCA, livestock feed, greenhouse gas emissions

1. Introduction

Food production represents around 1/3 of all human induced greenhouse gas (GHG) emissions (Foley et al., 2011) and together with the continuous growth in population and affluence level, this represents a major challenge for the earth's ecosystems. The production of animal products for human consumption amounts to 18% of the total anthropogenic GHG emissions (FAO, 2006). No single solution exists which can address all environmental problems caused by the livestock sector, but the use of digestibility improving enzymes in animal feed has the potential to be an important solution. When enzymes are used in animal feed, they help break down parts of the diet which the animal itself cannot break down in its digestive system. Therefore nutrients from the diet are released which previously were unavailable to the animal. In addition enzymes can help to remove the anti-nutrient effect of certain components in the diet. The enzymes can provide an improved growth performance for a specific diet, or allow for a higher ratio of cheaper feed ingredients with a lower nutritional value, while achieving the same growth performance (Barletta, 2011). Previous studies have documented the considerable environmental advantages of applying enzymes, such as phytase and other digestibility improving enzymes, in animal feed (Nielsen and Wenzel, 2007; Nielsen at al., 2008). The objective of the present study was to examine the possible environmental improvement potential of digestibility improving enzymes used in complex broiler diets. The study examined a specific enzyme product, Axtra XAP, developed by DuPont, Danisco Animal Nutrition. The paper accounts for the main results of the study.

2. Methods

The two feed formulations used in this study were formulated using a computer software tool that ensures the lowest feed cost while still providing the necessary nutrition for the animal. They are economically optimised according to US feed prices in 2011. Methane and nitrous oxide emissions caused by the broiler's manure and the manures end-use as an organic fertiliser are calculated according to IPCC (2006a; 2006b). Enteric fermentation from the broiler was omitted, as no standard exists for poultry in the guidelines and because enteric fermentation from poultry is very limited. A consequential approach (Weidema, 2003) was applied and indirect land use changes (ILUC) were included. The approach used to calculate the effect of ILUC was developed by Schmidt et al., (2011). The main principle in the model is that the current use of land reflects the current demand for land, and that changes in demand for land will result in changes in land use. The life cycle assessment (LCA) was modelled using the program SimaPro and it basically follows the ISO 14040 and 14044 standards.

2.1. Goal and Scope

Axtra XAP increases the digestibility of the feed thereby increasing the broiler's utilisation of the nutrients in the feed. When Axtra XAP is used in the feed formulation it is possible to include a higher amount of cheaper feed ingredients that have a lower nutritional value without compromising the growth of the broiler. The study strived to assess the changes in GHG emissions when a broiler was reared using a feed formulation without Axtra XAP compared to when a broiler was reared using a feed formulation with Axtra XAP. The functional unit applied in the study was the production of one kg live broiler, assuming that the production of one kg live broiler meat required 1.8 kg feed. The two feed formulations represented US commercial examples in 2011. In terms of life cycle impact assessment, the study addresses GHG emissions based on the IPPC 2007 GWP 100a impact method. As the nature of the LCA was comparative (comparing two scenarios), identical life cycle stages and processes in the two scenarios were omitted. Axtra XAP facilitated an increase in the digestibility of the feed, resulting in changes in the feed formulation and the manure emitted by the broiler. Changes in the manure resulted in changed methane and nitrogen oxide emissions from the manure and therefore changes in the end use of the manure.

2.2. Feed formulations

The effect of using Axtra XAP was analysed in eight feed trials made in collaboration with research institutions in different parts of the world (Danisco, no data). Between 144 and 1800 broilers were included in each feed trial. The results showed a reduction in the feed used to rear the same amount of broiler meat of 3.8% to 8.7%, depending on the feed formulation. The results of the feed trials were incorporated into the computer software used to formulate the two feed formulations. The feed formulations used in scenario one and scenario two are presented in table 1 and follow Dupont, Danisco Animal Nutrition's recommendations for using the Axtra XAP product.

Table 1. Feed formulations provided by DuPont, Danisco Animal Nutrition

Raw materials	Scenario one feed composition without Axtra XAP (g)	Scenario two feed composition with Axtra XAP (g)
Corn	1,171.8	1,162.9
DDGS	180.0	216.0
Phyzyme XP	0.4	0.4
Soya bean meal	347.2	334.3
Meat and bone meal	52.9	35.4
L-Lysine (HCL)	5.7	6.3
DL-Methionine	4.2	4.2
NaCL	6.0	7.0
Limestone	15.2	19.9
Dicalcium phosphate	1.4	5.1
Vitamins/ Minerals	1.8	1.8
Axtra XAP 102 TPT	0	0.9
Pig and poultry fats	11.9	4.5
L-threonine	1.5	1.3
Total	1,800	1,800

The improvement in digestibility and performance obtained by applying Axtra XAP was used to reduce the amount of corn, soybean meal, meat and bone meal, and pig and poultry fats in the diet and to increase the amount of dried distillers grains with solubles (DDGS) in the feed. This provided an economic benefit of 4.04 $ per ton of feed in scenario two compared to scenario one. A minimum level of fat is required in the feed formulation to ensure pellet quality, therefore, it was not possible to apply the full Axtra XAP recommendations for energy reduction in the feed formulation. Hence, in scenario two there is an excess amount of energy provided by the enzyme (48 kcal per kg feed) compared to scenario one. The excess energy is not accounted for in the study and the LCA therefore represents a conservative estimate of Axtra XAP's potential.

3. Results

3.1 Life Cycle Inventory

Figure 1 provides an overview of the system boundary including marginal mechanisms. Corn was not considered a constraint product and it was modelled as corn produced in the US, based on data from Dalgaard (2011). In scenario two, the amount of corn in the diet was decreased, resulting in less demand for corn production compared to scenario one. Soybean meal was not considered a constraint product either (Dalgaard, 2008), and it was therefore modelled as soybean meal produced in Brazil (Dalgaard, 2011). In scenario two less soybean meal was needed in the feed formulation compared to scenario one, reducing the demand for soybean meal.

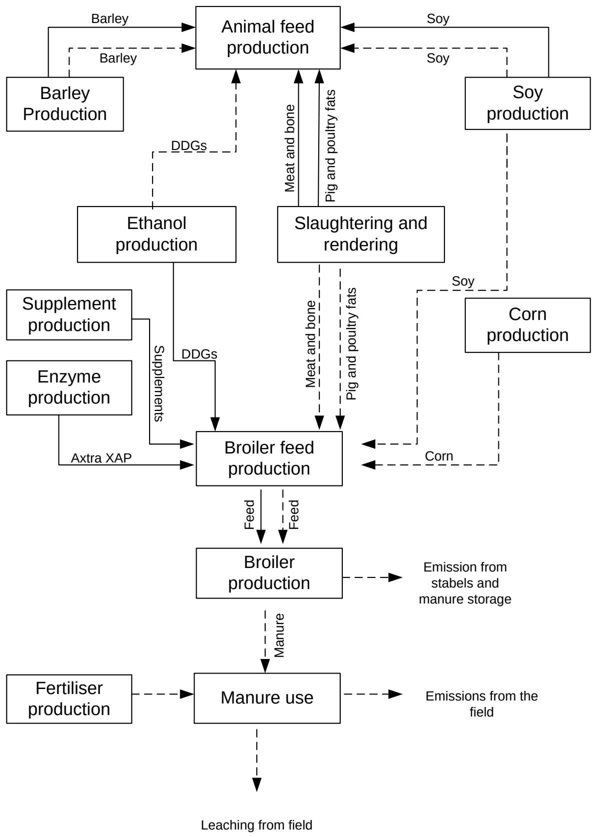


Figure 1. System boundary including marginal mechanisms. The dotted and full lines illustrate the changes in demand for different products between scenario one and two. The dotted line illustrates a decrease in demand for the product and the full line illustrates an increase in demand for the product in scenario two compared to scenario one.

DDGS is a constraint by-product from bio ethanol production. DDGS is predominately used in livestock feed (Saunders and Rosentrater, 2008). An increased demand for DDGS will not lead to an increased production of DDGS, as this is determined by the demand for ethanol, instead, it will result in an increased production of the marginal feed energy and the marginal feed protein. According to Weidema (2003), the marginal feed protein is soybean meal, and the marginal feed energy is barley. Data on soybean meal and barley was based on Dalgaard (2011). In scenario two, an increased ratio of DDGS was used in the feed formulation compared to scenario one, resulting in less DDGS available for other animal feed production, and thereby an increased demand for barley and soybean meal. Meat and bone meal was also considered a constraint by-product. The majority of rendered protein products in the US and Canada are used as animal feed (Jakanowski, 2011). In scenario two, the amount of meat and bone meal was decreased in the feed formulation compared to scenario one. This will result in more meat and bone meal available for the rest of animal feed production, thereby decreasing the demand for the marginal feed energy and the marginal feed protein. In the feed formulations pig and poultry fat was used as the fat source, but it could be any other suitable source of fats and greases. Rendered fats and greases were also considered a constraint by-product. Rendered fats and greases have several uses the main uses being as a feed ingredient, in the oleochemical industry and as a biofuel. The livestock industry is the largest user of rendered fats and greases, but the biodiesel production industry might be the fastest growing market for rendered fats and greases (NRA, 2011a; 2011b). In this study the livestock industry was used as the marginal use of rendered fats and greases. In scenario two, less pig and poultry fats were needed compared to scenario one, making more pig and poultry fat available for the rest of animal feed production, decreasing the demand for the marginal feed energy. In scenario two there was an increased demand for both supplements and enzymes compared to scenario one. The supplements were modelled using ecoinvent processes (Ecoinvent, 2007). Phyzyme was modelled using a fixed GHG emissions rate of 5.0 kg CO_2 per kg enzyme produced (Dalgaard, 2011). Axtra XAP was modelled based on production formulas and represents 90-100% of the actual ingredients and materials. Data for energy and material use from the production of Axtra XAP was from the enzyme factories' production records. Data on the ingredient production was from Ecoinvent (2007), LCA food database (2003), suppliers and literature. Transportation of raw materials was included for the most important materials. The end-use of the manure was assumed to be as an organic fertiliser. The increased digestibility of the feed ingredients, facilitated by Axtra XAP, resulted in less methane emissions from the manure in scenario two compared to scenario one from both the broiler sheds and the manure storage. The broiler's increased nitrogen retention in scenario two compared to scenario one, again as a result of the use of Axtra XAP, results in less nitrogen in the manure. This leaded to decreased nitrous oxide emissions from the sheds, the manure storage and from the fields. However, when the manure contained less nitrogen it then substituted less marginal nitrogen fertiliser. Thereby, the demand for the marginal nitrogen fertiliser increased. In the study, ammonium nitrate was assumed to be the marginal nitrogen fertiliser, as it is the dominant nitrogen fertiliser in Europe (Sonesson et al., 2009). Inorganic nitrogen fertiliser was modelled as ammonium nitrate. It was assumed that poultry manure offsets the use of inorganic nitrogen with 70% (FVM, 2011). Data used to model the processes was based on three databases; the ecoinvent database (Ecoinvent, 2007), the LCA food DK database (LCA food, 2003) and Danisco's database (Dalgaard, 2011). When using different databases, there will be unavoidable differences in the datasets and methodological choices. However, the majority of all the feed ingredients were based on data from the Danisco database (97% of the total weight) resulting in data and methodological consistency.

3.2 Results of the life cycle impact assessment

The results are presented in Fig. 2. The results of the LCA are given as a difference between scenario one and scenario two per FU, representing the changes induced by Axtra XAP. The total reduction in GHG emission per FU when applying Axtra XAP amounts to 90.5 g CO_2 eq. (dark blue column).

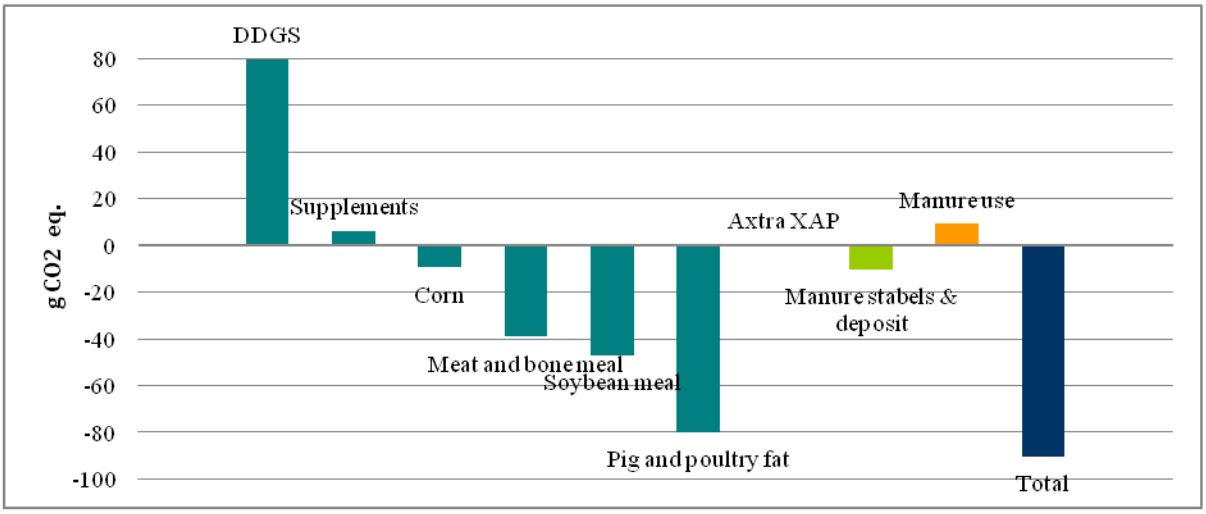


Figure 2. Changes in GHG emission caused by the use of Axtra XAP in scenario two compared to scenario one. The negative value indicates a decrease in GHG emissions and a positive value indicates an increase in GHG emissions in scenario two compared to scenario one.

The use of Axtra XAP in the feed formulation resulted in decreased GHG emissions from pig and poultry fat, meat and bone meal, soybean meal and corn, and increased GHG emissions from DDGS and the supplements amounting to a reduction in GHG emissions of 90.0 CO_2 eq. per FU. The increased GHG emissions from the supplements were mainly driven by the increased level of dicalcium phosphate in scenario two. The use of Axtra XAP resulted in an increase in GHG emissions of 0.85 g CO_2 per FU (red column). GHG emissions from manure management and storage were reduced by 10.4 g CO_2 eq. per FU (green column). GHG emissions from the use of the manure as a crop fertiliser were also reduced, as a consequence of the lower nitrogen content in the manure. However, when less organic nitrogen was emitted by the broiler to be used as fertiliser then less inorganic nitrogen fertiliser was displaced, resulting in an increase in GHG emissions of 9.1 g CO_2 eq. per FU.

3.3 Sensitivity analysis

In the sensitivity analysis the effects from changes in the feed formulation, the exclusion of ILUC, changes in the marginal use of rendered fats and greases and the impact from the production of soybean meal were examined. The most significant parameters proved to be changes in the feed formulations and the exclusion of ILUC. As mentioned earlier, it was not possible to apply the full benefits of Axtra XAP in scenario two. Therefore, two additional feed formulations were made, one without Axtra XAP and one with Axtra XAP, where no restrictions were made on the fat content, allowing for the full benefits of Axtra XAP to be applied. Changes in the feed formulations resulted in almost a tripling (281%) of the savings in GHG emissions indicating that either the feed formulations were too conservative or there is a large unutilised potential for further improvement. The exclusion of ILUC resulted in a reduction in the savings in GHG emissions of 93% compared to the reference scenario. It must be concluded that there are considerable uncertainties connected with both the methodological choices made and the data collection.

4. Discussion

To evaluate the savings facilitated by Axtra XAP it was compared to the total GHG emissions from the production of broiler meat. In total four studies were examined: LCA food (2003), Cederberg (2009), Williams et al., (2009) and Nielsen et al., (2011). The results from these studies varied considerably from approximately 1.9 kg CO_2 per kg bone free meat in Cederberg (2009) to approximately 3.4 kg CO_2 per kg bone free meat in Williams et al., (2009), assessing that the carcass weight made up 70% of live weight and bone free meat made up 77% of the carcass weight. To make a conservative estimate of Axtra XAP's improvement potential the highest GHG emissions from the studies reviewed was used. The calculations showed that savings in GHG emissions facilitated by Axtra XAP amounted to 4.9% per kg bone free broiler meat. In 2010, the total production of chicken meat was 86,064 million ton (FAO, 2011), thus Axtra XAP could potentially facilitate savings in GHG emissions of roughly 14 million ton CO_2 eq. This assumed that all producers used the same feed formulation, applied Axtra XAP to their diets and that digestibility improving enzymes were not already used in the diets. The results should however be applied on a global scale with caution, as consequential modelling accounts for marginal changes, and does not represent what will happen on the market if all broiler producers changed their feed formulation. Additionally, the savings in GHG emissions will depend on the prices of the raw ingredients, as the feed formulations are made including only consideration to the price and the nutritional value of the feed. As documented in the sensitivity analysis, the feed formulation was one of the most important parameters when determining Axtra XAP's potential. Thus, the savings facilitated by Axtra XAP could potentially change considerably depending on the feed prices. Furthermore, Axtra XAP will reduce production cost and most likely reduce the price of broiler meat, potentially resulting in an increasing consumption of broiler meat (rebound effect).

5. Conclusion

Axtra XAP facilitated savings in GHG emissions of 90.5 g CO_2 eq. per FU. Comparing the potential savings in GHG emission documented in the present LCA, with the total greenhouse gas emissions from the production of one kg bone free broiler meat documented in Williams et al., (2009), showed that Axtra XAP had the potential to reduce the impact from broiler rearing by 5% resulting in a global potential saving of 14 million ton CO_2 eq. each year. The results of the sensitivity analysis showed that the most significant parameters were the inclusion or exclusion of ILUC and changes in the feed formulation. However, all sensitivity analysis resulted in a reduction in the GHG emissions when Axtra XAP was included in the feed formulation compared to when it was not.

6. References

Barletta, A., 2011. Introduction: Current market and expected developments, in Bedford, M.R., Partridge G.G. (Eds), Enzymes in Farm Animal Nutrition, 2nd Edition, pp. 1-11, CAB International, Bodmin, UK

Cederberg, C., Sonesson, U., Henriksson, M., Sund, V., Davis J., 2009. Greenhouse gas emissions from Swedish production of milk and eggs 1990 and 2005. http://www.sik.se/archive/pdf-filer-katalog/sr793.pdf. (accessed 29.12.11)

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M. and Pengue, W.A., 2008. LCA of Soybean Meals. Int J LCA 13 (3) 240-254.

Dalgaard, R., 2011. Inventory of key feedstock. Developed by Randi Dalgaard, 2.0 LCA consultant, Denmark. Available on request (mthr@danisco.com)

Danisco, no data. Technical Report Axtra XAP: Axtra XAP significantly improved performance and AME of broiler feed corn/soy/gain by-product based diets. (internal document)

Ecoinvent, 2007. The lifecycle inventory data v2.2 Swiss Centre for Life Cycle Inventories.

FAO, 2006. Livestock's long shadow: environmental issues and options, Food and Agriculture Organization of the United Nations. ftp://ftp.fao.org/docrep/fao/010/a0701e/a0701e.pdf. (Accessed 22.11.2011)

FAO, 2011. Food and Agriculture Organization of the United Nations: FAPSTAT: Production: Livestock Primary. http://faostat.fao.org/site/569/DesktopDefault.aspx?PageID=569#ancor (accessed 22.12.11)

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D. and Zaks, D.P.M., 2011. Solutions for a Cultivated Planet. Nature Volume: 478 (p 337–342).

IPCC, 2006a. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4. Agriculture, Forestry and other Land Use, Chapter 10. N2O Emissions from Livestock and Manure Management.

IPCC, 2006b. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4. Agriculture, Forestry and other Land Use, Chapter 11. N2O Emissions from Managed Soils and CO_2 Emissions from Lime and Urea Application.

Jekanowski, M., 2011. Survey Says: A Snapshot of Rendering. Render Magazine. April 2011, 58-61.

LCA food, 2003. LCA food database. www.lcafood.dk. (accessed 28.12.11)

FVM, 2011. Ministeriet for Fødevarer, Landbrug og Fiskeri Vejledning om Gødsknings- og Harmoniregler: Planperiode 1. august 2011 til 31. juli 2012. Plantedirektoratet. http://www.netpublikationer.dk/FVM/978-87-7083-418-6/index.htm (accessed 18.12.11)

Nielsen, P.H., Dalgaard, R., Korsbak, A., Pettersson, D., 2007. Environmental Assessment of Digestibility Improvement Factors Applied in Animal Production: Use of Ronozyme® WX CT Xylanase in Danish Pig Production. Int J LCA 13(1), pp. 49-56

Nielsen, P.H., Wenzel, H., 2007. Environmental Assessment of Ronozyme®P5000 CT Phytase as an Alternative to Inorganic Phosphate Supplementation to Pig Feed Used in Intensive Pig Production. Int J LCA 12 (7), pp. 514-520

Nielsen, N.I., Jørgensen, M., Bahrndorff, S., 2011. Greenhouse Gas Emissions from the Danish Broiler Production estimated via LCA Methodology. The Danish Food Industry Agency. http://www.landbrugsinfo.dk/Fjerkrae/Sider/110628_vfc_engelsk_vfl_layout_web.pdf?List=%7Bfd90c3be-edf2-456f-bae3-f22a04409221%7D&download=true (accessed 18.08.2012)

NRA. 2011a. Economic impact: Statistics: Monthly U.S. Raw Material Usage for U.S. Biodiesel Production (2007-2009). http://assets.nationalrenderers.org/Monthly_US_Raw_Material_Useage_for_US_-Biodiesel_Production_2007_2009.pdf. (Accessed 08.12.11)

NRA (2011b) Economic impact: Statistics: US and EU Production of Biodiesel (2001-2009). http://nationalrenderers.org/assets/4d2db2f3dabe9d618f00005c/us_and_eu_production_of_biodiesel_20012009.pdf. (Accessed 08.12.11)

Schmidt, J.H., Reinhard, J., Weidema, B., (2011) Modelling of indirect land use changes in LCA: Report v2I. Denmark, Aalborg. 2.0 LCA consultants

Saunders, J.A., Rosentrater, K.A., 2009. Survey of US fuel ethanol plants. Bioresource Technology 100, 3277-3284

Sonesson, U., Cederberg, C., and Berglund M., 2009. Greenhouse gas emissions in animal feed production. http://www.klimatmarkningen.se/wp-content/uploads/2009/12/2009-2-feed.pdf. (accessed 23.11.11)

Weidema, B., 2003. Market information in life cycle assessment. Danish Environmental Protection Agency: Danish Ministry of the Environment. http://www.norlca.org/resources/780.pdf. (accessed 08.12.11)

Williams, A.G., Audsley, E. Sandars, D.L., 2009. A Lifecycle Approach to Reducing the Environmental Impact of Poultry Production. World Poultry Science Association (WPSA), 17th European Symposium on Poultry Nutrition, August 2009 Edinburgh, Scotland

Comparison of two production scenarios of chickens consumed in France

Vamilson Prudêncio da Silva Junior[1,*], Edivan Cherubini[2], Sebastião Roberto Soares[2]

[1] Empresa de Pesquisa Agropecuária e Extensão Rural de Santa Catarina, Florianópolis, Brazil.
[2] Universidade Federal de Santa Catarina – CICLOG, Florianópolis, Brazil.
[*] Corresponding author. E-mail: *vamilson@epagri.sc.gov.br*

ABSTRACT

The use of the Life Cycle Assessment - LCA - provides interesting comparisons between different scenarios of producing a same product. In this study we investigate if imported chickens from Brazil, fed with locally produced grains cause less or more impact than chickens produced in France, using a feed part of which comes from Brazil. We assume that the chicken produced in France (FR) was a standard intensive system with a feed made with French ingredients (maize, wheat, and rapeseed) and with soybean from Brazil. For the Brazilian case (BR), we assumed standard intensive systems, that chickens were fed mainly with maize and soybeans produced in the region in which the chickens were raised. As we have two scenarios that represent the Brazilian situation, we propose a scenario consisting of 75% of South chicken (SO - considered representative for the three southern states) and 25% of Centre West chicken (CW), adding to this scenario the transport distances. The LCA for the systems studied begins with the production of inputs and goods used to produce crops, passing through the phases of crop production, grain drying and processing, feed manufacturing, production of chicks, chicken rearing, slaughter, cooling and packaging of whole chicken, including all transport phases, up to the slaughterhouse gate, adding the transport to France, for the Brazilian scenario. The production and maintenance of chicken houses and of slaughterhouse buildings and machines were not included. Functional unit was 1 ton of chicken cooled and packed delivered in France. The method used for life cycle impact assessment was the CML 2 baseline 2000 with modifications. From an environmental point of view, importing chicken from Brazil rather than producing it in France with Brazilian soybeans, was better with respect to climate change and land occupation, which are both global impacts. With respect to acidification, terrestrial ecotoxicity and energy demand chicken imported from Brazil had larger impacts than the chicken produced in France.

Keywords: life cycle assessment, chicken production, Brazil, France

1. Introduction

In recent decades the poultry industry has developed and modernized, both in Brazil and in France. Increased productivity due to technological improvements, new models of integration and changes in the market favouring the increase in consumption of chicken meat, are factors that contributed to the growth of the sector in Brazil. According to the Brazilian Association of Chicken Producers and Exporters (ABEF, 2010), the total chicken production in Brazil increased from 2 million tonnes in 1989 to 12.3 million in 2010. In 2006, 2.7 million tons were exported (ABEF, 2010). In 2010, this number rose to 3.8 million tons. The Middle East, the European Union (EU) and Asia are the main destinations for Brazilian chicken. This significant increase in the export of chickens has caused international repercussions.

Meanwhile, in Europe, according to Jez et al., (2011), the rapid growth of consumption of cuts and especially processed products, for which the origin of the raw material is not promoted, has favoured imports from new producing regions which are highly competitive on the global market, like Brazil among others. In response some countries which were not meeting domestic demand, such as Poland and Germany, increased production, whereas the production of leaders such as France and the United Kingdom decreased.

According to Magdelaine (2008), the poultry French industry reached its maximum production at the end of the 90s, and then started to decline. Since then, the current picture of French poultry was set, characterised by a structural crisis that has resulted in a reduction in the volume of chicken produced by around 25% (equivalent to 550 tons of carcass). This decline in French production is primarily due to a loss of competitiveness in the light of the sharp reduction of exports extra and intra European Union, and an increase in imports. At the same time, domestic consumption, after reaching a maximum in 2001, stabilised and the market was heavily segmented allowing an increase of imported meat. Over the last ten years, poultry production in France, which remains Europe's leading poultry-meat producer and exporter to non-EU countries, has decreased by 20% (Jez et al., 2011).

The different characteristics of Brazil and France supply chains of poultry production lead to the establishment of different levels of environmental impacts. From the perspective of LCA, most of the environmental impacts of livestock production comes from the stage of feed production (raw materials), as already demonstrated by several authors (Carlsson-Kanyama, 1998; Cederberg and Mattsson, 2000; Basset-Mens and van der Werf, 2005; Pelletier, 2008; Thomassen et al., 2008; Williams et al., 2009), which draws attention to the origin of the raw material. Part of the soybean used in feed for chickens in France comes from Brazil. So the impacts of soy production (including deforestation) should be considered as part of the impact of the chickens produced in France. According to Patentreger and Billon (2008), 74% of imports of soybeans in France are from Brazil. French soybean production covers only 3% of national consumption. So, in this paper, we try to answer the following question: Do imported chickens from Brazil, fed with locally produced grains cause less (or more) impact than chickens produced in France, using a feed part of which comes from Brazil?

2. Methods

We ran a comparison assuming that the chicken produced in France (FR) was an intensive system with a feed made with French ingredients (maize, wheat, and rapeseed) and with soybean from Brazil. To represent this system of standard industrial chicken in France, we chose the region of Bretagne, which concentrates among the largest quantities of animal production in Europe.

For the Brazilian case (BR), chickens were fed mainly with maize and soybeans produced in the region in which the chickens were raised. As Brazil is a huge country, we consider that there are two distinct systems. a) In the Center-West (CW) there is a typical large scale chicken production system. To simplify, we choose a specific site in Rio Verde, located in the Southwest of Goiás state. b) In the Southern Brazil (SO), we choose the West of Santa Catarina state, a traditional region of industrial poultry production. The bureau of foreign trade of Brazil (SECEX, 2011) reported that 75% of exports of chicken come from the three southern states of the country. As we have these two scenarios that represent the Brazilian situation, we propose a scenario consisting of 75% of SO chicken (considered representative for the three southern states) and 25% of CW chicken, adding to this scenario the transport distances. Most stages of the life cycle were similar in both cases, with the greatest differences being the transportation distances involved. The distances considered were on average 1370 km from the Centre-West of Brazil to the port of Itajaí (Santa Catarina state), and on average 500 km from the South of Brazil to the same port, in a refrigerated truck. Then, we considered more 9700 km of transoceanic ship to the port of Bordeaux, France, and thereafter, another 500 km of railway, to Bretagne. Table 1 shows the technical indicators adopted for each studied system of poultry production.

Table 1. Technical indicators of poultry production systems in the West of France (FR - standard) and in Brazil (BR).

Indicator	FR	BR	
		SO	CW
Rearing time (days)	40	42	42
Final weight (kg)	1.92	2.48	2.40
Density (animals/m²)	22.0	11.7	15.0
Mortality (%)	4.1	4.4	4.2
Feed conversion (kg/kg)	1.87	1.86	1.89
No. of batches per year	6.0	6.4	6
Carcass yield (%)	70	74.6	74.6
From production site to Itajaí port – by truck (km)	-	500	1370
From Brazil (Itajaí) to France (Bordeaux) – by ship (km)	-	9700	9700
France internal transport – by train (km)	500	500	500

Sources: FR system – Peltier & Kollen (2005); CW system - Carfantan (2007); SO system - Martins et al., (2007).

In our approach, we consider various scenarios for the production of maize and soybeans for animal feed in Brazil. For the CW scenario, we consider that a small part of the soybean area (and therefore also of the maize area, since a field produces two crops within a year, maize after soybeans) was deforested, i.e. the year preceding the soybean or maize crop it was tropical rainforest or Cerrado. The impacts associated with this deforestation are included in the impacts of maize and soybeans from CW, where the CO_2 is the main issue. Further information on how we estimated the impacts of deforestation and also on scenarios of feed ingredients production can be found in Prudêncio da Silva, 2011.

The LCA for the systems studied begins with the production of inputs and goods used to produce crops, passing through the phases of crop production, grain drying and processing, feed manufacturing, production of chicks, chicken rearing, slaughter, cooling and packaging of whole chicken, including all transport phases, up to the slaughterhouse gate, adding the transport to France, for the Brazilian scenario. The production and maintenance of chicken houses and of slaughterhouse buildings and machines were not included. Functional unit was 1 ton of chicken cooled and packed delivered in France. The method used for life cycle impact assessment was the CML 2 baseline 2000 with modifications. We present results for the following impact categories: acidification, eutrophication, climate change, terrestrial ecotoxicity, land occupation and total cumulative energy demand.

3. Results

The results showed that the stage of feed production influenced the potential impacts the most. Second stage was the chicken production, and the stage that contributes least to the environmental impacts was slaughter (industrialisation). Table 2 summarises the results of the comparison.

Table 2. Contributions of the main life cycle stages for six impacts for 1 ton of chicken cooled and packaged produced in France (FR) and 1 ton of chicken cooled and packaged produced in Brazil (BR) and delivered in France.

Origin of chicken	Life cycle stage	Acidification kg SO_2eq	Eutrophi-cation kg PO_4eq	Climate change t CO_2eq	Terrestrial ecotoxicity kg 1,4DB eq	Land oc-cupation m^2a * 1000	Cumula-tive energy demand GJ
France (FR)	Slaughter	0.3	1.6	0.07	0.3	0.07	3.2
	Chicken production	27.8	6.6	0.80	1.3	0.23	6.0
	Feed production	12.4	12.8	2.30	7.0	3.52	20.8
	Total	**40.5**	**21.0**	**3.17**	**8.6**	**3.82**	**30.0**
Brazil (BR) (75% SO + 25% CW)	Slaughter	0.5	1.5	0.05	0.6	0.31	6.5
	Chicken production	20.1	4.7	0.59	1.7	0.11	7.3
	Feed production	24.3	14.1	1.51	7.0	3.14	17.5
	Transport Brazil-France[a]	3.0	0.4	0.25	0.6	0.00	4.5
	Total	**47.9**	**20.7**	**2.40**	**9.9**	**3.56**	**35.8**
Difference of total Brazil relative to FR – absolute and (%)		**7.4 (18)**	**-0.3 (-1)**	**-0.77 (-24)**	**1.3 (15)**	**0.26 (-7)**	**5.8 (19)**
Transport Brazil-France relative to FR (%)		**7**	**2**	**8**	**7**	**0**	**15**

[a] Transport by refrigerated truck, ship and train, from Brazil slaughter gate to France. Other transport stages, like feed transport, chicken transport, inputs transport, etc. are included in earlier stages.

In Table 2, the penultimate line, we highlight how the Brazilian chicken delivered in Europe cause more (or less) impact than the chicken produced in France, according to each impact category. In the last line, we highlight just the stage international transport, i.e., how the transport of chicken from Brazil up to France added on each impact category, related to chicken produced in France. This international transport of chicken stage adds about 7-8% in potential of acidification, climate change and terrestrial ecotoxicity, and 15% of cumulative energy demand.

4. Discussion

According to our scenarios, for climate change and land occupation it is better to produce chicken in Brazil and export it to France than to produce the same type of chicken in France. The international transport stage contributed only 8% to GHG emissions, and therefore, when imported in France, the Brazilian chicken still had 24% less emissions than the French chicken (Table 2). Fig. 1 shows the mains contributions for climate change.

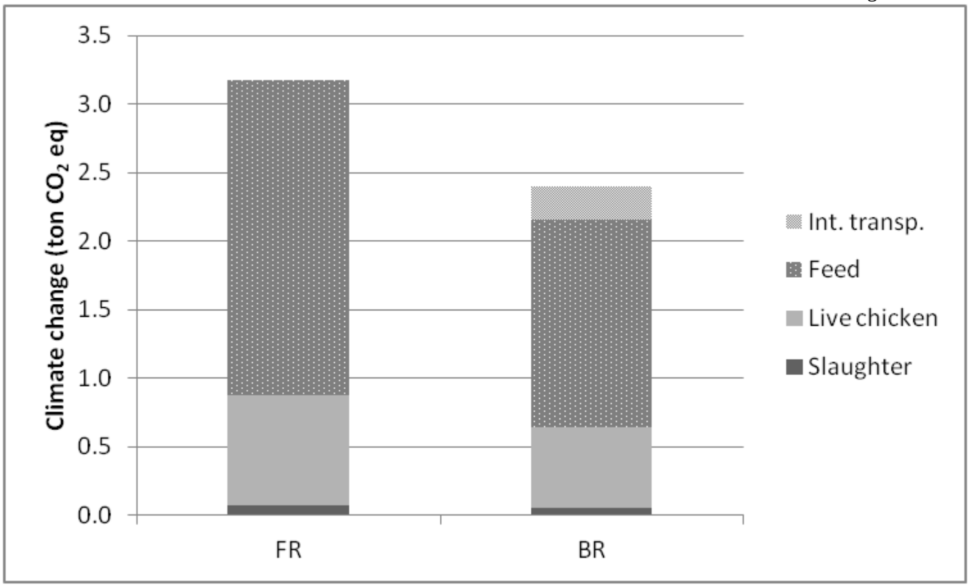


Figure 1. Contributions of the main life cycle stages for Climate Change for 1 ton of chicken cooled and packaged produced in France (FR) and in Brazil (BR) delivered in France.

For the French chicken, about 33% of greenhouse gas emissions resulted from the use of soybean meal from Brazil, as well as 24% of energy demand. It is very likely that these values would be lower if other locally produced protein-rich grains were used, in substitution of Brazilian soybeans, improving thus the environmental performance of the French chicken.

An interesting effect occurred for energy demand. On average, the Brazilian chicken consumed almost the same energy per ton of chicken at the slaughterhouse gate regarding French chicken, but due to energy demand for transportation to France, on delivery in France it required 15% more energy than the French chicken (Figure 2).

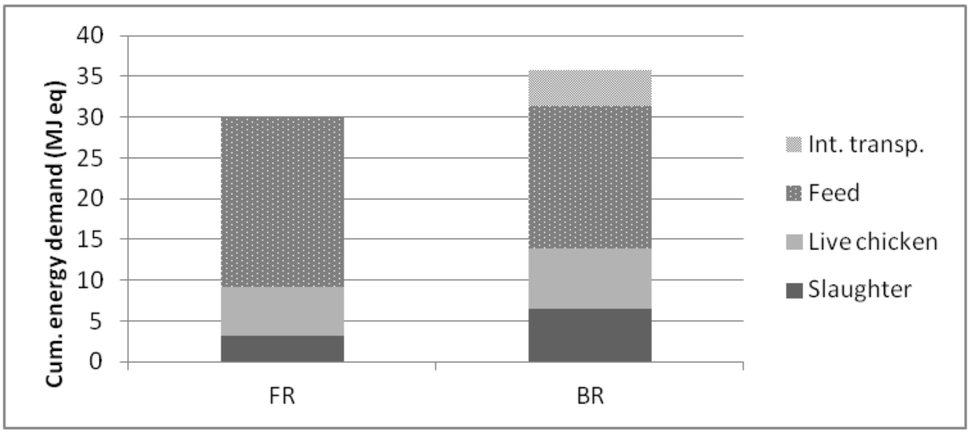


Figure 2. Contributions of the main life cycle stages for Cumulative Energy Demand for 1 ton of chicken cooled and packaged produced in France (FR) and in Brazil (BR) delivered in France.

Acidification was already higher for chicken production scenarios in Brazil, and transportation increased acidification by 7%, reaching 18% more acidifying emissions than the French chicken on delivery in France (Figure 3). A similar phenomenon occurred for terrestrial ecotoxicity.

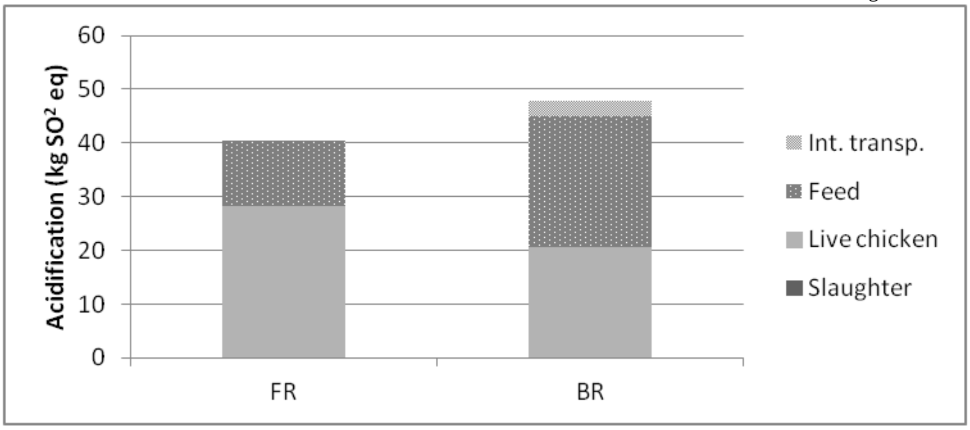


Figure 3. Contributions of the main life cycle stages for Acidification for 1 ton of chicken cooled and packaged produced in France (FR) and in Brazil (BR) delivered in France.

In the FR system, the most emissions (62%) of substances that contribute to the total acidification potential, come from ammonia emitted in the chicken house. In the BR system, the ammonia emission in chicken house is only 39% of the total acidification potential. But on the other hand, in Brazil the stage of feed production contributes over 44% of emissions of acidifying substances, mainly because of the ammonia emitted due to the use of urea as a nitrogen fertiliser for maize production.

From an environmental point of view, importing chicken from Brazil rather than producing it in France with Brazilian soybeans, was better with respect to climate change and land occupation, which are both global impacts. With respect to acidification, terrestrial ecotoxicity and energy demand chicken imported from Brazil had larger impacts than the chicken produced in France. It is therefore not simple to answer this question. If one considers that climate change is the most important environmental issue, then the import of Brazilian chicken would seem preferable and stopping deforestation in Brazil would strongly reduce the climate change impact of both Brazilian and French chicken.

5. Conclusion

The grain production stage is the largest contributor to the overall environmental impacts along the chicken meat supply production chain. In general, recommendations that may improve the environmental performance of feed crop production will also reduce the impacts of chicken production.

Importing chicken from Brazil rather than producing it in France with Brazilian soybeans, was better with respect to climate change and land occupation. With respect to acidification, terrestrial ecotoxicity and energy demand chicken imported from Brazil had larger impacts than the chicken produced in France. If one considers that climate change is the most important environmental issue, then the import of Brazilian chicken would seem preferable.

6. References

ABEF – Associação Brasileira de Exportatores de Frango, 2010. Estatísticas [online] Available from: <http://www.abef.com.br>. 15 jul 2011.

Basset-Mens C, van der Werf HMG., 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. Agriculture, Ecosystems & Environment 105: 127-144.

Carfantan, J.-Y., 2007. Le poulet-voyageur. Dynamique et prospective de la filiere poulet bresilienne. Uberlândia, Brasil: Céleres. 147 p.

Carlsson-Kanyama A., 1998. Energy consumption and emissions of greenhouse gases in the life-cycle of potatoes, pork meat, rice and yellow peas. Department of Systems Ecology, University of Stockholm.

Cederberg C, Mattsson B., 2000. Life cycle assessment of milk production - a comparison of conventional and organic farming. Journal of Cleaner Production 8: 49-60.

Jez, C.; Beaumont, C.; Magdelaine, P., 2011. Poultry production in 2025: learning from future scenarios. World's Poultry Science Journal, v. 67, n. 01, p. 105-114.

Magdelaine, P., 2008. La situation des filières avicoles françaises. Rapport réalisé dans le cadre du projet AVITER – ITAVI, Paris, 75p.

Martins, F. M.; Talamini, D. J. D.; Souza, M. V. N. de., 2007. Coeficientes técnicos e custos agregados na cadeia produtiva do frango no Oeste Catarinense. Concórdia, SC, Brazil: Embrapa Suínos e Aves. 50 p.

Patentreger B, Billon A., 2008. L'impact de l'agriculture et de l'alimentation industrielles sur la forêt dans le monde - rôle de la France. WWF - France: Paris, France.

Pelletier N., 2008. Environmental performance in the US broiler poultry sector: Life cycle energy use and greenhouse gas, ozone depleting, acidifying and eutrophying emissions. Agricultural Systems 98: 67-73. DOI: doi: 10.1016/j.agsy.2008.03.007.

Peltier, J.-L.; Kollen, A., 2005. La filière avicole - Volaille standard (poulet, dindes, pintades). Catalogue des Filières. France: Chambre d'Agriculture AIN. Available in: <http://rhone-alpes.synagri.com/synagri/pj.nsf/TECHPJPARCLEF/04596/$File/CATALOG%20FILIERES%20fil%20avivolstandar.pdf?OpenElement>. 19 dec 2005.

Prudêncio da Silva, V., 2011. Effects of intensity and scale of production on environmental impacts of poultry meat production chains. Rennes, France / Florianópolis, Brazil: Agrocampus Ouest - Université Européenne de Bretagne / Universidade Federal de Santa Catarina. Thesis. 192p.

SECEX – Secretaria Executiva do Ministério do Desenvolvimento, Indústria e Comércio Exterior - Bureau of foreign trade of Brazil, 2011. Available in: <http://www.mda.gov.br/portal/institucional/SECEX>. 10 jul 2011.

Thomassen MA, van Calker KJ, Smits MCJ, Iepema GL, de Boer IJM., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. Agricultural Systems 96: 95-107.

Williams AG, Audsley E, Sandars DL., 2009. A lifecycle approach to reducing the environmental impacts of poultry production. Presented at the 17th European Symposium on Poultry Nutrition. Endinburg, UK. 23 August, 2009.

Effect of ethics on integral ecological impact of organic eggs

S.E.M. Dekker[1], I.J.M. de Boer[2], A.J.A. Aarnink[3], P.W.G., Groot Koerkamp[1]

[1] Farm Technology Group, Wageningen University, P.O. box 17, 6700 AA, Wageningen, the Netherlands
[2] Animal Production Systems Group, Wageningen University, P.O. box 338, 6700 AH, Wageningen, the Netherlands
[3] Wageningen UR Livestock Research, P.O. box 65, 8200 AB, Lelystad, the Netherlands
[*] Corresponding author. E-mail: *sanne.dekker@wur.nl*

ABSTRACT

Organic agriculture has a specific perspective on sustainability, captured in their goal, definition and four principles of organic agriculture and resulting certification (IFOAM, 2012). We refer to this perspective as the organic-ethical framework. For organic egg production, this framework implies three main requirements: (1) loose hen housing, (2) outdoor access, and (3) limited use of external resources, i.e. artificial fertilisers, pesticides, herbicides, genetically modified organisms, and medication. Our research objective was to assess the effect of these three requirements on the integral ecological impact of Dutch organic egg production. We approached this objective in three steps. In step 1 we compared life cycle assessment (LCA) results of egg production with and without an organic ethical framework. In step 2 we identified main ecological issues of current Dutch organic egg production by means of LCA. In step 3 we explored options to reduce integral ecological impact of Dutch organic egg production within the boundaries of the organic ethical framework, i.e. replace single-tiered with multi-tiered loose hen housing and replace imported with regional diet ingredients.

Comparison of battery cage with barn egg production shows that the requirement of loose hen housing results in a 12% to 176% higher impact for the ecological issues studied, except for phosphorus (P) deficit which was equal (Table 1). This higher impact is mainly explained by a higher conversion of feed to eggs of loose housed hens (2.33 kg feed kg^{-1} egg) compared with hens in battery cages (1.99 kg feed kg^{-1} egg) (Dekker et al., 2011b). A second reason for the higher acidification potential of loose hen housing was that reduction of ammonia emission by drying and removal of manure in loose laying hen houses is problematic. Comparison of barn with free range egg production shows that the requirement of access to an outdoor run results in a relative small, i.e. 0% to 10%, increase of the ecological impact (Table 1). Comparison of free range and organic egg production shows that the requirement of limited use of external resources results in a 10% lower global warming potential, a 15% lower energy use, a 93% lower fossil P use, a 108% lower nitrogen (N) surplus and a 114% lower P surplus, but a 82% higher land occupation, a 68% higher acidification potential, a 1767% higher N surplus and a 900% higher P deficit (Table 1). We found that differences in the ecological impact of egg production systems with and without limited use of external resources resulted mainly from differences in type and amount of fertilisation and conversion of feed to eggs. Multi-tiered housing to dry and remove manure (mitigation housing in Table 1; Dekker et al., 2011a) and production of diet ingredients and eggs in the same region to assure availability of manure and increase yields (mitigation diet in Table 1; Dekker et al., 2012b), have potential to reduce the ecological impact of organic egg production within the ethical boundaries of organic egg production. If these mitigation options are applied we predict a lower energy use, fossil P use, N surplus and P surplus, but a higher global warming potential, land occupation, acidification potential, N deficit and P deficit for organic compared with battery cage egg production.

We conclude that increases of the ecological impact caused by the organic ethical framework mainly result from inefficient N and P management and inefficient conversion of feed to eggs. Issues in the current egg production chain regarding manure management are: (1) a lack of regionally available manure (Dekker et al., 2011b), (2) unbalanced application and N-P ratios of manure fertiliser, (3) high N-emissions from faeces in loose housing systems, and (4) loss of N and P from manure in the outdoor run (Dekker et al., 2011b; Dekker et al., 2012a). The higher conversion of feed to eggs may be caused by: (1) a higher body mass of loose housed hens, (2) increased freedom of movement, (3) prohibited use of external resources, caused by a worse amino acid profile of the diet, (4) limited use of medication, and (5) a smaller number of hens per m^2 in the hen house, which generally results in a lower indoor temperature (Van Knegsel and van Krimpen, 2008; Dekker et al., 2012b; Dekker et al., 2011a). Research is required to determine whether inefficient N and P management and inefficient conversion of feed to eggs in organic egg production must be accepted as an implication of the organic ethical framework or can be further reduced within the boundaries of the organic ethical framework.

Table 1. Production characteristics and ecological impacts of existing egg production systems with and without three ethical requirements (i.e. loose hen housing, outdoor access, and limited use of external resources), mitigation housing (i.e. replace single-tiered with multi-tiered housing), and diet (i.e. replace imported with regional diet ingredients in percent relative to battery cage egg production).

	Battery cage eggs [a]	**Barn eggs** [a]	**Free range eggs** [a]	**Organic eggs** [a]	**Mitigation housing** [a]	**Mitigation diet** [b]
Loose hen housing	No	Yes	Yes	Yes	Yes	Yes
Outdoor access	No	No	Yes	Yes	Yes	Yes
Limited use of external resources	No	No	No	Yes	Yes	Yes
Single-tiered housing	No	Yes	Yes	Yes	No	No
Multi-tiered housing	No	No	No	No	Yes	Yes
Regional diet ingredients	No	No	No	No	No	Yes
Global warming potential	100%	120%	123%	113%	114%	104%
Energy use	100%	112%	115%	100%	98%	77%
Land occupation	100%	115%	125%	207%	207%	141%
Fossil P use	100%	115%	118%	25%	25%	-
Acidification potential	100%	276%	283%	351%	209%	207%
N deficit	100%	133%	133%	1900%	1900%	1615%
P deficit	100%	100%	100%	1000%	1000%	410%
N surplus	100%	115%	122%	14%	14%	14%
P surplus	100%	115%	124%	10%	10%	8%

[a] Dekker et al., 2011b; [b] Dekker et al., 2012b

References

Dekker, S.E.M., de Boer, I.J.M., Aarnink, A.J.A., Groot Koerkamp, P.W.G., 2011a. Emissions of ammonia, nitrous oxide, and methane from aviaries with organic laying hen husbandry. Biosystems Engineering, 110: 123-133.

Dekker, S.E.M., de Boer, I.J.M., Vermeij, I., Aarnink, A.J.A., Groot Koerkamp, P.W.G., 2011b. Ecological and economic evaluation of Dutch egg production systems. Livestock Science, 139 (1-2): 109-121.

Dekker, S.E.M., Aarnink A.J.A., de Boer, I.J.M., Groot Koerkamp, P.W.G., 2012a. Total loss and distribution of nitrogen and phosphorus in the outdoor run of laying hens. British Poultry Science, (*in press*).

Dekker, S.E.M., de Boer, I.J.M., van Krimpen, M.M., Aarnink, A.J.A., Groot Koerkamp, P.W.G., 2012b. Effect of composition of concentrates on ecological impact of organic egg production. Livestock Science, (*in press*).

IFOAM, 2012. http://www.ifoam.org.IFOAM: International Federation of Organic Agricultural Movements.

Van Knegsel, A.T.M., van Krimpen, M.M., 2008. Energy and protein requirements of organic housed poultry (Energie- en eiwitbehoefte van biologisch gehouden pluimvee). Report 122. Wageningen UR Animal Sciences Group, Lelystad, Netherlands, 22 pp.

Environmental impacts of different pork and chicken meat production systems in Switzerland and selected import sources

Florian Grandl*, Martina Alig, Johanna Mieleitner, Thomas Nemecek, Gérard Gaillard

Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland
* Corresponding author. E-mail: *florian.grandl@art.admin.ch*

ABSTRACT

Three production systems in Switzerland were compared to selected imports for pork and chicken meat. The analysis was carried out at the level of the farm gate and at the retail store gate. Agricultural production was responsible for at least 75% of the environmental impacts in each analysed category, so it was more important how the animals were produced than where production occurs. In all production systems, feedstuffs were the dominant inputs. For pork, the differences between conventional and label production were minor, as the level of productivity remained more or less the same. In chicken production, the differences were considerable. Fattening performance in label production was lower as slower growing strains are used. A reduction of environmental impacts by improved feeding seems to be most promising. More intensive production systems tend to have less environmental impacts. However, systems with improved animal welfare can produce with similar environmental impacts as long as the animal performance reaches the same level of productivity.

Keywords: LCA, environmental impacts, meat production systems, pork, chicken

1. Introduction

Animal production faces many challenges today and consumer awareness of animal welfare and of the environmental impacts of animal production rises. However, there is a risk of a conflict of goals when only improving one of these aspects. So consumers, producers and policy makers need comprehensive information on the impacts of differently produced agricultural commodities in order to make a decision on what products to buy and to produce. In this study, we compared three different production systems (conventional, label production with improved animal welfare and organic) in Switzerland and selected import products both for pork and chicken meat. The aim was to analyse the environmental impacts of the meat production systems and to identify their major drivers as well as possible strategies meant for improving their environmental performance.

2. Data and Methods

Data and design of model farms for Swiss pig production systems were derived from the project Life Cycle Assessment – Farm Accountancy Data Network (LCA-FADN), where data from more than 100 farms have been collected (Hersener et al., 2011). For Swiss chicken production, production data were collected from a large Swiss chicken meat processor. The agricultural production of imported pork and chicken meat was modelled according to literature, statistical and expert data from the particular countries (Alig et al., 2012). Agricultural production data for pig and chicken is shown in Table 1 and Table 2. For Switzerland, three different production systems were investigated: Swiss conventional production (CH conv.), production with improved animal welfare standards (CH label) and organic production (CH organic). Imported meat was assumed to be produced conventionally. In case of Brazilian chicken meat, two production systems are modelled in order to represent the regionally different situation in the main production areas in the south (BR S) and in the centre-west of Brazil (BR CW).

Post-agricultural processes included transport of animals, slaughtering and meat processing in Switzerland and/or the country of origin and packaging and transport of the meat pieces ready for sale at retail store. Data for slaughter and meat processing and storage and transport were obtained from a Swiss meat processor and a Swiss retail company as well as from literature.

LCAs were calculated with the method SALCA (Swiss Agricultural Life Cycle Assessment, Nemecek et al., 2010). The analysis was carried out at two levels: at farm gate with the functional unit of 1 kg live weight (LW) and at the retail store gate, with the functional unit of 1 kg packed and cooled meat.

The environmental impacts considered in this analysis are summarised in Table 3. The results are grouped into three main categories (resource use management, nutrient management and toxicity) and cover relevant environmental impacts of agricultural production and food products. For more details, see Nemecek et al., (2010).

Table 1. Agricultural production data for three pig production systems in Switzerland (CH conv., label, organic), Germany (DE) and Denmark (DK).

	CH conv.	label	organic	DE	DK
Rate of turnover [number/year]	3.0	3.1	2.9	2.7	3.9
Mortality [%]	1.29	1.12	1.12	3.0	4.3
Live weight at start [kg LG]	24	24	24	25	30
Live weight at slaughter [kg LG]	102	100	103	115	102
Length of fattening period [d]	106	103	110	128	89
Daily gains [g/d]	740	740	720	700	810
Feed conversion ratio 1: [kg/kg]	2.87	3.04	2.85	3.00	2.76
Live weight gain per place unit [kg LW/year]	234	236	229	243	269
Floor type	partly slatted	partly slatted	partly slatted	fully slatted	50:50 partly / fully slatted
Outdoor run	no	yes	yes	no	no

Table 2. Agricultural production data for three chicken production systems in Switzerland (CH conv., label, organic) and France (FR) and Brazil (BR CW and BR S).

	CH conv.	label	organic	FR	BR CW	S
Length of fattening period [d]	35	56	21 + 42[a]	40	42	42
Rate of turnover [number/year]	8.69	5.79	7.45	6.0	6.0	6.4
Break between production cycles [d]	7	7	7	20.8	18.8	15.0
Live weight at slaughter [kg LG]	1.92	1.85	1.75	1.92	2.40	2.48
Daily gains [g/d]	54	32	27	47	56	58
Feed conversion ratio 1: [kg/kg]	1.65	2.17	2.42	1.87	1.89	1.86
Mortality [%]	3.05	2.50	3.50	4.1	4.2	4.4
Outdoor run [m^2 per animal]	-	2.00	2.05	-	-	-

[a] A 21-day rearing phase takes places in a second building parallel to fattening.

Table 3. Considered environmental impacts (Nemecek et al., 2010).

Category	Environmental impact
Resource use management	• Demand for non-renewable energy resources (oil, coal and lignite, natural gas and uranium), using the upper heating or gross calorific value for fossil fuels according to Hischier et al., (2010). • Global warming potential over 100 years (according to IPCC, 2007). • Ozone formation potential (so-called “summer smog” according to the EDIP2003 method, Hauschild und Potting, 2003). • Use of phosphorus and potassium resources is based on life cycle inventory (LCI) data • Land competition is based on LCI data according to the CML01 method (Guinée et al., 2001) • Deforestation is calculated based on LCI data as the difference of area transformed from forest and area transformed to forest • Total water use (blue) is calculated on LCI data
Nutrient management	• Eutrophication potential (impact of the losses of N and P to aquatic and terrestrial ecosystems, according to the EDIP2003 method, Hauschild und Potting, 2003). • Acidification potential (impact of acidifying substances released into ecosystems, according to the EDIP2003 method, Hauschild und Potting, 2003).
Toxicity	• Terrestrial and aquatic ecotoxicity potentials (according to the CML01 method, Guinée et al., 2001). • Human toxicity potential (impact of toxic pollutants on human health, according to the CML01 method, Guinée et al., 2001).

3. Results

When considering the whole supply chain up to the retail store, agricultural production was found to be responsible for at least 75% of the environmental impacts in each analysed environmental impact category (Figure 1 and Figure 2). Post-agricultural processes were of noticeable influence only for the consumption of non renewable energy and to some extent for global warming potential, ozone formation and human toxicity. Most of the environmental impacts from post-agricultural processes originated from slaughtering and meat processing and from transports in case of imported meat (see Mieleitner et al., (2012) for more details).

The major potential of environmental improvement lies in animal production. Therefore, the rest of the result section focuses only the agricultural processes and refers to the functional unit of 1 kg LW.

Figure 1 and Figure 2 show the differences in environmental impacts between the three Swiss production systems of pork and chicken meat, respectively (1 kg meat at the retail store). Both show substantial differences between organic and conventional production in arable land competition (e.g. in pig production: 3.8 m^2a per kg LW vs. 6.9 m^2a per kg LW under conventional and organic conditions), nutrient management and toxicity. These differences are explained by the different cropping practise in organic and conventional agriculture. In organic agriculture, the abandonment of pesticides leads to lower toxic impacts and the application of organic instead of mineral fertilisers increases ammonia emissions (e.g. in pig production: terr. eutrophication is 1.9 m^2 per kg LW in conventional production vs. 4.19 m^2 per kg LW under organic farming) but reduces the use of phosphorus and potassium resources compared to conventional farming. Lower yields per area in organic farming require a larger area for feed production.

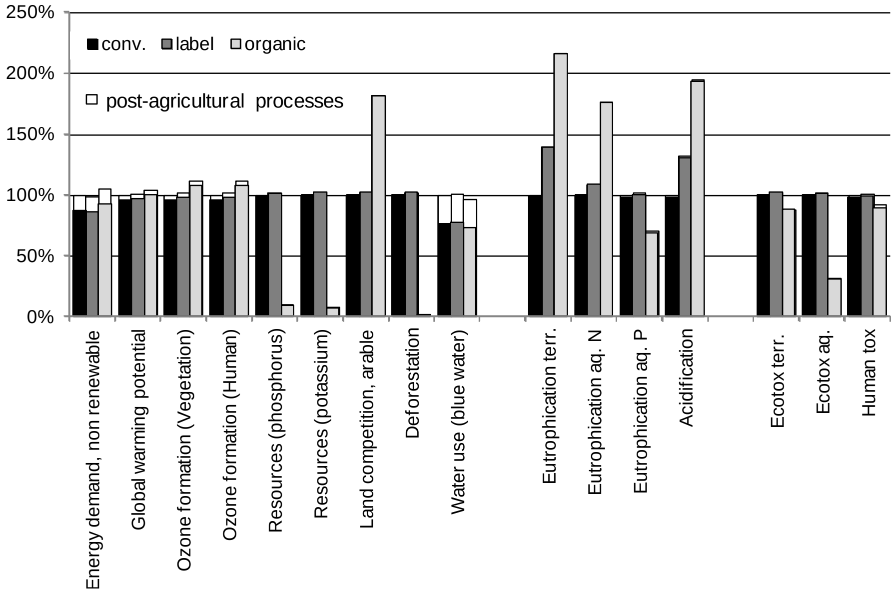


Figure 1. Relative environmental impacts for 1 kg of pork at farm gate and at the retail store for three production systems in Switzerland. 100% = conventional production in Switzerland.

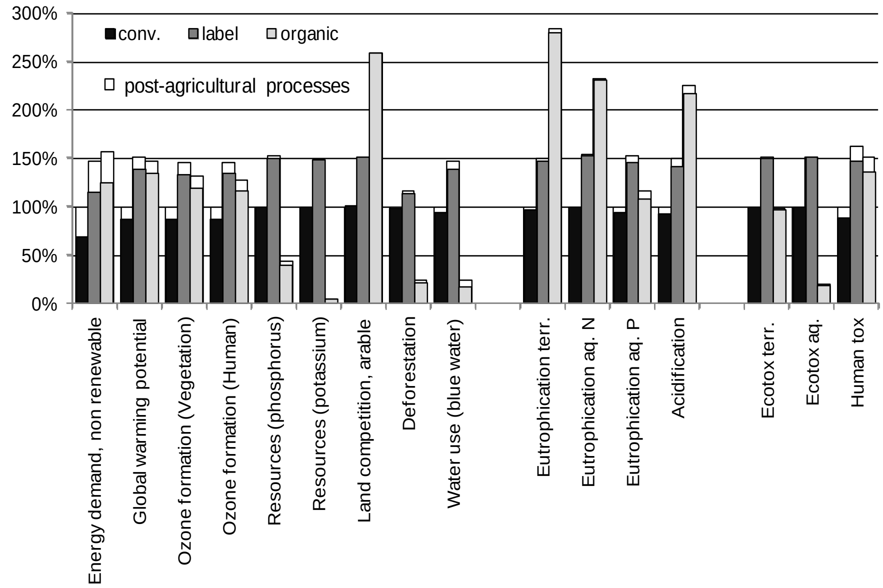


Figure 2. Relative environmental impacts for 1 kg of chicken meat at farm gate and at the retail store for three production systems in Switzerland. 100% = conventional production in Switzerland.

Figure 2 also clearly indicates the difference in efficiency of the chicken production systems. Animals in label and organic production have a lower fattening performance (see Table 2), mainly due to the use of slower growing strains which are better adapted to label production conditions. Comparing conventional and label production at the farm level, label production has around 30% higher environmental impacts than conventional production (e.g. energy demand is 17.3 for conventional and 26.93 MJ-eq. per kg LW for organic chicken meat, respectively), which is more or less the same as difference in fattening performance. In pig production, the differences between conventional production and production under improved animal welfare conditions are minor, as the level of productivity (daily gains, feed conversion, etc.) remains more or less the same and hardly any further inputs are needed. There is only an increase in eutrophication and acidification potential which results from higher ammonia emissions due to the straw bedding and the outdoor run.

Both conventional pig and chicken production are relatively standardised production systems in developed countries. Animal genetics and production techniques and methods are similar and therefore differences in animal performance are rather small (see Table 1 and Table 2). Accordingly, Figure 3 shows similar results in environmental performance for pig production in different countries. The same applies for chicken production, where production is even more standardised all over the world. A decisive factor the impact category deforestation is the production of soybean meal. So different shares of this feedstuff in diets and the use of soybeans from non-deforested areas influences this category greatly and has also an effect on global warming potential due to CO_2-emissions from deforested areas.

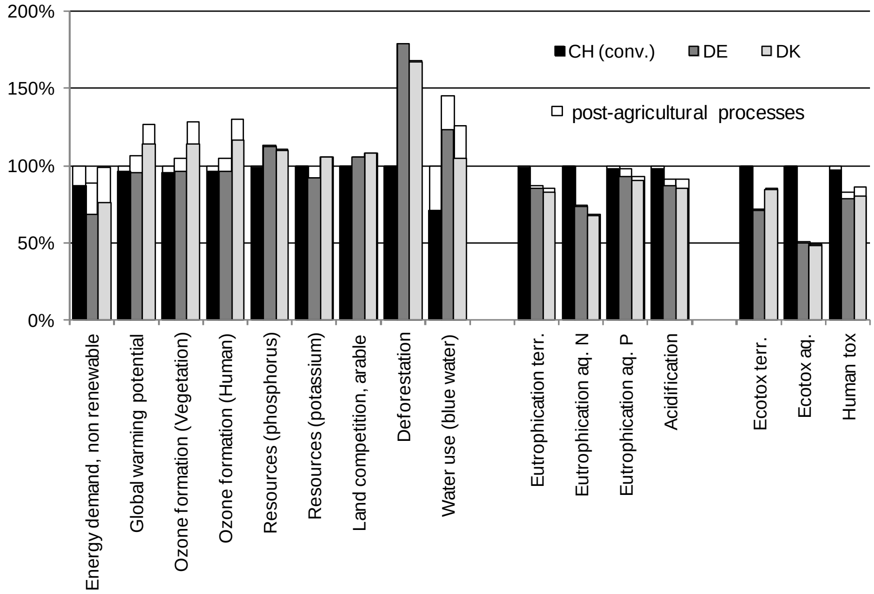


Figure 3. Relative environmental impacts for 1 kg of pork at farm gate and at the retail store for conventional production in Switzerland (CH conv.), Germany (DE) and Denmark (DK). 100% = conventional production in Switzerland.

Both in pork and in chicken production, feedstuffs are by far the most dominant inputs. Feedstuffs account for about 40% of the non-renewable energy consumption and global warming potential and for around two thirds of the human toxicity and ecotoxicity impacts of pork at the farm gate (Figure 4). In chicken production, where environmental impacts per kg LW are generally lower compared to pork, feedstuffs are even more important, accounting for more than 50% of the non-renewable energy consumption and more than 70% of the global warming potential of 1 kg chicken (LW) (Figure 5).

The environmental impacts of feedstuffs result mainly from crop production. So both the method of production of feeds and the efficiency of feed conversion turned out to be decisive factors for environmental impacts per kg meat.

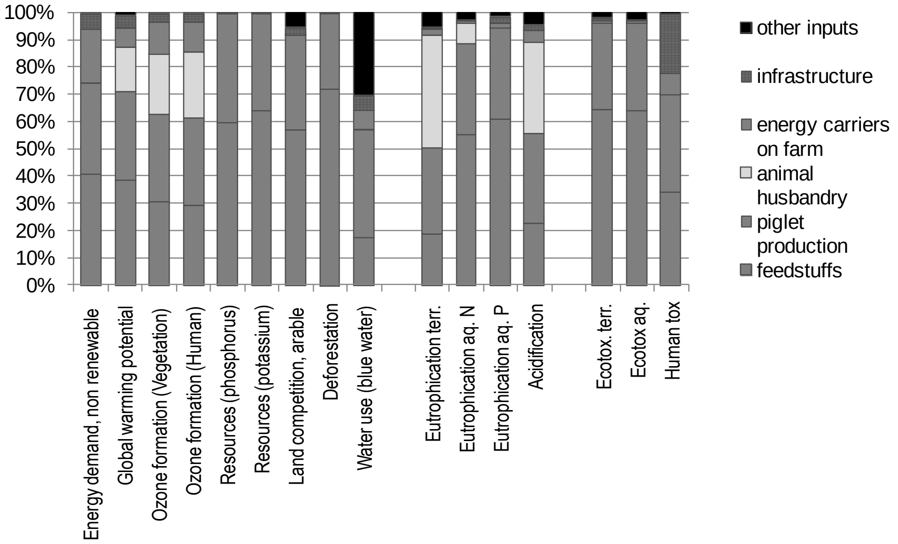


Figure 4. Relative contribution of inputs to environmental impacts of 1 kg of pork (live weight, at farm gate) for Swiss conventional production.

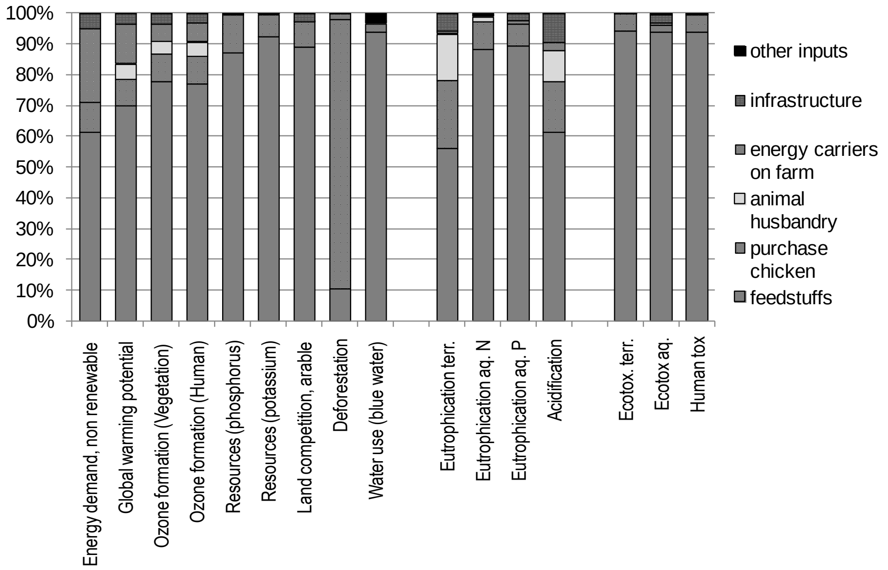


Figure 5. Relative contribution of inputs to environmental impacts of 1 kg of chicken meat (live weight, at farm gate) for Swiss conventional production.

4. Discussion

Generally, the results of this study fitted well into other analyses of meat production. Strid and Röös (2011) performed a literature review for pig production systems in Europe and found for the most environmental impact categories similar tendencies. However, most studies are restricted to only a few impact categories and the different calculation methods, system boundaries and functional units make sound comparisons often very difficult. This is especially the case for chicken meat production, where apart from live weight and slaughter weight various other functional units were used for the analyses (e.g. different steps of processing, cooled or frozen products, whole animals or parts).

The difference in fattening performance was comparatively small in pig production and higher in broiler production where again slower growing strains and moderate deficiencies in feed rations led to slower growth rates and less feed efficiency. Similar results were found by Prudêncio da Silva et al., (2010) who compared label and conventional chicken production in France and by Willams et al., (2006) for chicken production in the UK.

5. Conclusion

Both for pork and chicken meat, agricultural production dominated the environmental impacts of the whole meat production chain. Therefore, it is much more relevant *how* animal production is done than *where* it takes place. Apart from various specific measures (e.g. improved manure management), generally the decisive factors for the environmental performance of meat production systems were found to be the overall efficiency of the production system and the whole complex of feeding.

For monogastric animals, a reduction of environmental impacts by improved feedstuff production and feeding strategies seems to be most promising. The improvement (e.g. by optimising fertilisation and crop protection schemes and machinery usage) in the area of feedstuff production is rather a general challenge for crop production than for animal production. Yet, animal nutrition offers several starting points for reductions in environmental impacts from feeding. Apart from further increasing the adaptation of diets to the nutritional needs and therefore reducing excess supply, the integration of environmental aspects in diet planning has a great potential for reducing environmental impacts of diets fed to animals.

For pork and chicken meat, more intensive production systems tend to have less environmental impacts per kg of produced meat because of a more efficient use of inputs, particularly feed. This was clearly shown in this study for chicken production. However, systems with improved animal welfare can produce with similar environmental impacts as long as animal performance reaches the same level of productivity as it was the case for pig production. To improve the overall performance of animal production systems it is vital to keep all aspects of sustainability in mind in order to avoid negative consequences of single improvements of one aspect in another.

6. Acknowledgement

These research results were developed by ART with the support of COOP. The authors want to thank Vamilson Prudêncio da Silva for his permission to use data of French and Brazilian chicken production.

7. References

Alig, M., Grandl, F., Mieleitner, J., Nemecek, T. & Gaillard, G., 2012. Ökobilanz von Rind-, Schweine- und Geflügelfleisch. Agroscope Reckenholz-Tänikon Research Station, Zurich.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., de Bruijn, H., van Duin, R., Huijbregts, M.A.J., Lindeijer, E., Roorda, A.A.H. & Weidema, B.P., 2001. Life cycle assessment – An operational guide to the ISO standards. Ministry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML), Den Haag and Leiden, Netherlands.

Hauschild, M. & Potting, J., 2003. Spatial differentiation in Life Cycle impact assessment – The EDIP2003 methodology. Institute for Product Development, Technical University of Denmark.

Hersener, J.-L., Baumgartner, D.U., Dux, D. (Eds.), 2011. Zentrale Auswertung von Ökobilanzen landwirtschaftlicher Betriebe (ZA-ÖB). Agroscope Reckenholz-Tänikon Research Station, Zurich/Ettenhausen.

Hischier, R., Weidema, B., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Frischknecht, R., Hellweg, S., Humbert, S., Jungbluth, N., Köllner, T., Loerincik, Y., Margni, M. & Nemecek, T., 2010. Implementation of Life Cycle Impact Assessment Methods. Final report ecoinvent v2.2 No. 3. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

IPCC, 2007: Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC (2007): Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 996 p.

Mieleitner, J., Alig, M., Grandl, F., Nemecek, T. & Gaillard, G., 2012: Environmental impact of beef – role of slaughtering, meat processing and transport. Poster presented at the 8th Int. Conf. on LCA in the Agri-Food Sector, St. Malo, France.

Nemecek, T., Freiermuth Knuchel, R., Alig, M. & Gaillard, G., 2010. The advantages of generic LCA tools for agriculture: examples SALCAcrop and SALCAfarm. In: 7th Int. Conf. on LCA in the Agri-Food Sector, Notarnicola, B. (eds.). Bari, Italy. 433-438.

Prudêncio da Silva, V., van der Werf, H. & Soares, S., 2010. LCA of French and Brazilian broiler poultry production systems. In: Notarnicola, B., Settanni, E., Tassielli und Giungato, P. (Eds.): Proceedings lcafood 2010, Bari, 475-480.

Strid, I. & Röös, E., 2011. Environmental impact assessment of pig production systems in Europe – from land use to feed efficiency. In: KTBL (Ed.): Emissionen der Tierhaltung. KTBL-Schrift 491.

Williams, A.G., Audsley, E. & Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Bedford: Cranfield University and Defra.

Evaluation of the environmental sustainability of different European pig production systems using life cycle assessment

J.Y. Dourmad[1,2,*], J. Ryschawy[1,2], T. Trousson[1,2], M. Bonneau[1,2], J. Gonzalez[3], H.W.J. Houwers[4], M. Hviid[5], C. Zimmer[6], T.L.T. Nguyen[7], L. Morgensen[7]

[1]INRA, UMR1348 Pegase, 35590 Saint-Gilles, France,
[2]Agrocampus Ouest, F-35000 Rennes, France,
[3]IRTA, Finca Camps i Armet, 17121 Monells, Girona, Spain.,
[4]Wageningen UR Livestock Research, PO Box 65, 8200 AB Lelystad, The Netherlands,
[5]DMRI, Maglegaardsvej 2, DK-4000 Roskilde, Denmark,
[6]BESH, Haller Str. 20, 74549 Wolpertshausen, Germany,
[7]DJF, Univ. of Aarhus, 8830 Tjele, Denmark.
[*] Corresponding author. E-mail: *jean-yves.dourmad@rennes.inra.fr*

ABSTRACT

The environmental sustainability of 15 European pig production systems has been evaluated within the EU Q-PorkChains project, using life cycle assessment (LCA). One conventional and two differentiated systems were evaluated from each of five countries: Denmark, Netherlands, Spain, France and Germany. The information needed for the calculations was obtained from an enquiry conducted on 5 to 10 farms from each system. The different systems were categorized among conventional (C), adapted conventional (AC), traditional (T) and organic (O). Compared to conventional, the differentiation was rather limited for AC systems with only some changes in order to improve meat quality, animal welfare or environmental impact. The difference was much more marked for the traditional systems with the use of fat slow-growing traditional breeds and generally the outdoor raising of the fattening pigs. The environmental impacts were calculated at farm gate, including the inputs, and expressed per kg live pig and per ha land use. For the conventional systems, the impact per kg live pig on climate change, acidification, eutrophication, energy use, and land occupation were 2.25 kg CO_2-eq, 44.0 g SO_2-eq, 18,5 g PO_4-eq, 16.2 MJ and 4.13 m^2, respectively. Compared to C, the corresponding values were on average 13, 5, 0, 2 and 16% higher for AC; 54, 79, 23, 50 and 156% higher for T, and 4, -16, 29, 11 and 121% higher for O. Conversely, when expressed per ha of land use, the impacts were lower for T and O differentiated systems, by 10 to 60% on average, depending on the impact category. This was mainly due to larger land occupation per kg pig produced as well for feed production and for the outdoor raising of sows and/or fattening pigs. The use of litter bedding tended to increase climate change impact per kg pig. The use of traditional local breeds, with reduced productivity and feed efficiency, resulted in higher impacts per kg pig produced, for all categories. Differentiated T systems with extensive outdoor raising of pigs resulted in markedly reduced impact per ha land use. Eutrophication potential per ha was substantially lower for O systems. Conventional systems were generally better for global impacts, expressed per kg pig, whereas differentiated systems were better for local impacts, expressed per ha land use.

Keywords: pig production, systems, environment, Life Cycle Assessment

1. Introduction

World livestock production has major impacts on the environment, because of its emissions to the environment which affect air, water and soil quality, and the use of limited or non renewable resources (Steinfeld et al., 2006). In this context the EU pork production system is facing major challenges. There is increasing societal concern regarding the currently prevailing intensive production systems (Petit and van der Werf, 2003), mainly because of environmental and animal welfare shortcomings. Although, non conventional production systems are often believed to be more sustainable, their real benefits for the environment may be controversial (Basset-Mens and van der Werf, 2005). An inventory at farm level of pig production systems, mainly from EU countries, has recently been performed within the Q-PorkChains EU project (Bonneau et al., 2011). This inventory was used as a basis for selecting contrasting systems that were evaluated in the present study. This evaluation was performed using a toolbox developed from the literature (Edwards et al., 2008) with life cycle assessment (LCA) as the method for the evaluation of the environmental sustainability.

2. Methods

2.1. Goal definition, system description and collection of data

Fifteen EU pig production systems were chosen among the 84 systems available in the inventory of pig production systems (Bonneau et al., 2011). One conventional and two differentiated systems were evaluated from each of five countries: Denmark, Netherlands, Spain, France and Germany. The different systems were categorized according to the typology defined by Bonneau et al., (2011) among conventional (C, n=5), adapted conventional (AC, n=5), and differentiated, including traditional (T, n=3) and organic (O, n=2). The information needed for LCA calculations was obtained from an enquiry conducted on about 10 farms from each system. Data collected concerned: (i) animal performance, including sows productivity, mortality rates, pig growth and feed intake during post-weaning and fattening periods, slaughter characteristics, (ii) feed composition including metabolisable energy (ME), protein and phosphorus contents, (iii) animal housing including type of housing (indoor, outdoor, free range...), type of floor (litter bedding, complete of partially slatted floor...) and (iv) manure handling, including management in the building (liquid, solid, frequency of removal...) and during storage (type and duration of storage), manure treatment (composting, anaerobic or aerobic digestion) and type and distance of spreading. From the collected data, an "average" system was built for each production system. Performance and nutrient flows and emissions were calculated for each production stage, *i.e.* the sows and their piglets until weaning, the post-weaning piglets and the fattening pigs. In this way it was easy to aggregate the whole production systems, considering number of piglets weaned per sow per year, and mortality rates of pigs during post weaning and fattening periods.

2.2. System boundaries and functional units

This is a cradle-to-farm-gate study over the whole pig production system including the reproducing sows and their piglets until weaning, the post-weaning piglets and the fattening pigs. The definition of system and subsystem boundaries was mainly derived from Basset-Mens and van der Werf (2005) and Nguyen et al., (2010). The main sub-system is the pig unit which includes the production of piglets and their raising until slaughter weight. This unit is considered to be landless as assumed by Nguyen et al., (2010) but it interacts with land use through the import of feed and the deposition/use of manure produced by the animals. The land used in case of outdoor pig raising is also considered within the system. The studied system includes the production and delivery of feed produced off-farm, herd management, and emissions from the animals and manure storage. The environmental consequences of manure utilisation are evaluated using system expansion as described by Nguyen et al., (2010). The transport and slaughter of animals leaving the system are not included. Veterinary medicines and hygiene products are not included because of lack of data in the enquiries. The functional units were 1 kg of live weight pig leaving the pig unit, including culled sows and slaughter pigs, and 1 ha of land occupied for the production of feed and the raising of animals.

2.3. Life cycle inventory analysis

The amount of complete feed used by the different categories of pigs was obtained from the enquiry, as well as their nutrient contents. However, no information was generally available on ingredients content. It is why these contents were estimated in a similar way as performed by Nguyen et al., (2010), assuming that the complete feed resulted from a mixture of cereals (wheat, barley and maize), protein rich ingredients (soybean meal, rapeseed meal and peas) and minerals (phosphate and calcium carbonate). This calculation was performed for all diets used by the different categories of pigs. A detailed description of the methodology used for the evaluation of impacts of production of non organic feed ingredients is given by Mosnier et al., (2011). Values for organic feed ingredients used in organic pig production systems were estimated from LCA food Database (2007).

Emissions to air were estimated for NH_3, N_2O, NO_x and CH_4. Emission of CH_4 from enteric fermentation and manure management were calculated from Rigolot et al., (2010a,b) and IPCC (2006). Direct N_2O emissions from manure during in-house and outdoor storage and during field application were calculated from IPCC (2006) and emissions of NOx were estimated according to Nemecek and Kägi (2007). NH_3 emission during in-house storage, outside storage and field application of manure were calculated from Rigolot et al., (2010a,b) according to type of effluent (slurry, solid manure) duration and type of storage and method of spreading. A description of the CML 2001 and CED methods can be found in Frischknecht et al., (2007).

2.4. Life Cycle impact assessment

The following impact categories were considered: climate change (CC), eutrophication potential (EP), acidification potential (AP), cumulative energy demand (CED), and land occupation (LO). The indicator result for each impact category was determined by multiplying the aggregated resources used and the aggregated emissions of each individual substance with a characterisation factor for each impact category to which it may potentially contribute. CC, EP, AP CED and LO were calculated using the CML2 "baseline" and "all categories" 2001 characterisation methods as implemented in the Ecoinvent v2.0 database. CC was calculated according to the 100-year global warming potential factors expressed in kg CO_2 equivalent (eq). EP was calculated using the generic EP factors in kg PO_4 (Guinée et al., 2002). AP was calculated using the average European AP factors in kg SO_2 eq (Guinée et al., 2002). Cumulative energy demand (CED, MJ) was calculated according to its version 1.05 as implemented in the Ecoinvent v2.0 database. Land occupation (m^2.yr) refers to on farm and off-farm area used for the production of feed or for the outdoor raising of pigs.

3. Results

3.1. Animal performance and system description

On average there were 310 sows per farm with sows, and farms with fattening pigs produced on average 3260 pigs per year (Table 1). The variability in average farm size per system (±270 sows, CV= 85% and ±1960 fattening pigs, CV=60%) was high with large differences between systems. Herd size was the highest for C and AC systems and the lowest for traditional systems, O systems being intermediate (Table 1). On average sows weaned 22.6 piglets per year. The highest performances were measured in C systems (26.9). Performances were slightly lower in AC systems (24.2) and the lowest in O and T systems (18.9 and 15.1, respectively). Consumption of feed per sows and per year was higher in T and O systems and this feed tended to be more concentrated in protein and phosphorus, compared to C and AC systems.

Feed conversion ratio during the post weaning period was 1.96 (±0.44) on average. It was the lowest for C systems and the highest for T ones (Table 1). Mortality rate (2.9% on average of systems) was markedly higher for T systems, with small differences among the others systems. Dietary crude protein content of post-weaning diets, 174 g/kg on average of systems, was lower in T systems (162 g/kg) and higher in O systems (193 g/kg). Total dietary phosphorus content was the highest in O systems with no marked difference among the other systems.

Table 1. Description of the pig production systems: performance of sows, piglets and fattening pigs, and average composition of diets.

	All systems		Conven-	Adapted	Traditional	Organic
	Average	Std[1]	tional	conventional		
Number of systems	15	15	5	5	3	2
Number of sows / farm[2]	310	270	395	475	59	128
Fattening pigs $year^{-1}$ $farm^{-1}$	3260	1960	4910	3570	510	2510
Sows						
piglets weaned / year	22.6	5.7	26.9	24.2	15.1	18.9
weaning weight / kg	8.4	1.8	7.3	7.4	9.3	12.1
feed per sow, kg/year	1390	312	1330	1340	1460	1590
crude protein, g/kg	138	14	134	134	137	158
total P, g/kg	5.0	0.6	4.7	4.9	5.2	6.0
Post-weaning						
final weight, kg	27.7	4.2	28.1	27.8	25.4	29.7
feed conversion ratio, kg/kg	1.96	0.44	1.67	1.90	2.42	2.20
mortality rate,%	2.9%	3.8%	1.9%	1.8%	7.0%	2.1%
crude protein, g/kg	174	19	175	173	162	193
total P, g/kg	5.6	0.5	5.5	5.6	5.5	6.4
Fattening pigs						
slaughter weight, kg	122	16.2	113	124	140	109
feed conversion ratio, kg/kg	3.44	1.37	2.74	3.18	5.29	3.03
mortality rate,%	3.5%	1.5%	3.4%	2.9%	4.5%	3.5%
crude protein, g/kg	155	14	157	153	145	174
total P, g/kg	4.7	0.475	4.7	4.5	4.8	5.1
Live weight / sow, kg/year	2570	555	2930	2840	1900	1990

[1] Standard deviation, [2]average for farms with sows

Average pig slaughter weight was 113 kg in C systems, rather close to O systems (109 kg). It was higher in AC and T systems, by 11 and 27 kg, respectively. Feed conversion ratio during fattening period was 3.44 (±1.37) on average. It was the lowest for C systems and the highest for T ones. Mortality rate (3.5% on average) was higher for T systems, with small differences among the others systems. Dietary crude protein content of fattening diets, 155 g/kg on average of systems, was lower in T systems (145 g/kg) and higher in O systems (174 g/kg). Total dietary phosphorus content was the highest in O and T systems with no marked difference between C and AC. Live weight pig produced per sow per year amounted 2570 kg on average of systems. It was higher in C and AC systems (2880 kg) and lower in T and O systems (1950 kg).

Conventional pigs were all housed indoor, on slatted floor and their manure was handled as slurry, only a small percentage of the slurry being treated. In AC systems slatted floor was also the most frequent but in some cases sows and/or fattening pigs were raised on straw bedding with the production of solid manure. In O systems animals were raised outdoor or indoor with outdoor access or in open buildings. The use of slatted floor was the most frequent for fattening pigs. In T systems sows might be raised outdoor or indoor, whereas fattening pigs were most often raised outdoor.

3.2. Environmental impacts of pig production

The environmental impacts of the different systems are presented per kg of pig produced and per ha of land occupied during a year (Table 2). There were large differences between systems for all impact categories expressed per kg pig produced. On average, CC, EP, AP, CE and LO amounted 2.61 (±27%; mean ± CV) kg eq CO_2, 0.022 (±41%) kg eq PO_4, 0.047 (±23%) kg eq SO_2, 18.2 (±26%) MJ , and 6.60 (±56%) m^2 per kg pig, respectively. There were substantial differences between extremes values for all impacts (up to x4). On average, CC per kg pig was the lowest for C and the highest for T (+54% compared to C), AC and O systems being intermediate. EP per kg pig was similar for C and AC systems; it was higher for T systems (+79%) and lower O systems (-16%). In the same way, AC per kg pig was similar for C and AC systems, whereas higher values were calculated for T and O systems (+23 and +29%, respectively). Energy demand per kg pig was the lowest for C and AC systems and was higher for O (+11%) and T (+50%) systems. Marked differences were found for LO, between C and AC systems, on one hand (4.5 m^2/kg pig), and T and O systems, on the other hand (9.9 m^2/kg pig).

When expressed per ha of land occupied, there were also large differences between systems for all impact categories (Table 2). On average, CC, EP, AP, CE and PP per ha, amounted 4680 (±26%) kg eq CO_2, 38.6 (±28%) kg eq PO_4, 86.3 (±30%) kg eq SO_2, 32.5 (±25%) TJ, and 1925 (±36%) kg pig per ha, respectively. There were marked differences between extreme values for all impacts. On average, CC per ha was the lowest for O and the highest for C and AC (+100% compared to O), T systems being intermediate. Eutrophication potential per ha was substantially lower for O systems; it was the highest for C systems (+170%) followed by AC and T. Acidification potential per ha was similar for O and T systems, whereas higher values were obtained for C and AC systems (+70 and +45%, respectively). In the same way, CED per ha was the lowest for O and T systems, and was higher for C (+98%) and AC (+75%) systems. Substantial differences were found for pig produced per ha land occupation, between C and AC systems, on one and (2300 kg/ha), and T and O systems, on the other hand (1170 kg/ha).

Table 2. Potential environmental impact expressed per kg pig produced or per ha of land use

	All systems		Conven tional	Adapted conventional	Traditional	Organic
	Average	Std[1]				
Number of systems	15	15	5	5	3	2
Impact per kg live weight						
Climate change, kg eq CO_2	2.61	0.70	2.25	2.55	3.47	2.35
Eutrophication, kg eq PO_4	0.022	0.009	0.019	0.020	0.034	0.016
Acidification, kg eq SO_2	0.047	0.011	0.044	0.044	0.054	0.057
Energy demand, MJ	18.2	4.6	16.2	16.5	24.3	18.1
Land occupation, m^2	6.30	3.52	4.13	4.78	10.6	9.14
Impact per ha land use						
Climate change, kg eq CO_2	4680	1220	5470	5320	3670	2610
Eutrophication, kg eq PO_4	38.6	10.7	46.3	41.4	35.3	17.3
Acidification, kg eq SO_2	86.3	26.2	106.1	89.9	63.8	61.6
Energy demand, MJ (x 1000)	32,5	8.0	39.4	34.8	25.7	19.98
Pig produced, kg LW	1925	684	2429	2162	1229	1114

[1] Standard deviation

4. Discussion

Results on environmental impacts of pig production evaluated with LCA were recently reviewed by de Vries and de Boer (2010). For CC the values obtained in the present study (2.25 to 3.47 kg eq CO_2 / kg pig) are within the large range of values (2.3 to 5.0 kg eq CO_2 / kg live pig) reviewed in that study. For conventional systems the observed average value (2.25 kg eq CO_2) is close to those reported by Basset-Mens and van der Werf (2005) and Nguyen et al., (2011): 2.3 and 2.2 kg eq CO_2, respectively. The value obtained for O systems (2.4 kg eq CO_2 / kg pig) is lower than those published for the same system by Halberg et al., (2010; 2.8 to 3.3 kg eq CO_2 / kg pig) and Basset-Mens and van der Werf (2005; 4.0 kg eq CO_2 / kg pig). The main reason for that difference is likely the higher animal performance in our study, both in terms of sow productivity and feed efficiency, and the higher N_2O emission in the study from Basset-Mens and van der Werf (2005) due to the use of straw bedding. Traditional systems have higher CC impact per kg pig. This is mainly due to the lower feed efficiency in these systems, in connection with the raising of traditional breeds. This results in a higher CC impact due to the production of feed, only partially compensated by decreased CH_4 emission due the outdoor raising of animals. AC systems have a slightly higher CC impact than C systems, mainly because reduced animal performance and the more frequent use of straw bedding with increased N_2O emission.

For EP the values obtained in the present study (0.016 to 0.034 kg eq PO_4 / kg pig) are also within the range of values (0.012 to 0.038 kg eq PO_4 / kg live pig) reviewed by de Vries and de Boer (2010). For C systems the observed average value (0.019 kg eq PO_4) is close to those reported for similar systems by Basset-Mens and van der Werf (2005) and Nguyen et al., (2011): 0.021 and 0.018 kg eq PO_4, respectively. The value obtained for organic production (0.016 kg eq PO_4 / kg pig) is lower than those published for this system by Basset-Mens and van der Werf (2005; 0.022 kg eq PO_4 / kg pig) and by Halberg et al., (2010; 0.025 to 0.038 eq PO_4 / kg pig), mainly because of higher animal performance in the present study. Among the evaluated systems, O systems have the lowest EP impact in connection with a much lower EP impact of feed in that system. For the same reason as for CC, T systems have the highest EP impact.

For AP the values obtained in the present study (0.044 to 0.057 kg eq SO_2 / kg pig) are also within the large range of values (0.008 to 0.120 kg eq SO_2 / kg live pig) reviewed by de Vries and de Boer (2010). For AC and AC systems the observed average value (0.044 kg eq SO_2) is close to those reported for similar systems by Basset-Mens and van der Werf (2005) and Nguyen et al., (2011): 0.044 and 0.043 kg eq SO_2, respectively. The value obtained for organic production 0.057 kg eq SO_2 / kg pig is higher than that published for this system by Basset-Mens and van der Werf (2010; 0.037 kg eq SO_2 / kg pig) and similar to those reported by Halberg et al., (2010; 0.050 to 0.061 eq SO_2 / kg pig). This is mainly related to the production of solid manure with reduced NH_3 emission in the study of Basset-mens and van der Werf (2005).

For CED the values obtained in the present study (16 to 24 MJ / kg pig) are within the large range of values (10 to 25 MJ / kg live pig) reviewed by de Vries and de Boer (2010). For C and AC systems the observed average value (16.3 MJ) is close to thoses reported for similar systems by Basset-Mens and van der Werf (2005) and Nguyen et al., (2011): 15.9 and 13.6 MJ, respectively. The observed value for organic production 18.1 MJ / kg pig is slightly lower than that published (22.2 MJ / kg pig) for this system by Basset-Mens and van der Werf (2005). In relation with the use of larger amounts of feed, T systems have the highest CED impact per kg pig.

The values obtained for LO in the present study (4.1 to 10.6 m^2 / kg pig) are partly outside the range of values (4.2 to 6.9 m^2 / kg live pig) reviewed by de Vries and de Boer (2010). This is mainly related to T and O systems which obtained higher values for LO. For T systems the main reason is the outdoor raising of fattening pigs. In the case of O systems the larger LO is mainly related to the higher LO impact for feed production, due to reduced yield of organic crops. For C systems the observed value (4.13 m^2 / kg pig) is close to those reported for similar systems by Basset-Mens and van der Werf (2005) and Nguyen et al., (2011): 5.4 and 4.4 m^2 / kg pig, respectively. The value obtained for organic production, 9.1 m^2 / kg pig, is close to the values published for this system by Basset-Mens and van der Werf (2010; 9.9 m^2 / kg pig) and Halberg et al., (2010; 6.9 to 9.2 m^2/kg pig).

When impacts are expressed per ha of land used, the ranking of systems is very different for most impacts. They are generally the lowest for O followed by T systems and the highest for C systems. The degree of intensification inversely correlates with the environmental impact per kg pig, whereas the opposite is found when the impact is expressed per ha. The same effect of the functional unit on the results was reported by Basset-Mens and van der Werf (2005). Our results clearly indicate that the choice of the functional unit has a major effect on the ranking of systems in terms of environmental impact in line with previous results (Basset-Mens and van der Werf, 2005). The use of plural functional units is rather common in the application of LCA in agriculture, but still under debate. As suggested by different authors this refers to two essential functions of agriculture: the production of food and land occupation. It is why some authors have suggested to adapt the choice of the functional unit to the category of impact, *i.e.* the kg of product for global impacts and ha of land occupation for local impacts (de Boer, 2003)

5. Conclusion

The diversity in production systems considered in the present study results in very large variations in all environmental impacts. However, the results depend on the functional unit. The degree of intensification inversely correlates with the environmental impact per kg pig, whereas the opposite is found when the impact is expressed per ha. According to the results from this study, LCA appears a suitable methodology for the evaluation of the environmental sustainability of pig production systems and can contribute to the overall assessment of sustainability.

6. Acknowledgements

The authors gratefully acknowledge the European Community financial participation under the 6th Framework Programme for Research, Technological Development and Demonstration Activities for the Integrated Project Q-PORKCHAINS (FOOD-CT- 2007-036245). The views expressed in this study are the sole responsibility of the authors and do not necessarily reflect the views of the European Commission. Neither the European Commission nor any person acting on behalf of the Commission is responsible for the use that might be made of the information. The information in this study is provided as it is, and no guarantee is given that the information is fit for any particular purpose. The user, therefore, uses the information at its sole risk and liability.

7. References

Basset-Mens, C., van der Werf, H.M.G., 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. Agric. Ecosys. Env., 105, 127-144.

Bonneau, M., Antoine-Ilari, E., Phatsara, C., Brinkmann, D., Hviid, M., Christiansen, M.G., Fàbrega, E., Rodríguez, P., Rydhmer, L., Enting, I., de Greef, K.H., Edge, H., Dourmad, J.Y., Edwards, S., 2011. Diversity of pig production systems at farm level in Europe. J. Chain and Network Sci., 11, 115-135

de Boer IJM 2003. Environmental impact assessment of conventional and organic milk production. Livest. Prod. Sci., 80, 69-77.

de Vries, M., de Boer, I.J.M., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessment. Livest. Sci., 128, 1-11.

Edwards, S.A., Dourmad, JY., Edge, H.L., Fabrega, E., de Greef, K., Antoine-Ilari, E., Phatsara, C., Rydhmer, L., Bonneau, M. 2008. Q-porkchains: tools for assessing sustainability of pigmeat production systems. Proc 59th Annual Meeting of the EAAP, Vilnius, Lithuania.

Frischknecht, R., Jungbluth, N. Althaus, H.J., Bauer, C., Doka, G., Dones, R., Hischier, R., Hellweg, S., Humbert, S., Köllner, T., Loerincik, Y., Margni, M., Nemecek, T., 2007. Implementation of Life Cycle Impacts Assessment methods. Ecoinvent report no. 3, v2.0. Swiss Centre for Life Cycle Inventories, Dubendorf, Switzerland.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., de Bruijn, H., van Duin, R., Huijbregts, M.A.J., 2002. Life Cycle Assessment. An operational guide to the ISO Standards. Centre of environmental science, Leiden University, Leiden, The Netherlands.

Halberg, N., Hermansen, J.E., Kristensen, I.S., Eriksen, J., Tvedegaard, N., Petersen, B.M., 2010. Impact of organic pig production on CO_2 emission, C sequestration and nitrate. Agr. Sust. Dev., 30, 721-731.

IPCC 2006. 2006 IPCC Guidelines for national greenhouse gas inventories. Eggleston HS, Buendia L, Miwa K, Ngara T and Tanabe K (Eds). IGES, Japan.

LCA Food database, 2007. Available online: http://www.lcafood.dk/

Mosnier, E., van der Werf, H.M.G., Boissy, J., Dourmad, J.Y., 2011. Evaluatio of the environmental implications of the incorporation of feed-use amino acids in the manufacturing of pig and broiler feed using Life cycle Assessment. Animal, 5, 1973-1983.

Nemecek, T., Kägi, T., 2007. Life cycle inventories of Swiss and European Agricultural production systems. Final report Ecoinvent report v 2.0., no. 15, Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for life cycle inventories, Zurich and Dübendorf, Switzerland.

Nguyen, T.L.T., Hermansen, J.E., Mogensen, L., 2010. Fossil energy and GHG saving potentials of pig farming in the EU. Energy Policy, 38, 2561-2571.

Petit, J., van der Werf, H.M.G., 2003. Perception of the environmental impacts of current and alternative modes of pig production by stakeholder groups. J. Env. Manag., 68, 377-386.

Rigolot, C., Espagnol, S., Pomar, C., Dourmad, J.Y., 2010a. Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part I: animal excretion and enteric CH_4, effect of feeding and performance. Animal, 4, 1401-1412.

Rigolot, C., Espagnol, S., Robin, P., Hassouna, M., Béline, F., Paillat, J.M., Dourmad, J.Y., 2010b. Modelling of manure production by pigs and NH_3, N_2O and CH_4 emissions. Part II: effect of animal housing, manure storage and treatment practices. Animal, 4, 1413-1424.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., de Haan, C., 2006. Livestock's long shadow. Environmental issues and options. Livestock, environment and development initiative. United Nations Food and Agriculture Organization, Rome.

Life cycle assessment of pig slurry treatment technologies

M. ten Hoeve[1,*], S. Bruun[1], L.S. Jensen[1], N.J. Hutchings[2], R. Jensen[2]

[1] University of Copenhagen, Faculty of Life Sciences, Department of Agriculture and Ecology, Thorvaldsensvej 40, 1871 Frederiksberg C, Denmark.
[2] Aarhus University, Department of Agroecology, Blichers Allé 20, 8830 Tjele, Denmark.
[*] Corresponding author. E-mail: *mtho@life.ku.dk*

ABSTRACT

Animal manure management is associated with negative impacts on global warming, acidification and eutrophication of ecosystems. In the current study an LCA is used to assess the environmental impacts of treatment of 1000 kg of slurry ex-animal of five different technologies. These technologies are *(a)* direct land application (reference scenario), *(b)* separation by mechanical screw press, *(c)* screw press separation and composting of the solid fraction, *(d)* separation by decanter centrifuge, and *(e)* decanter centrifuge separation with ammonia stripping of the liquid fraction. In all separation scenarios, the liquid fraction was land applied close to the farm and the solid fraction was transported and land applied 100 km from the farm. In general, the treatment technologies analysed in this LCA show environmental impact potential reduction compared with the reference scenario. The decanter centrifuge scenarios have equal or lower impact potentials than the screw press scenarios. Relative ranking of scenarios does not change after the sensitivity analysis, where field emission factors for N_2O, NH_3 and P were varied within the range observed in literature. Choice of the technology to implement in any given situation depends on the environmental problem in focus.

Keywords: life cycle assessment, pig slurry, slurry treatment technologies

1. Introduction

Large amounts of animal manure are produced in Europe and its management is associated with a number of environmental impacts. Impact categories that are mainly affected by animal manure management are *(a)* global warming potential, caused by methane and nitrous oxide emissions, *(b)* acidification potential, induced by ammonia emissions, and *(c)* eutrophication potential caused by nitrogen and phosphorus emissions.

These impacts are accentuated because an increasing share of agricultural land in Europe receives excessive amounts of animal manure. These so-called hotspot areas are characterised by high livestock densities and insufficient land for manure application. This has resulted in phosphorus surplus in these areas and associated risks for losses to the environment. In other areas, agricultural fields do not receive sufficient amounts of nutrients from manure and farmers need to apply mineral fertilisers to their fields. Non-renewable natural resources like phosphate rock, oil and natural gas, are used for the production of mineral fertiliser and there are considerable environmental emissions, such as CO_2 and phosphate leaching, related to the extraction, manufacturing and use of these fertilisers. In order to decrease the consumption of mineral fertiliser and avoid losses from areas with excessive availability of animal manure, geographical redistribution of animal manure needs to improve.

To enhance nutrient re-distribution, slurry treatment technologies have been developed that focus on the separation of slurry into a solid and a liquid fraction. The liquid fraction contains most of the easily available nitrogen but less than half of the phosphorus, so is mainly valued as a nitrogen fertiliser. However, the high water content in the liquid fraction makes the fraction relatively heavy and less suitable for long distance transportation. Agricultural land nearby the farm often has been treated with slurry for multiple years, which implies that phosphorus concentrations in the soil are already sufficiently high to supply the crop demand. The solid fraction is more transportable, due to its relatively low water content. It has a high concentration of slowly-available nitrogen and phosphorus, so is mainly valuable as a phosphorus fertiliser. Additional technologies that are developed to avoid environmental impacts include ammonia stripping from the liquid fraction, which reduces emissions of ammonia, various technologies for energy extraction and upgrading of the solid fraction such as composting which may improve the solid fraction as a soil amendment.

The objectives of this study are to determine environmental impact potentials of slurry treatment technologies in a Life Cycle Assessment (LCA) and to compare impact potentials of treatment technologies with a reference.

2. Methods

2.1. Model description

The LCA is based on information from literature research. The LCA itself was conducted by using the GaBi software (www.pe-international.com). The functional unit that forms the basis for the assessment is the handling of 1000 kg of slurry excreted by pigs. Emissions from slurry, liquid and solid fractions were analysed from the moment the slurry is excreted by pigs until 10 years after field application. The environmental impact categories that are included in the analysis are the climate change potential, including CO_2 (both biogenic and non-biogenic), CH_4 and N_2O, acidification potential, including NH_3, freshwater eutrophication potential, including P, and marine water eutrophication potential, including NO_3^- and NH_3. ReCiPe2008 is used for analysis of environmental impact potentials (Goedkoop et al., 2009).

2.2. Scenarios

For this LCA, it is assumed that the pigs are raised in Denmark, and that they consume an average Danish feed composition. The pigs are housed in pig housing with partially-slatted flooring. All liquid manure (slurry and liquid fraction) is assumed to be stored in a covered tank. Storage of the solid fraction is covered to reduce self-heating, and compost is well aerated. Liquid manure is applied by trailing hose to fields around the farm, with an average distance of 8 km from storage. It is assumed that liquid manure is a substation for mineral nitrogen fertiliser (ammonium nitrate). Solid fractions are transported to areas where phosphorus is needed, over an average distance of 100 km. As farmers on these fields with low phosphorus status would normally apply mineral phosphorus fertiliser, it is assumed that the solid fraction is a substitution for mineral phosphorus (single superphosphate) and nitrogen (ammonium nitrate) fertiliser. Phosphorus losses from the soil are calculated as a proportion of the difference between the phosphorus that is applied as fertiliser, and the average phosphorus need that plants have (assumed to be 21.5 kg P ha^{-1} yr^{-1}). If less phosphorus is applied than plants take up, a negative freshwater eutrophication potential results. Solid fractions are applied by broadcast spreading followed by rapid incorporation. Main emission factors are provided in Table 1.

Four treatment scenarios were compared to a reference scenario:

- Reference Scenario – Conventional manure management in the form of slurry, with in-house storage of 6 weeks, covered out-house storage for 6 months, and field application with a trailing hose
- Screw Press Scenario – Slurry separation with a mechanical screw press, with donor farm application of the liquid fraction, and transportation to and application on a recipient farm of the solid fraction
- Screw Press with Composting Scenario – As Screw Press Scenario with the difference that the solid fraction is windrow composted before transportation to recipient farm
- Decanter Centrifuge Scenario – Slurry separation with a decanter centrifuge, with donor farm application of the liquid fraction, and transportation to and application on recipient farm of the solid fraction
- Decanter Centrifuge with Ammonia Stripping Scenario – As Decanter Centrifuge Scenario with the difference that ammonia is stripped from the liquid fraction resulting in a compact nitrogen fertiliser

Table 1: LCI parameters used for in-house storage, ex-house storage and field application.

	NH_3-N	N_2O-N	N_2-N	NO_3^--N	Degradation	CH_4-C	CO_2-C	MFE
	kg/kg TAN-N	kg/kg N	kg/kg N	kg/kg N	kg/kg OM	kg/kg OM deg.	kg/kg OM deg.	%
House	0,250				0,185	0,150	0,310	
Storage								
slurry / liquid	0,013				0,185	0,230	0,230	
solid	0,003	0,002	0,004		0,070	0,230	0,230	
compost	0,004	0,008	0,024		0,600	0,060	0,400	
reject	0,950							
Field application								
slurry	0,160	0,020	0,041	0,395				75
liquid	0,120	0,020	0,038	0,405				85
solid	0,390	0,020	0,038	0,332				65
compost	0,390	0,020	0,050	0,304				45
reject	0,950	0,020	0,038	0,405				85
NH_3-fration	0,020	0,020	0,025	0,418				100
min. fertilizer	0,027	0,020	0,024	0,407				100

TAN = total ammoniacal nitrogen, OM = organic matter, reject is the liquid fraction after ammonia stripping, NH3-fraction is the ammonia-rich fraction after ammonia stripping.

3. Results and discussion

3.1. Main model

A comparison of the treatment scenarios with the reference scenario is done for four impact potential categories. The reference scenario is set to 100%. For the treatment scenarios, values higher than 100% imply that the environmental impact is higher in the treatment scenario than in the reference scenario. *Figure 1* provides an overview of the relative climate change potential. The screw press with composting scenario has an impact potential that is 11.5% higher than the reference scenario due to CO_2 (24.2 kg CO_2-eq), CH_4 (33.1 kg CO_2-eq), and N_2O (2.7 kg CO_2-eq), emissions during the composting process. All other treatment scenarios have a lower global warming potential than the reference. The decanter centrifuge scenario show the lowest impact potential.

All treatment scenarios show a lower freshwater eutrophication potential than the reference scenario, see *Figure 1*. For the two screw press scenarios, this reduced impact potential is approximately 24% smaller than in the reference, while for the decanter centrifuge with ammonia stripping scenario, this reduced impact potential is approximately 55% smaller than in the reference.

Application of phosphorus from the liquid fraction in the centrifuge scenario is lower than the plant-need. 71% of the phosphorus in slurry ends up in the solid fraction after centrifuge separation. Consequently, 29% of the initial phosphorus ends in the liquid fraction. Nitrogen separation is less efficient, resulting in a relatively larger share of nitrogen in the liquid fraction compared to phosphorus. As field application is assumed to follow application limit of 170 kg N ha^{-1} yr^{-1} for nitrogen, a smaller amount of phosphorus in the liquid fraction is spread in the centrifuge scenario. Another explanation for the negative contribution to freshwater eutrophication for the two centrifuge scenarios is that the production of mineral fertiliser contributes to freshwater eutrophication. The production of mineral phosphorus fertiliser is replaced in the scenarios where the solid fraction is applied to fields that are low in phosphorus. In the decanter centrifuge scenarios, a large share of phosphorus is in the solid fraction, 71%, and thus replacing 0.45 kg P_2O_5 (mineral phosphorus fertiliser). After screw press separation, only 17% of initial phosphorus is in the solid fraction, implying a replacement of 1.95 kg P_2O_5 (mineral phosphorus fertiliser).

Looking at Fig. 1, it can be seen that all treatment scenarios have a lower contribution to marine eutrophication than the reference scenario. In the ReCiPe2008 method, marine eutrophication potential is calculated from the emissions of nitrogen, because growth in saltwater bodies is assumed to be limited by nitrogen. Reductions range from 29% less for the screw press with composting scenario to 39% for the centrifuge with ammonia stripping scenario. This is mainly caused by the assumed mineral fertiliser replacement efficiency for the different slurry fraction. With 45%, the lowest replacement efficiency is assumed for the screw press with composting scenario, while the highest is assumed for the stripped ammonia, with 100%.

The relative acidification potential ranges from 86% for the centrifuge with ammonia stripping scenario, to 102% for the screw press with composting scenario. The main contribution in all scenarios is caused during storage under the partly-slatted flours in the pig housing. As the storage during housing is similar in all scenarios and treatment first takes place after this stage, differences in acidification potential for the scenarios are mainly caused after field application of the slurry. For the centrifuge with ammonia stripping scenario, emissions during treatment are higher than in any other scenario due to increased pH. However, this increase during treatment is more than compensated for after field application, leading to a net reduction in ammonia loss from the complete stripping process. The increased acidification potential in the screw press with composting scenario is due to the addition of rape straw during composting, increasing the total amount of nitrogen.

A noticable difference between this research and that of others is that biogenic carbon is included in the analysis in this study but not in the others (De Vries et al., 2012; Lopez-Ridaura et al., 2009; Prapaspongsa et al., 2010). Both De Vries et al., (2012) and Prapaspongsa et al., (2010) show a negative contribution to climate change field, which implies that the use of mineral fertiliser has a higher impact on climate change than the use of pig slurry. Lopez-Ridaura et al., (2009) and the current research show the opposite.

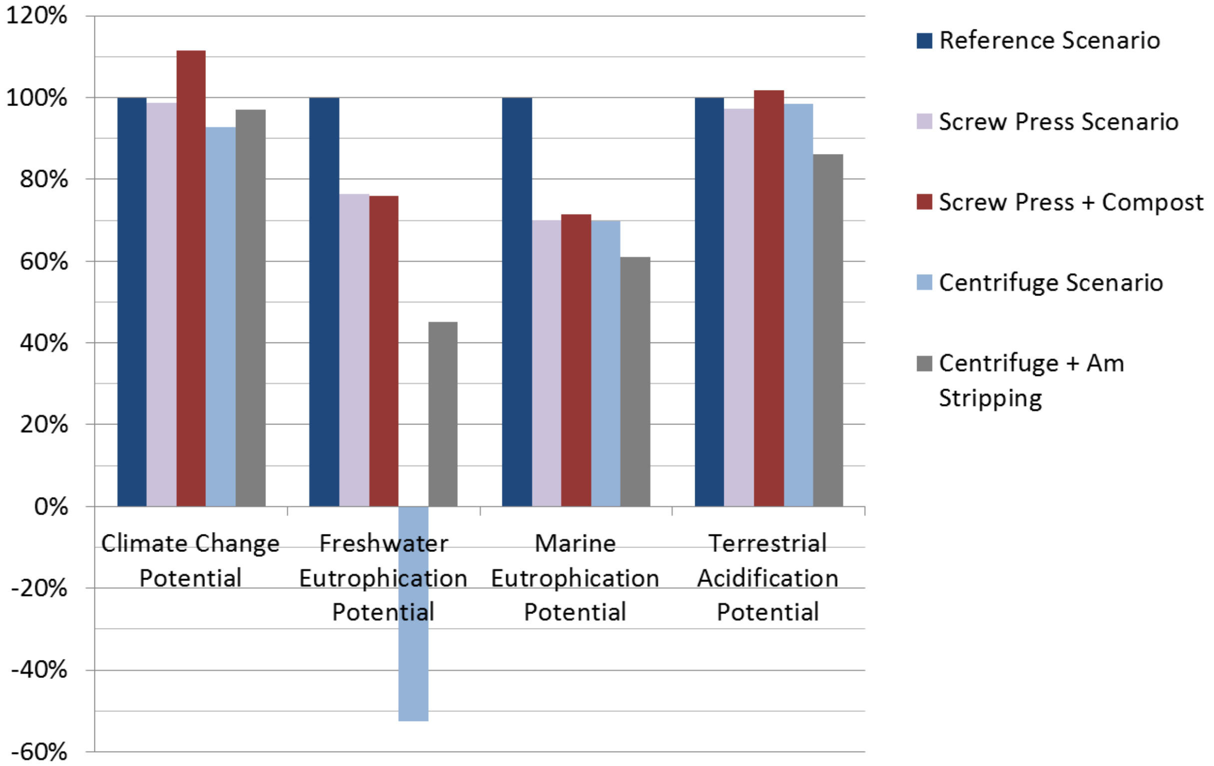


Figure 1. Relative Climate Change Potential, Freshwater Eutrophication Potential, Marine Eutrophication Potential and Terrestrial Acidification Potential of treatment scenarios compared to reference (100%).

3.2. Sensitivity analysis

In the sensitivity analysis, the effect of varying three parameters was analysed, N_2O emission rate, NH_3 emission rate and P loss rate. The parameters were all related to emissions after field application, as large uncertainties are associated with these estimates (Hansen et al., 2008; IPCC, 2006; Nielsen & Wenzel, 2007).

3.2.1. N_2O emission rate (field)

According to the IPCC (2006), the N_2O emission rate after field application of slurry ranges from 0.7% to 6.0% of total N. These values were included in the sensitivity analysis, and results are shown in *Figure 5*. For the low emission model, the global warming potential is decreases the least with 16% for screw press with composting scenario to the most for the centrifuge scenario with 19%. For increasing the N_2O emission rates to 6.0%, again the screw press with composting scenario is affected the least with 49% and the centrifuge scenario is affected the most with 59%. The relative ranking of scenarios is not changed by editing the N_2O emission rate.

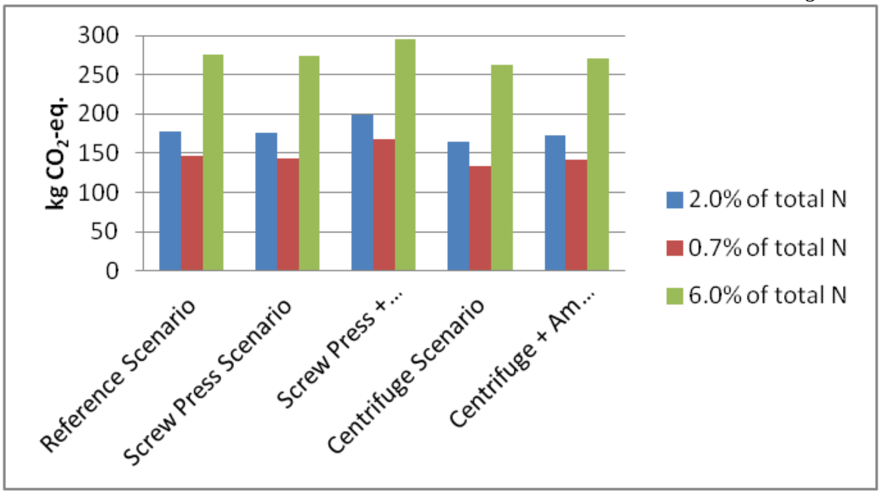


Figure 2. Sensitivity of climate change potential to field N_2O emission rates of different slurry fractions.

3.2.2. NH_3 emission rate (field)

Values for the sensitivity analysis for the NH_3 emission rate are based on ranges provided in Hansen et al., (2008). Table 1 provides an overview of the emission rates that were analysed.

Table 2: NH_3-N emission rates in main sensitivity models (% of TAN-N).

	Main model	Low	High
Slurry	16	3.1	20
Liquid fraction	12	1.6	10
Solid fraction	39	13	65
Compost	39	13	65

The impact category that is affected most by the changing emission rates is marine eutrophication (Table 3). Lower NH_3 emission rates (Table 2) correspond to a higher contribution to marine eutrophication. Marine eutrophication is mainly caused by NO_3^- emissions to water bodies and indirectly by NH_3 emissions to the air. It is assumed that NH_3 emissions mainly take place within the first hours to days after land application, NO_3^- emissions take place later in the 10 year period that is tracked. Therefore NO_3^- emissions depend on how much N is removed from the soil system in the form of NH_3. More NH_3 emitted means a decrease in NO_3^- leaching. The relative ranking of scenarios is not changed by editing the NH_3 emission rate.

Contrary to the marine eutrophication potential, total impact on terrestrial acidification decreases if the NH_3 emission rates decrease. NH_3 is a main contributor to terrestrial acidification. Therefore, the impact potential decreases if the NH_3 emission decreases. The ammonia stripping scenario decreases less than other scenarios. In this scenario only a small share of N is applied in the regular way, the rest is converted into mineral fertiliser. For the sensitivity analysis with high NH_3 emission rates (Table 2), mainly rates for the solid fraction and compost are higher than in the main model. The emission rate of liquid fractions is lower than in the main model. The relatively low sensitivity of the screw press scenario to high NH_3 emission rates is due to the relatively large amount of liquid fraction in this scenario.

Table 3: Changes in marine eutrophication potential and terrestrial acidification potential after sensitivity analysis for field NH_3 emission rates compared to main model.

	Marine eutrophication potential		Terrestrial acidification potential	
	Low	High	Low	High
Reference Scenario	34%	-10%	-28%	8.4%
Screw Press Scenario	44%	-1.5%	-27%	2.5%
Screw Press + Composting	46%	-3.7%	-29%	5.1%
Centrifuge Scenario	45%	-4.8%	-28%	4.8%
Centrifuge + Am Stripping	13%	-13%	-10%	10%

3.2.3. P loss rate (field)

Changes in the loss rate of P that is applied to fields only influence freshwater eutrophication. The influence is large, also with minor changes to the loss rate. As in Nielsen and Wenzel (2007), the sensitivity analysis is calculating the net model outputs if P loss rates are varied from 0% to 100% of P surplus. The model with 100% loss of P surplus, is more than 1 order of magnitude larger than the main model. The model with 0% loss of P surplus leads to an emission of a freshwater eutrophication potential that is close to 0. The slightly negative values for the four scenarios with slurry separation, are due to losses during production of mineral P fertiliser.

Table 4: Sensitivity of freshwater eutrophication potential to field P loss rates.

	5% of P surplus	**0% of P surplus**	**100% of P surplus**
Baseline Scenario	0,022	0,000	0,445
Screw Press Scenario	0,017	-0,001	0,352
Screw Press + Composting	0,017	-0,001	0,352
Centrifuge Scenario	-0,012	-0,003	-0,183
Centrifuge + Am Stripping	0,010	-0,003	0,253

4. Conclusion

In general, the treatment technologies analysed in this LCA show environmental impact potential reduction in comparison to a reference scenario. The only scenario that seems to have a higher impact potential, with respect to the global warming potential and acidification, is the scenario with screw press separation and composting of the solid fraction. The decanter centrifuge scenarios show higher impact potential reductions than the screw press scenarios. The sensitivity analysis shows that results are particularly sensitive to phosphorus loss rates. For all variables included in the sensitivity analysis, relative scenario ranking does not change.

5. Acknowledgements

The research was performed as a part of research project “CLEANWASTE” funded by The Danish Council for Strategic Research.

6. References

De Vries, J.W., Groenestein, C.M., De Boer, I.J.M., 2012. Environmental consequences of processing manure to produce mineral fertilizer and bio-energy. J Environ Manage 102, 173-183.

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., van Zelm, R., 2009. ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level No. Report 1: Characterisation). The Hague: Ministry of Housing Spatial Planning and the Environment.

Hansen, M.N., Sommer, S.G., Hutchings, N.J., Sørensen, P., 2008. Emmision factors for calculation of ammonia volatilization by storage and application of animal manure. DJF Husdyrbrug nr. 84. Published: Aarhus University, Tjele.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, Eggleston, HS., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (eds.), Published: IGES, Japan.

Lopez-Ridaura, S., Van der Werf, H., Paillet, J.M., Le Bris, B., 2009. Environmental evaluation of transfer and treatment of excess pig slurry by life cycle assessment. J Environ Manage 90, 1296-1304.

Nielsen, P.H. & Wenzel, H. 2007. Environmental assessment of Ronozyme® P5000 CT Phytase as an alternative to inorganic phosphate supplementation to pig feed used in intensive pig production. Int. J LCA 12, 514-520.

Prapaspongsa, T., Christensen, P., Schmidt, J.H., Thrane, M., 2010. LCA of comprehensive pig manure management incorporating integrated technology systems. J Clean Prod 18, 1413-1422.

Communication of LCA results in the French environmental experimentation context: user-friendly web-based tool for the case of coffee

Simone Pedrazzini[1,*], Carole Dubois[1], Raphael Halloran[1], Christophe Porté[1], Sandi Northey[2], Urs Schenker[3], Sebastien Humbert[1]

[1] Quantis, Parc scientifique EPFL, Bât. D, 1015 Lausanne, Switzerland
[2] Nestlé, Av. Nestlé 55, 1800 Vevey, Switzerland
[3] Nestlé Research Center, Vers-chez-les-Blanc, 1000 Lausanne, Switzerland
[*] Corresponding author. E-mail: *simone.pedrazzini@quantis-intl.com*

ABSTRACT

In the French environmental experimentation context, the communication of life cycle assessment (LCA) results to the general public is one of the main challenges of this initiative. The *Nescafé® LCA Communication Tool* is presented as an example of user-friendly interactive solution to communicate the results of an LCA made on a cup of coffee. The main advantages of this kind of communication are the limited amount of information on display at a time, allowing analysis of complex results of an LCA in a simple and engaging way, as well as creating interactivity which attracts the attention of the user.

Keywords: French labelling experimentation, Grenelle, communication, web-based tool, coffee, multi-indicators

1. Introduction

Since the 1st of July 2011 a French experimentation on environmental labelling of consumer products is on-going. This experimentation results from the "Grenelle de l'Environnement" initiative, promoted by the French Parliament (Cros et al., 2010). The objective of the Grenelle initiative is to inform consumers about the environmental performance of consumer products throughout their life cycle, as a choice criterion when making a purchase.

One of the main challenges of this initiative is the communication of life cycle assessment (LCA) results to the general public; that is moving from scientific report based results to simple and user-friendly means of communication. In this context, Nestlé contracted Quantis to help develop a web-based tool for *Nescafé®* - one of the products participating in the French pilot. The result is the *Nescafé® LCA Communication Tool*, which presents the results of an LCA made on a cup of coffee. The tool is a Nestlé proprietary initiative intended to help communicate LCA results and understanding to the general public. A key aim was to develop an interactive web-based tool that displays results in an attractive and user-friendly way (http://nescafe.outil-acv.com/).

2. Methods

The full life cycle impacts of a cup of spray dried soluble coffee are based on Humbert et al., (2009) and are assessed for the climate change, water consumption and land occupation impact categories, using IMPACT 2002+ (Jolliet et al., 2003, as adapted by Humbert et al., 2012). The results are compliant with the standard developed for the French experimentation (BPX 30-323 2011) and follow the recommendations provided by the specific working group concerning food products leaded by ADEME-AFNOR.

3. Results and discussion

The Nescafé® LCA Communication Tool allows consumers to analyse the results per life cycle stage and per indicator. Consumers can "navigate" through the results and discover for themselves the environmental impacts of the different life cycle stages on the selected environmental indicator. This allows displaying complex results of an LCA in a simple and engaging way by having only a limited amount of information on display at a time.

The home page presents the seven life cycle stages assessed (see Figure 1). The user can both click on each life cycle stage to analyse its contribution for the three environmental indicators or click on a single environmental indicator to analyse the contribution of each life cycle stage (see Figures 2, 3, and 4).

Figure 1. *Nescafé® LCA Communication Tool* interface, "From bean to cup".

The greenhouse gas emissions indicator has a total score of about 28 g CO_2-eq./cup of coffee and the agriculture life cycle stage represents the main contribution, mainly due to fertiliser use, as seen in Figure 2 below.

Figure 2. *Nescafé® LCA Communication Tool*, LCA results for the greenhouse gas emissions indicator.

In terms of water consumption (relative), the total score is about 1.4 dl water-eq/cup of coffee and it is mainly due to crop irrigation during the agriculture life cycle stage, as seen in Figure 3 below. This indicator represents the amount of water consumed (i.e. evaporated, integrated into the product or moved to another watershed) at each life cycle stage weighted by the local water stress indicator. Therefore this indicator can be described as evaluating the amount of water consumed that is "in competition with another user".

Figure 3. *Nescafé® LCA Communication Tool*, LCA results for water consumption (relative) indicator.

The utilised surface (relative) indicator has a total score about 300 cm^2/cup of coffee of utilised surface due almost exclusively to coffee plantations, as seen in Figure 4 below. This indicator represents the amount of surface used weighted by its specific pressure on biodiversity: lower the pressure on biodiversity lower the indicator will be.

Figure 4. *Nescafé® LCA Communication Tool*, LCA results for utilised surface (relative) indicator.

To be transparent, the methodology used is described giving details about the environmental indicators definitions and the hypotheses used in this study (Figure 5).

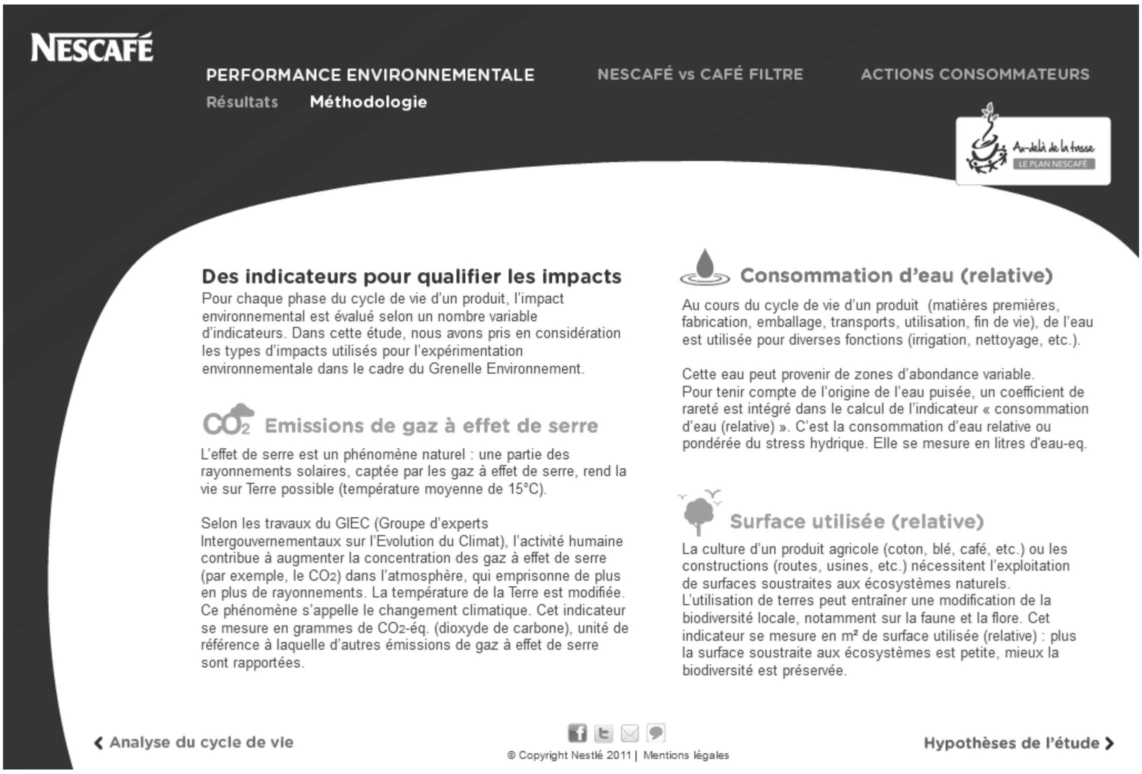


Figure 5. *Nescafé® LCA Communication Tool*, description of the indicators assessed.

4. Conclusion

In the French environmental experimentation context, the communication of LCA results to the general public is one of the main challenges of this initiative. The *Nescafé® LCA Communication Tool* is an example of user-friendly interactive solution that allows consumers to analyse for themselves the results per life cycle stage and per indicator. The main advantages of this kind of communication are the limited amount of information on display at a time, allowing analysis of complex results of an LCA in a simple and engaging way, as well as creating an interactivity which attracts the attention of the user. The “Grenelle de l'Environnement” initiative is a multi-indicator approach which allows coverage of a wider range of environmental impacts (e.g. it is not only focused on carbon footprint). A multi-indicator approach results in a communication challenge. The idea to show a limited amount of information on display at a time is a good option to overcome this challenge.

While it is clear that consumers are still becoming familiar with a life cycle assessment approach to understanding environmental impacts, this kind of solution can be a valuable start in moving from scientific report based results to more engaging consumer communication.

5. References

BP X 30-323, 2011. Principes généraux pour l'affichage environnemental des produits de grande consommation. Référentiel de bonnes pratiques. AFNOR. France.

Cros, C., Fourdrin, E., Réthoré, O., 2010. The French initiative on environmental information of mass market products. International Journal of Lice Cycle Assessment 15, 537-539.

Humbert, S., Loerincik, Y., Rossi, V., Margni, M., Jolliet, O., 2009. Life cycle assessment of spray dried soluble coffee and comparison with alternatives (drip filter and capsule espresso). Journal of Cleaner Production 17, 1351-1358.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., Rosenbaum, R., 2003. IMPACT 2002+: a new life cycle impact assessment method. International Journal of Life Cycle Assessment 8(6), 324–30.

Humbert, S., De Schryver, A., Margni, M., Jolliet, O., 2012. IMPACT 2002+: User Guide, Draft for version Q2.2 (version adapted by Quantis). Available at http://www.quantis-intl.com.

Nescafé website, 2011. Available at http://nescafe.outil-acv.com.

Food labelling from a consumers' perspective

Ulrike Eberle

Ulrike Eberle, corsus – corporate sustainability, Nernstweg 32-34, D-22765 Hamburg, E-mail: *u.eberle@corsus.de*

ABSTRACT

In particular in the food sector the variety of products is enormous. Appropriate labels on food attributes that consumers have to trust in could play a key role in supporting more sustainable and environmental friendly purchasing decisions. Various studies with a focus on the German market and on environmental impacts have been analysed and brought together. The results show that LCA could be the method of choice, but research on some environmental impact categories and how to include them best in LCA is still necessary. However, most important from a consumers perspective is that the information in the label is presented in a way that it is reliable and that it really supports sustainable purchasing decisions. Thus, research has to be carried out how to 'translate' LCA results appropriately and how labels are used by consumers.

Keywords: food labelling, consumers' perspective, environmental impacts, useability

1. Introduction

In particular in the food sector the variety of products is enormous. Large stores in Germany sell more than 30,000 different articles, the market as a whole offers more than 100,000 articles. Thus, consumers have to choose between huge varieties of products, some of them quite similar. For decision making they can use the information given on the product. Some of these declarations are obligatory, like the list of ingredients, best-before-date, or nutritional information; some are voluntary, like carbon labels or animal welfare labels and mark food attributes consumers have to trust in. Because of the amount of different declarations it is in most cases really difficult for consumers to handle all these detailed information (Eberle et al., 2011). Since the 1950s it is known very well that most people are only able to take seven (plus or minus two) different information into account for their purchasing decision - the so called 'magical number seven' (Miller 1956). On the other side, information overload can lead to excessive demands and a refusal to even study the offered choice (Walsh 2002).

In consumer research the role and success factors of signals like labels as effective information instruments have been under investigation for a long time. Hence, it is known that consumers simplify decisions via selective perception of information (e.g. Peter et al., 1999). In this kind of consumer research labels are understood as 'information chunks', which are particularly important for comparing products and which substitute and bundle other information (Kröber-Riel & Weinberg 2003, p. 284). Consumers expect essential or even sufficient information about the product or process quality from labels. But one has to keep in mind that information has to be as simple as possible and at the same time as exact as possible. Thus, labels "can play a key role when it comes to trust-related properties of products or services, as consumers do not have a reliable alternative source of this information" (Eberle et al., 2011, p. II).

Against this background the recent practice regarding labelling of food attributes that consumers have to trust in like environmental impacts or social issues is analysed with a focus on Germany. The analysis focusses mainly on environmental issues and identifies problems in recent practice.

2. Methods

For this purpose the findings from different studies and strategy papers of commissions and Advisory boards on food labelling have been analysed and brought together. The analysis has been carried out from a 'consumers' perspective' – a research methodology developed in the German research project 'Ernährungswende' (www.ernaehrungswende.de) funded by the German Ministry for Education and Research (Hayn et al., 2005).

3. Results

In general, a "label means any tag, brand, mark, pictorial or other descriptive matter, written, printed, stencilled, marked, embossed or impressed on, or attached to, a container of food" (WHO/FAO 2007, p. 2). Furthermore, consumers also understand results of product tests as labels, i.e. the test results of the German 'Stiftung Warentest' which gives the opportunity to print the results on the product. Also brands can be seen as labels from a consumers' perspective in particular if concrete product qualities are promised with the brands name or logo (Eberle et al., 2011).

In the past years more and more attributes of confidence, such as environmental or social issues, are labelled in food products, some claim for less environmental impacts or better environmental performance, others for better animal welfare, or for more sustainability of the labelled food product. Some labelling schemes are already in place, e.g. fair trade or organically grown products; others such as climate labels, labels on animal welfare or labels that show aggregated sustainability or environmental aspects are in place in few countries, but in most countries they are still under development or ready to push into the market.

3.1 German food labels

In the German food market ten different labels for organically grown products could be found[3]. One of them is the obligatory EU Label for organic food. Another one is the German 'Bio-Siegel' which is with 90% the best known food label in Germany. In contrast, the EU Label for organic food is known only by 14% of German consumers. The other labels belong to the different organically growers associations like Demeter or Bioland which are the two biggest. These labels are also known by consumers - the better the bigger the association. However, the information given with the labels is not really clear to most consumers (Buxel 2010).

According to Eberle (2012) at the end of 2011 four different labels could be found in the German food market which claimed for less greenhouse gas emissions: one of them is a label of an independent organisation, which could be obtained only for organically grown food products. Three of them are self-declarations of companies.

In addition there exists a variety of further food labels. Some belong to independent organisations like Marine Stewardship Council (MSC) for fish and fish products caught with practices that avoid overfishing; Fair Trade, for food that is produced following the standards of fair trade along the value chain; or Rainforest Alliance, for food products from tropical countries e.g. coffee that follows the standards for a more sustainable coffee cultivation. Some are based on legislation like the label on genetically modified organism (EC regulations 1829/2003 and 1830/2003) or the national German label "GMO free" (EGGenTDurchfG) (Eberle 2011). Others are company claims like the ProPlanet Label of the REWE Group (Eberle 2012).

3.2 Methodological basis and methodological challenges

Sengstschmid et al., (2011) showed in their study that only single issue labels (e.g. on climate change) are based on a life cycle approach like the British Carbon Trust's carbon label. Also in Germany the labels based on a life cycle approach are single issue labels (Eberle 2011, 2012): Three of the carbon labels on the German market are based on a life cycle approach (ISO 14040series, PAS 2050/German Memorandum Product Carbon Footprint (Grießhammer and Hochfeld, 2009), or a comparable self-developed standard (Stop Climate Change, 2008)); one of them gives no information on the standard on which the calculation of greenhouse gas emissions and assessment of impacts on climate change is based upon.

Nevertheless various labels which are not based on a life cycle approach cover a broad range of important environmental impacts with their standards, but do not quantify them nor compare them to the impacts of similar products, e.g. labels for organically grown products (Sengstschmid et al., 2011). It is state of the art that at the moment no label for food or meals exists that is based on an assessment of the environmental performance of the product's life cycle (Eberle et al., 2011, Sengstschmid et al., 2011). Only the French approach (Grenelle de l'Environnement, http://www.legrenelle-environnement.fr/) has foreseen that the analysis of the product's life cycle will form the basis of the information given by the label.

Sengstschmid et al., (2011) asked stakeholders and consumers which would be the main issues and important impact categories to be covered by an environmental food label. According to their results the main environmental issues and important environmental impact categories would be waste and development of recycling systems, water usage, water pollution, greenhouse gas emissions, eco-toxicity and pesticide use. From an expert point of view there are some important issues lacking related to environmental impacts of food, such as land use, degradation of soils, acidification, quality of water bodies, eutrophication, and impacts on ecosystems and biodiversity. Others like development of recycling systems have not been ranked in first priority by experts (e.g. SRU 2002). Having the need to quantify those important issues/impact categories it could be easily seen that most of them could be quantified best by using life cycle assessment (LCA) according to ISO 14040 series. In contrast, others are up to date quite difficult to quantify with LCA. Amongst them is the impact on ecosystems and biodiversity for which different methodological approaches are in discussion (e.g. Müller-Wenk 1998, Brentrup et al., 2002, Kyläkorpi et al., 2005, Lindner 2008, Milà i

[3] http://www.nachhaltigkeitsrat.de/news-nachhaltigkeit/2011/2011-04-21/der-nachhaltige-warenkorb-rne-einkaufsfuehrer-aktualisiert/?blstr=0

Canals et al., 2007, Michelsen 2008), none of them ready to be used by default. But also degradation of soils is difficult to cover with the LCA approach. For some impact categories still exists a scientific discussion about which assessment method would be best, e.g. eco-toxicity (e.g. Pant et al., 2005), for other important impacts it is quite difficult and complex to get valid and appropriate data, e.g. water use (e.g. Morrison et al., 2010).

Labels have the need for simple, aggregated information that could be easily understood. Thus, a methodology that allows aggregating the results of the assessment of the various environmental indicators is needed (Eberle et al., 2011). But up to date, there is no consensus about how to do such an aggregated assessment of the environmental performance of products, even though various approaches have been developed (e.g. MIPS (Schmidt-Bleek, 1997), EcoGrade (Grießhammer et al., 2007)). In France, currently relevant indicators for the declaration of food products' environmental performance will be defined; obligatory is to take greenhouse gas emission into account. This is done within a stakeholder approach (http://affichage-environnemental.afnor.org/actualites/liste-des-gt-sectoriels). Furthermore a public database is under development containing the relevant life cycle data for French agricultural raw products (van der Werf et al., 2010).

3.3 Status quo or process label?

Almost all labels at least on the German market set standards which have to be fulfilled, i.e. the standards for organic agriculture, or they state the status quo of the environmental performance of one environmental indicator, e.g. carbon labels (Eberle 2011). In contrast, labels which in addition support development processes - so-called 'process labels' (Eberle 2002) - are still rare, but the need for such labels is seen. Recently the Scientific Advisory Boards on Consumer and Food Policy and on Agricultural Policy at the German Federal Ministry of Food, Agriculture and Consumer Protection recommended such a label for food (state owned or supported) in their Strategy Paper on Food Labelling (Eberle et al., 2011). The intention of such a label is to stimulate further development towards more sustainability or environmental friendliness.

The Advisory Boards also stated that it is very important for the success of such a label that empirical analyses will be performed which examine how consumers understand and use the information given by labels. This should be done before launching a new label and continuously after the launch. Up to date such investigations are still rare, none of the state labels like the EU Label for organically grown food or the German 'Bio-Siegel' carries out such research. However, the French Grenelle de l'Environnement has foreseen a phase to evaluate the experiences which have been made within the test phase with respect to consumers and their understanding of the tested labels.

3.4 Recommendations

The analysed studies give recommendations how to face the identified problems. Aim has to be to avoid misguidance of consumers, to avoid misuse of labels only for marketing purposes, and to support more sustainable or more environmental friendly purchasing decisions through credible and meaningful labels. Thus, the analysed studies recommend the following (VK BaWü 2011, Eberle 2011, Eberle et al., 2011, Eberle 2012):

- The basic requirements for product labels and information on food attributes that consumers have to trust in should be publicly available and this should be fixed by law. These are requirements like label criteria, labelling authority, monitoring, control and financial dependencies.
- An information tool should be established that informs about the labelling schemes. Such a tool should be independent from the labelling authorities and should present background information and evaluations on the different labels schemes.
- The useability of the label for consumers has to be assured. Thus, accompanying empirical research (ex-ante and continuously) is urgently necessary to understand how consumers use and understand the information given with the label.
- Binding methodological approaches (e.g. LCA, product category rules for different food products, assessment and aggregation methodology) have to be developed and agreed upon. Furthermore a consistent LCA database has to be provided.

4 Discussion and conclusions

Labels "can play a key role when it comes to trust-related properties of products or services, as consumers do not have a reliable alternative source of this information" (Eberle et al., 2011, p. II).

Today in the German food market a huge variety of labels is in place. Up to date there are no food labels in place that consider the whole products life cycle based on a LCA approach and at the same time cover all important environmental impact categories. However, LCA would certainly be the most appropriate approach to get insights on a products' environmental performance even if today no commonly accepted approach exists to assess and aggregate the environmental performance of food. In France such an approach is under development. The approach is promising, in particular because the useability of the labels tested in the pilot phase will be analysed and because of the involvement of various stakeholders. This is important because results for the German market have shown that the useability of many labels for consumers is low because it is sometimes difficult for consumers to understand what the message of the label is. Another important fact is that it is quite difficult for consumers to find out at the point of sale which of the food labels is reliable. But the success of an environmental food label will very much depend on the reliability of a labelling scheme.

Thus, from a consumers' perspective few but reliable food labels are needed in the market which deliver clear and simple messages. In order that this could be achieved research is still needed:

- Research on the translation of LCA results in understandable and thus appropriate information for consumers that could be used in labels that really support more environmental friendly purchasing decisions;
- on the determining factors of useability of labelling schemes for consumers, and
- on an appropriate and thus agreeable aggregating assessment methodology for the environmental performance of food.

Furthermore, politics is requested to support labelling of attributes of confidence by providing a general setup that prevents misuse of labels and misguidance of consumers.

5. References

Brentrup, F.; Küsters, J.; Lammel, J.; Kuhlmann, H., 2002. Life Cycle Impact Assessment of Land Use Based on the Hemeroby Concept, Int J LCA, pp. 339 - 348

Buxel, H., 2010. Akzeptanz von Güte- und Qualitätssiegeln auf Lebensmitteln. Ergebnisse einer empirischen Untersuchung. Fachhochschule Münster

Eberle, U., 2002. Das Nachhaltigkeitszeichen als Prozesslabel, in: Scherhorn, G.; Weber, C. (eds.): nachhaltiger Konsum. Auf dem Weg zur gesellschaftlichen Verankerung, Ökom Verlag, pp. 191-196

Eberle, U., 2011. Bestandsanalyse Nachhaltigkeitssiegel. Kurzgutachten im Auftrag der Bundestagsfraktion Bündnis 90 / Die Grünen. Berlin

Eberle, U., 2012. Auslobung klimarelevanter Aspekte bei Lebensmitteln. Verbraucherzentrale Schleswig-Holstein (Federführung), Verbraucherzentrale Niedersachsen, Verbraucherzentrale Hessen, Verbraucherzentrale Saarland, Verbraucherzentrale Nordrhein-Westfalen und Verbraucherzentrale Bayern (eds.)

Eberle, U., Spiller, A., Becker, T., Heißenhuber, A., Leonhäuser, I.U., Sundrum, A., 2011. Political strategy for food-labelling; Joint Statement of the Scientific Advisory Boards on 'Consumer and Food Policy' and 'Agricultural Policy' at the Federal German Ministry of Food, Agriculture and Consumer Protection (BMELV), Berlin

Grießhammer, R., Buchert, M., Gensch, C., Hochfeld, C., Manhart, A., Rüdenauer, I., 2007. Prosa – Product Sustainability Assessment.

Grießhammer, R., Hochfeld, C., 2009. Memorandum Product Carbon Footprint. Position statement on measurement and communication of the product carbon footprint for international standardization and harmonization purposes. Freiburg/Berlin, http://www.bmu.de/files/pdfs/allgemein/application/pdf/memorandum_pcf_en_bf.pdf

Hayn, D., Eberle, U., Rehaag, R., Simshäuser, U., Scholl, G., 2005. Konsument/innenperspektive. Ein integrativer Forschungsansatz für sozial-ökologische Ernährungsforschung. Discussion paper No. 8, www.ernaehrungswende.de

Krober-Riel, W., Weinberg, P., 2003. Konsumentenverhalten. 8th edition, Munich: Vahlen

Kyläkorpi, K., Rydgren, B., Ellegård, A., Miliander, S., Grusell, E., 2005. The Biotope Method. A method to assess the impact of land use on biodiversity. Vattenfall, Sweden

Lindner, J.P., 2008. Development of an indicator for biodiversity consideration in Life Cycle Assessment. Diplomarbeit, University of Stuttgart

Michelsen, O., 2008. Assessment of Land Use Impact on Biodiversity - Proposal of a new methodology exemplified with forestry operations in Norway. Int J LCA, pp. 22-31

Milà i Canals, L.; Bauer, C.; Depestele, J.; Dubreuil, A.; Freiermuth Knuchel, R.; Gaillard, G. et al., 2007. Key Elements in a Framework for Land Use Impact Assessment Within LCA. Int J LCA, pp. 5-15

Miller, G., 1956. The Magical Number Seven, plus or minus two: Some limits on our capacity for processing information; in: The Psychological Review, 63, pp. 81-97

Morrison, J., Schulte, P., Schenck, R., 2010. Corporate Water Accounting. An Analysis of Methods and Tools for Measuring Water Use and Its Impacts. United Nations Environment Programme, United Nations Global Compact, Pacific Institute.

Müller-Wenk, R., 1998. Land use — the main threat to species. How to Include Land Use in LCA. IWÖ Diskussionsbeitrag Nr. 64. Institut für Wirtschaft und Ökologie, Universität St.Gallen, St.Gallen, Switzerland

Pant, R., Van Hoof, G., Schowaneck, D., Feijtel, T.C.J., de Koning, A., Hauschild, M., Pennington, D.W., Olsen, S.I., Rosenbaum, R., 2004. Comparison between three different LCIA methods for aquatic eco-toxicity and a product environmental risk assessment. Insight from a detergent case study within OMNITOX. Int J LCA, pp. 295-306

Peter, J.P., Olson, J.C., Grunert, K.G., 1999. Consumer behaviour and marketing strategy: European Edition. London: McGraw Hill.

Schmidt-Bleek, F., 1997. Wieviel Umwelt braucht der Mensch?: Faktor 10 - das Maß für ökologisches Wirtschaften. dtv, Munich

Sengstschmid, H., Sprong, N., Schmid, O., Stockebrand, N., Stolz, H., Spiller, A., 2011. EU Ecolabel for food and feed products – feasibility study (ENV.C.1/ETU/2010/0025) http://ec.europa.eu/environment/ecolabel/about_ecolabel/pdf/Ecolabel%20for%20food%20final%20report.pdf

SRU (Sachverständigenrat für Umweltfragen - German Advisory Council on the Environment), 2002. Umweltgutachten 2002, Metzler-Poeschel, Stuttgart

Stop Climate Change (2008): Zertifizierungsprogramm zur Reduzierung von Treibhausgas-Emissionen. Standards zur Implementierung von Stop Climate Change. 10. Dezember 2008; S. 2; http://www.stop-climate-change.de/de/Standards.htm; Status: 15. October 2011

Van der Werf, H.M.G., Gaillard, G., Biard, Y., Koch, P., Basset-Mens, C., Gac. A., Lellahi, L., Deltour, L., 2010. Creation of a public LCA database of French agricultural raw products: agriBALYSE. Proceedings of the VII International Conference on Life Cycle Assessment in the agri-food sector, Bari, 22-24 September 2010, pp. 239-242

Verbraucherkommission Baden Württemberg (VK BaWü), 2011. Vom Labelmissbrauch zu Vertrauenslabeln (From misusage of labels to labels of confidence)

Walsh, G., 2002. Konsumentenverwirrtheit als Marketingherausforderung. DUV: Wiesbaden.

WHO/FAO (eds.), 2007. Codex Alimentarius, 5th edition, Rome

Communicating LCA results to the interested consumer - development of a criteria-based meat guide

Elin Röös

Department of Energy and Technology, Swedish University of Agricultural Sciences (SLU), Uppsala, Sweden, E-mail: *elin.roos@slu.se*

ABSTRACT

A low carbon footprint of meat is generally associated with low impacts across animal species in many environmental impact categories, but there is a risk of conflicts with categories such as biodiversity loss and pesticide use. Therefore, this project sought to develop a consumer guide that could assist Swedish consumers in making less environmentally harmful meat choices and that could also act as a communication tool, raising awareness of the different environmental aspects of meat production and potential conflicts with animal welfare. Four indicators (carbon footprint, biodiversity, use of pesticides and antibiotics and animal welfare) were chosen to represent the impact on the environment and animal welfare from different choices of meat and other protein sources. For each indicator, criteria were developed that placed the products included in the meat guide in one of three different groups, represented by the well-known traffic light system of red/yellow/green.

Keywords: animal welfare, biodiversity, carbon footprint, consumer guide, meat

1. Introduction

The carbon footprint (CF) of meat has received increasing attention in the quest for more sustainable eating habits. While a low CF of meat is generally associated with low impacts across animal species in many environmental impact categories, such as eutrophication, acidification and land use, there is a risk of conflicts with categories such as biodiversity loss and pesticide use (Röös et al., 2012). In addition, the focus on reducing emissions of greenhouse gases (GHG) risks decreasing animal welfare and increasing the use of antibiotics. This project sought to develop a consumer guide that could assist Swedish consumers in making less environmentally harmful meat choices and that could also act as a communication tool, raising awareness of the different environmental aspects of meat production and potential conflicts with animal welfare. The purpose of this paper was to describe how the guide was designed and discuss methodological difficulties and data limitations in conveying environmental information on generic meat production across many different production systems. The target groups for the guide were ‘the interested consumer’, i.e. those interested in accurately comparing the environmental impacts of different meat production systems, and employees working with food as their profession, e.g. buyers and personnel in the retail, restaurant and public procurement sectors. The products included in the guide were meat products based on certification schemes or other control programs with a high level of participation, in order to ensure that the criteria could be verified. Alternative sources of protein (game meat, eggs, cheese and vegetarian alternatives) that could replace livestock meat on the dinner plate were also included in the guide. However, fish was not included since its characteristics are quite different from those of arable and livestock production, and since several fish guides are already in use in Sweden.

2. Methods

2.1. Choice of indicators

One of the challenges was to find suitable indicators that accurately communicated the environmental impacts of meat production, while still being easy to understand. One option would have been to use an endpoint indicator that aggregated the results from many impact categories. This has been done in a meat guide for Swiss consumers, which uses the ‘potential species loss per year’ indicator from the ReCiPe-method (Blonk et al., 2010). However, such aggregation did not satisfy our requirements for a guide that also acted as a communication tool, raising awareness of the origins and causes of environmental impacts of meat production, the potential conflicts between different environmental goals and the implications for animal welfare. Therefore, CF was chosen as the first indicator, due to its familiarity and its ability to act as a proxy for eutrophication, acidification and land and energy use in most cases of meat production relevant for the guide, which was concluded in a study by Röös et al., (2012). The same study identified toxicity impacts (through the use of pesticides) and biodiversity as areas that risked coming into conflict with CF. In a previous study the possible conflict between CF and animal welfare had been highlighted (Röös, 2011). Thus, use of pesticides, biodiversity and animal welfare were chosen as the other indicators. In the review process, in which the guide was distributed to a large number of companies, NGOs and researchers, there was a general acceptance of the categories, but with the addition of the need to add information on the use of antibiotics, due to the concern about antibiotic resistance. Thus use of antibiotics was added to the same category as use of pesticides, as is further discussed in section 4.

2.2. Development of the criteria

For each indicator, criteria for three levels of environmental or animal welfare 'harm' were developed, judged as the level of complexity for the target group and corresponding to the well-known traffic light symbols used to communicate the levels. The criteria were developed on a relative scale, hence describing best and worst in class, rather than using absolute sustainability thresholds, which are very difficult to define on this detailed level. Data on CF, biodiversity impacts, use of antibiotics and pesticides and animal welfare (the chosen indicators, see section 2.1) for the products included in the guide were collected from the literature, trade associations and experts. From this information criteria were developed with the aim of differentiating between different types of production systems. The ambition in the meat guide was to develop the criteria for all indicators using a life cycle perspective related to the functional unit of 1 kg of product (bone-free meat in the case of meat). This proved to be difficult for some indicators due to methodological problems and lack of essential data. Therefore, in this first version of the meat guide it was necessary to take a pragmatic approach and develop the criteria considering current data availability and calculation methods, as well as certification schemes for verification (see sections 2.3-2.6).

2.3. Carbon footprint

The CF criteria were chosen to reflect the inherent variation in GHG emissions from the production of different types of protein sources based on biophysical differences in either 1) Directly consuming the vegetal protein, 2) feeding it to monogastric animals and losing a large part of the energy; or 3) producing meat from mainly cellulose using ruminants, which cause large methane emissions. Numerical results from numerous life cycle assessment (LCA) studies were used to set the boundaries (Röös, 2012; Röös et al., 2012). A green light was given to foods with CF less than 1.5 kg CO_2e/kg product, which includes most plant-based, protein-rich foods such as legumes. Game animals not given extra feed were also given a green light, based on the reasoning that wildlife methane emissions belong to the natural ecosystem and are not anthropogenic. This reasoning holds only as long as the amount of wild game is kept at natural levels. A yellow light was given to products that cause emissions of 1.5-12 kg CO_2e /kg product. These include eggs, chicken, pork and cheese. Products with CF greater than 12 kg CO_2e /kg product, i.e. meat such as beef and lamb from ruminant livestock, were given a red light. The variation in CF between studies can be large for the same product (± 50%) depending on differences in production systems, system boundaries and calculation methods. However, the CF for the products within the three levels still stayed within the wide boundaries of green, yellow and red light, making the need for more detailed data collection redundant for this indicator.

2.4. Biodiversity

Including impacts on biodiversity in LCA is challenging for many reasons. Biodiversity is a broad concept and includes diversity at gene, species and ecosystem level, so measuring this complexity for use in LCA is difficult. In addition, the impact on biodiversity from different types of land use varies considerably depending on the original habitat type, production intensity and surrounding landscape (Henle et al., 2008). Recent work based on the UNEP/SETAC life cycle initiative proposed a conceptual framework for land use in LCA (Milà i Canals et al., 2007). Building on this framework, others have proposed methodology for developing characterisation factors for biodiversity, as well as actual factors (Schmidt, 2008; de Baan et al., 2012). The characterisation factors describe the impact on biodiversity as a change in species richness when transforming a reference land use to different types of land use practices such as pasture, annual crops, agroforestry, etc.

Since the meat guide includes products from different countries, the impacts on biodiversity needed to be assessed on a scale that encompassed global effects, but was still detailed enough to enable differentiation between production systems. The ambition in the meat guide was to use global characterisation factors for biodiversity in order to assess the product's biodiversity impact per kg of product. Owing to time limitations, it was not possible in this first version of the meat guide to develop specific characterisation factors for Sweden and the factors found in the literature were either too blunt to be used in the meat guide (no differentiation between production systems, e.g. organic versus conventional production) or referred to baseline scenarios for land use that did not match the desired land use in Sweden from a biodiversity perspective. Using species richness as a proxy for land use also has several limitations, e.g. it does not consider red-listed or desirable species. Many of the endangered and red-listed species in Sweden are found in the traditional mosaic agricultural landscape that is disappearing due to agricultural intensification. Agriculture only represents 8% of the area in Sweden, planted forest being the dominant land use covering 53% of the area (SS, 2008). Hence, keeping traditional semi-natural pastures grazed and conserving the traditional mosaic landscape has been identified as one of the most important measures for preserving biodiversity in Sweden and in many other parts of Europe (Henle et al., 2008).

Therefore, the criteria for biodiversity in the meat guide were developed based on a qualitative assessment of the impact of different production systems on biodiversity. This assessment employed a combination of the concept developed by Geyer et al., (2010), which coupled species to habitat preferences, and the notion that the most important driver of biodiversity loss is land conversion from a natural state to human use (MEA, 2005). A green light was thus given to meat from production systems that help to conserve semi-natural pastures through grazing, thereby helping to preserve many red-listed species. Products requiring less than 5 m^2 of land use per kg (vegetal protein sources), were also given a green light, based on the concept of 'land saving' (avoiding the need for new agricultural land) (Green et al., 2005). A yellow light was given to organic production systems, based on literature showing higher biodiversity on organic farms (Bengtsson et al., 2005; Rahmann, 2011). A yellow light was also given to production systems that do not use imported soy as protein feed, owing to the risk of land use change in areas very rich in biodiversity due to expansion of soy cultivation. All other production systems were given a red light, since intensive agricultural production generally affects biodiversity negatively (Henle et al., 2008).

In future versions of the meat guide, the criteria for biodiversity impacts can be improved by developing characterisation factors for Sweden using the national biodiversity monitoring data. Taking into account red-listed species would be necessary in such an initiative to accurately reflect the value of grazing semi-natural pastures in comparison with grazing temporary leys. The development of such characterisation factors would enable biodiversity assessment on a per kg of product basis, hence including the aspect that different products require very different amounts of land (5-9 m^2 for chicken, 11-37 m^2 for pork and 24-244 m^2 for beef; Röös et al., 2012), which is omitted in this version of the meat guide.

2.5. Use of pesticides and antibiotics

Use of pesticides and antibiotics varies greatly between production systems, and in comparison with CF is not governed by inherent geophysical differences between animal species. Ruminants require more feed than monogastric animals per kg of meat produced, which could require more pesticides during feed production. However, the majority of the feed for ruminants is roughage, production of which usually uses very low amounts of pesticides. Hence, for this indicator it was not possible to base the criteria on general literature data from LCA studies on pesticide use in livestock production. Thus, the use of pesticides and antibiotics needed to be investigated in greater detail for the different production systems in order to find relevant boundaries. Pesticide use at farm level must be recorded by law in Sweden, as must the use of antibiotics in many certification schemes or control programs. However, such data are not related to the functional unit of 1 kg of meat, are located on individual farms and are not collected in a manner proving that 1 kg of product from a specific certification program included in the guide has given rise to a specific amount of pesticide or antibiotic use. In addition, for imported feed, meat and other food products in the guide this information is lacking. Hence, the criteria for this indicator needed to be based on national statistics on pesticide use (Eurostat, 2007) and use of antibiotics (EMA, 2011), and a rough approximation of typical feeding strategies for different production systems. As a starting point, a green light regarding use of pesticides was given to products originating from production systems that did not use pesticides at all, with organic farming representing the best in class. A red light was given to those using more than twice the amount of pesticides compared with the country with the lowest level (2 g active substance per kg of bone-free meat). The best in class level for use of antibiotics was set at a maximum of twice the lowest use found in the literature (2x14 mg/kg live weight), since was judged to be impossible and undesirable to have a zero level (sick animals need to be treated for animal welfare reasons). The worst in class level for the use of antibiotics was set to more than 10 times the lowest reported level (10x14 mg/kg live weight). To obtain a green light for this indicator as a whole, a green light was required for the use of both antibiotics and pesticides, while to obtain a yellow light one green and one yellow light in either the use of antibiotics and pesticides was required. All other combinations gave a red light.

2.6. Animal welfare

Designing animal welfare criteria is difficult since the subjective experience of the animal is difficult to quantify. The starting point for this indicator was the five freedoms for animal welfare originally developed in a UK report on livestock husbandry in 1965 (FAWC, 2012) and now widely used as a basis for animal welfare regulations by a number of organisations. These are: 1) Freedom from hunger and thirst, 2) freedom from discomfort, 3) freedom from pain, injury or disease, 4) freedom to express normal behaviour and 5) freedom from fear and distress. The judgement as to how these freedoms were met by different production systems was made in collaboration with animal welfare experts. To be awarded a green light, the livestock had to be kept according to Swedish animal welfare legislation (or similar legislation), which explicitly requires the first four freedoms. In addition, to receive a green light there was a requirement on outdoor pasture, which gives extra weight to freedom (4) and hopefully also freedom (5), which is impossible to measure. This judgment is based on the assumption that animals can better express their natural behaviour in outdoor environments with access to pasture, where their strong urge for food searching can be satisfied. Enriched indoor environments can fulfil the same function, but today there are few such systems, and no verification or certification that ensures that systems provide a certain level of opportunities for natural behaviour and stimuli. To be given a yellow light the product had to come from systems that are either covered by Swedish legislation on animal welfare or the like or from animals out grazing at least half the year. All other systems were given a red light. The criteria for this indicator were not related to the functional unit of 1 kg of product, but were based on production system level, and did not take into account the number of individual animals affected by the production of 1 kg of meat.

3. Results

The beef part of the current guide is shown in Figure 1. Although the guide contains many simplifications it still provides considerable amounts of information that allow a deeper understanding of the underlying origins and causes of environmental impacts from livestock production, to which the target audience, food professionals and interested consumers, are presumably receptive. For example, it can be seen that animal welfare-friendly, pasture-based beef production, which keeps pastures open and helps conserve their rich biodiversity and minimises the use of pesticides and antibiotics, is also associated with high emissions of greenhouse gases.

4. Discussion

All simplifications involve difficulties in conveying a complete and fair picture. The meat guide presented here has several limitations. All indicators are given equal weight, although the CF indicator acts as a proxy for several other impact categories. For the biodiversity and animal welfare indicators, the criteria are not developed from a life cycle perspective and do not relate to the functional unit of 1 kg of product. The animal welfare criteria are set on production system level and do not consider the number of individual animals affected in order to supply a certain amount of meat. Systems which account for this could be designed by e.g. developing an equivalent to DALY (disability adjusted life years) for animals, as discussed by Blonk et al., (2010). Assessing biodiversity per kg of product could be made feasible through the development of characterisation factors applicable for Swedish conditions.

	Carbon foot-print	Biodiversity	Pesticides and antibiot-ics	Animal welfare
Beef from semi-natural pastures, organic	☹	🙂	🙂	🙂
Beef from semi-natural pastures	☹	🙂	😐	🙂
Organic beef, KRAV label	☹	😐	🙂	🙂
Organic beef, EU label	☹	😐	🙂	😐
Swedish beef, Svensk Sigill label	☹	😐	😐	😐
Swedish generic beef	☹	☹	😐	😐
Irish generic beef	☹	☹	😐	😐
Polish and German generic beef	☹	☹	☹	☹
South-American generic beef	☹	☹	😐	😐

Figure 1. The beef section of the Swedish meat guide

It might seem strange to include legumes, as well as eggs and cheese, in what is called a meat guide. The reason for including these alternatives was partly to avoid sub-optimisations from exchanging meat for cheese, a product with considerable environmental impact. Another reason was to show and discuss the possibility of exchanging some meat consumption for plant-based protein sources, which inherently has considerably lower impacts and no associated animal welfare issues. Many studies show that in future sustainable food supply systems, meat consumption in the developed world needs to decrease considerably (see e.g. Foley et al., 2011), which is why we believed it to be important to also show the environmental impacts of other protein sources.

Another area of possible criticism concerning the design of the meat guide is the inclusion of pesticide use and use of antibiotics in the same category. This was done due to the strong desire to keep the number of indicators to a maximum of four in order to limit the complexity of the guide. Although the use of pesticides and antibiotics is unrelated, they do have some common features. They are both chemical inputs to production systems that might leak into the environment and affect ecosystems and species, and the use of both can be limited by taking precautionary measures (well-designed crop rotations in the case of pesticides and preventive health programs in the case of antibiotics). In this first version of the guide, use of pesticides and antibiotics are presented as LCI results rather than LCIA results. In later versions this could be improved by using LCIA models to capture e.g. variations in toxicity effects between different pesticides.

The design of the criteria for the different indicators and the choice of the indicators involved subjective judgements. This is unavoidable when condensing a large amount of scientific literature down to consumer communications. However, many experts and representatives from trade organisations, authorities and NGOs were involved in development of the criteria and the indicators chosen were as robust and well-conceived as possible. All underlying assumptions are openly presented in the meat guide, so although there might be different opinions as to how the criteria should be developed and how different production systems should be valued, the guide could function as a basis for discussion and raise awareness of the issues related to livestock production. This was the primary objective with the meat guide as well as making an attempt to make LCA results on the food product category with the greatest environmental impact (meat) accessible to the wider public.

5. Conclusions

Basing choice of meat on an environmental impact and animal welfare perspective is complex and cannot be rely solely on the carbon footprint, as this can lead to goal conflicts. The meat guide described here attempts to condense and simplify the scientific literature in order to facilitate active choices by the 'interested consumer' and food professionals, while still capturing and explaining the complexity in comparing different production systems. This first attempt to develop a meat guide for the Swedish market has several limitations, but should provide guidance and can act as a basis for discussion in the important task of decreasing meat consumption and choosing better meat alternatives.

6. References

Bengtsson, J., Ahnström, J. and Weibull, A-C. 2005. The effects of organic agriculture on biodiversity and abundance: a meta-analysis. Ecology 42, 261-269.

Blonk, H., Marinussen, M., Goedkoop, M. 2010. Developing an LCA based consumer guide for environmental and animal welfare performance of meat/animal products. Proceedings of LCA Food 1, 381-386.

de Baan, L., Alkemade, R., Koeller, T. 2012. Land use impacts on biodiversity in LCA: a global approach. Int J of Life Cycle Assess. Published online: 24 April 2012.

EMA (European Medicine Agency). 2011. Trends in the sales of veterinary antimicrobial agents in nine European countries. Reporting period: 2005-2009.

Eurostat. 2007. The use of plant protection products in the European production. Eurostat Statistical Books. 2007 edition.

FAWC. 2012. Five freedoms. Farm Animal Welfare Council. London, UK. http://www.fawc.org.uk/freedoms.htm

Foley, J., Ramankutty, N., Brauman, K., Cassidy, E., Gerber, J., Johnston, M., Mueller, N., O'Connell, C., Ray, D., West, P., Balzer, C., Bennett, E., Carpenter, S., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D. 2011. Solutions for a cultivated planet. Nature 478, 337-342.

Geyer, R., Stoms, D., Lindner, J., Davis, F, Wittstock, B. 2010. Coupling GIS and LCA for biodiversity assessments of land use. Int J Life Cycle Assess 15, 692-703.

Green, R., Cornell, S., Scharlemann, J., Balmford, A. 2005. Farming and the fate of wild nature. Science 307, 550-555.

Henle, K., Alard, D., Clitherow, J., Cobb, P., Firbank, L., Kull, T., McCracken, D., Moritz, R., Niemelä, J., Rebane, M., Wascher, D., Watt, A., Young, J. 2008. Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe – A review. Agriculture, Ecosystems and Environment 124, 60-71.

MEA (Millennium Ecosystem Assessment). 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

Milá i Canals, L., Bauerm C., Depestele, J., Dubeuil, A., Knuchel, R.F., Gaillard, G, et al., 2007. Key elements in a framework for land use impact assessment within LCA. International Journal of Life Cycle Assessment 12(1), 5-15.

Rahmann, G. 2011. Biodiversity and organic farming: What do we know? Landbauforschung Volkenrode
61 (3), 189-208.

Röös, E. 2011. The trade-off between animal welfare and lowered emissions of greenhouse gases from pork production. Conference paper, 24th Nordic Association of Agricultural Scientists (NJF) Conference, Uppsala, Sweden. Juni 14-16, 2011.

Röös, E. 2012. Mat-klimat-listan. Report 040. Department of Energy and Technology, Swedish University of Agricultural Sciences.

Röös, E., Sundberg, C., Tidåker, P., Strid, I., Hansson, P-A. 2012. Can carbon footprint serve as an indicator of the environmental impact of meat production? Ecological Indicators, accepted for publication 120807.

Schmidt, J. 2008. Development of LCIA characterisation factors for land use impacts on biodiversity. Journal of Cleaner Production 16, 1929-1942.

SS (Statistics Sweden). 2008. Land use in Sweden. Fifth edition.

Life cycle impacts of protein-rich foods for consumer information. How LCA results can be presented to consumers to make environmentally sustainable supermarket choices

Marieke Head, Maartje Sevenster, Bart Krutwagen*, Harry Croezen

CE Delft, Delft, The Netherlands

* Corresponding author. E-mail: *krutwagen@ce.nl*

ABSTRACT

CE Delft has calculated the life cycle environmental impacts of 98 different animal products and animal product alternatives from the farm to the supermarket. The goal of this study was to provide the consumer with scientifically sound environmental data for several protein-rich products. The results of this expansive life cycle assessment have been used in a mobile application. This consumer support tool will allow Dutch consumers to scan product barcodes and obtain relevant environmental impact information on their mobile phones. The impact assessment for the products was carried out using a customised version of the ReCiPe (hierarchic) method on both midpoint and endpoint levels. The results from the analysis show substantial differences between the lowest and the highest scoring products, particularly in terms of the effects on nature & environment. However, the most pronounced trend is that the difference in environmental impact within product groups can be just as great as the environmental impacts between product groups.

Keywords: LCA, agriculture, animal products, app, consumer, variation in impact

1. Introduction

Developed by the Varkens in Nood foundation (Pigs in Need), the *Vleeswijzer* (Meat Index) was launched at the end of 2009 and offers consumers information about the environmental and animal welfare impacts of the most common meat and meat alternative products. The *Vleeswijzer* was giving however only general impact data that did not distinguish between products within a product group. With the large variety of protein products within one productgroup available in the supermarkets, such as conventional, organic, free-range options, it is often difficult for the consumer to assess which products are the more sustainable ones.

To overcome this problem CE Delft was asked to make a follow up, the *Superwijzer*; the end goal of this study was to update and expand the *Vleeswijzer* such that the consumer is able to make informed purchasing decisions amongst various meat, dairy, eggs, and alternative products, in terms of the environmental and animal welfare performance of those products.

CE Delft’s contribution (Head et al., 2011) was determining the climate change, biodiversity and human health impact of meat, meat alternatives, dairy and additional product types. In comparison to the assessment done for the *Vleeswijzer*, the list of product types was expanded and the most recent scientific developments were included in the product assessments. The environmental data for this app are derived from the thorough study of 98 protein-rich product types available in the Dutch supermarkets, within 10 different product categories. Such a wide variety of products were examined in order to not only compare between product groups but also within product groups. It also made assessment of less obvious products like coffee creamer and whipped cream possible.

For this information to be accessible and convenient for the consumer, Varkens in Nood created a smartphone App. This App allows the consumer to have an interactive version of the *Superwijzer,* allowing them to have product information available while shopping. Simply by scanning the barcode of the products.

2. Methods

As stated above, 98 protein-rich product types within 10 categories were assessed (Head et al., 2011). The results of this assessment were used to score over 15,000 products available in Dutch supermarkets. Relevant issues concerning the extend of the life cycle, allocation and cut-off and included impact categories are discussed in the following sections.

2.1 Extent of the life cycle

The life cycles of all the products are modelled up to the point of retail (including cooling). Although the products are diverse, there is much overlap between the life cycles and therefore the system boundaries of each life cycle can be summarised by a simplified diagram (see Figure 10).

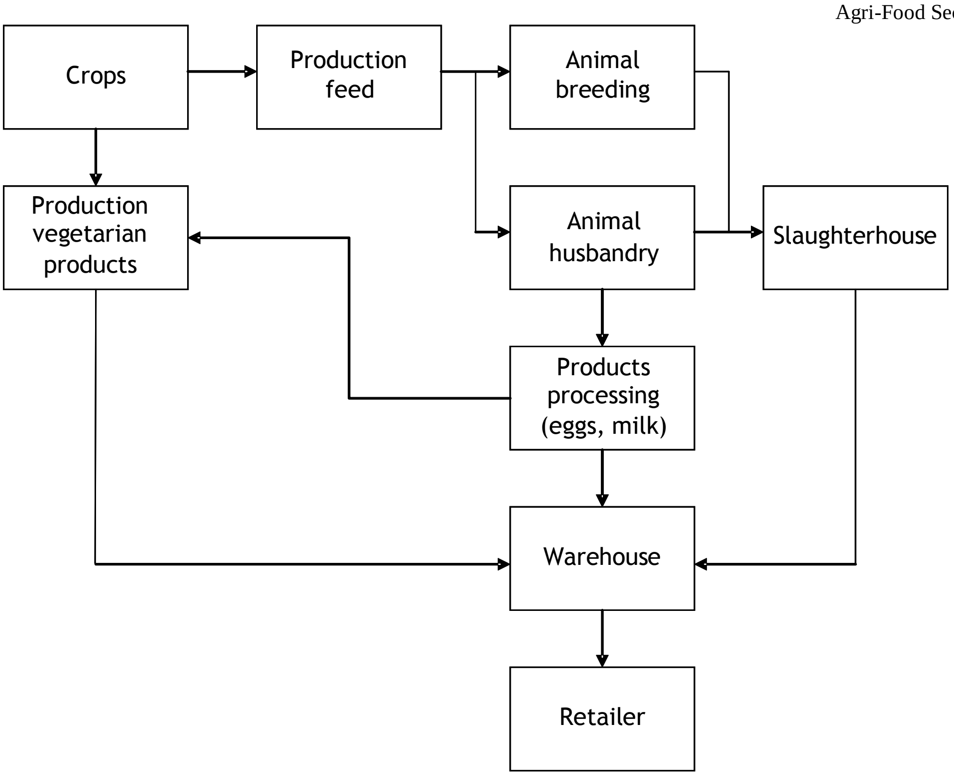


Figure 10. General overview of processes that are included in the product life cycles

2.2. Allocation and cut-off

Agriculture, particularly animal agriculture, can become rather complex to model accurately. There are many reasons for this complexity, much having to do with the fact that farms can be very different and one often has to rely on average systems. However, some of the complexity is derived from the multiple outputs from processes and the multiple usages of these outputs, which can make the product focused goals of a life cycle assessment a daunting task. In order to solve this problem, various methods were employed:

Applying a cut-off for manure: Emissions from management, as well as paddock manure, are included to the animal system, but emissions from later applications are allocated to the crop system or energy system involved.

The allocation of crop products: Crop types can often be processed from their constituent parts (proteins, oils, fibres, etc.) into multiple products, thus an allocation approach based on economic value was applied.

The allocation of animal products: Economic allocation was used to determine the relative share of the burden that such co-products have at the farm stage and at the slaughter stage.

The allocation of raw milk to dairy products: According to the foot printing methodology of the International Dairy Federation (IDF, 2010), raw milk is allocated to products of dairy processing via milk solids content.

2.3. Data and model

The basis for the *Superwijzer* is the study published by Blonk et al., (2008), the main basis of the *Vleeswijzer*. Given the complexity of the product life cycles and the large range in geographic coverage needed, a multitude of sources were used in the models to supply data concerning animal feed, vegetarian products, land use and land transformation emissions, animal emissions, farming systems, slaughter, processing, transport, distribution, storage and retail. While efforts were made to include all relevant aspects of the life cycles of the various products, not everything could be included. The effects of these exclusions on the results are most likely not significant; results were compared with existing results in literature, and the differences can be explained by assumptions and methodological choices. A full description on included and excluded processes can be found in (Head et al., 2011).

2.4. Impact categories

A customised version of ReCiPe (hierarchic endpoint) method is used in the assessment of the environmental impact of the various product types. The impact categories of the ReCiPe method have been clustered into four main categories, in order to make them more understandable for consumers:

- Biodiversity: The effects of environmental damage on biodiversity and ecosystems, measured in species.yr or PDF (potentially disappearing fraction). These include the impact on the biodiversity in ecosystems due to: terrestrial acidification, freshwater eutrophication, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, effects of climate change on ecosystems, agricultural land occupation, urban land occupation, natural land transformation.
- Human Health: The effects of environmental damage on human health are measured in DALY (disability-adjusted life years) and are measured as endpoints. The following midpoint impact categories are included: ozone depletion, human toxicity, photochemical oxidant formation, particulate matter formation, ionising radiation;, effects of climate change on human health.
- Climate Change: Climate change is measured in terms of kg $CO_{2\text{-eq}}$ and includes the following categories: climate change (process), climate change, land transformation.
- Land Use: Land use, measured in m^2, takes into consideration the physical space that is occupied by a given system. It includes the follow categories: agricultural land occupation, urban land occupation. This category was not used in the *Superwijzer*, but was used as a separate indicator during the research.

In the *Superwijzer* App the impact of the product is presented in four categories that mainly overlap the categories mentioned above: Biodiversity, Climate change, Harmful substances (combination of the Human health assessment by CE Delft and an internal review by Pigs in Need concerning e.g. use of antibiotics) and Animal welfare (assessed by Science for Society). The App-user is free to weight the categories according to their own interest (a default is automatically included).

3. Results

The results show that there are large differences between the product groups, but also within product groups. The differences in livestock management, feed, feed conversion and greenhouse gas production by ruminants are the main causes for the differences in the categories biodiversity, climate change and land use impacts. Human health impacts differ, relative to the volatisation of N-compounds from fertilisers, the amount of stable emissions occurring and the emissions from transport.

The most significant results of the environmental assessment of the 98 products are shown in

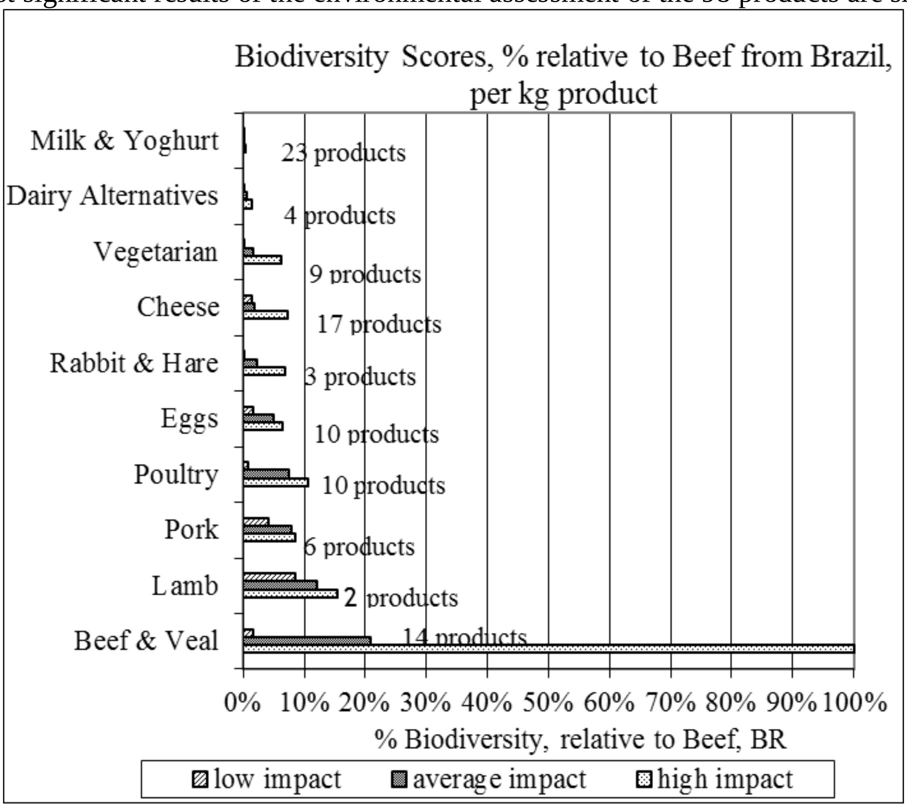


Figure 11 and Figure 12, for biodiversity and climate change, respectively. These are the two categories directly used by Pigs in Need, and show the largest differences between groups and within groups or product. The bars indicate the environmental impact or damage (larger bars indicate that products are worse for the environment), relative to the product with the highest score. For biodiversity, climate change and human health the product with the highest score is Brazilian beef, for land use it is Argentinian beef.

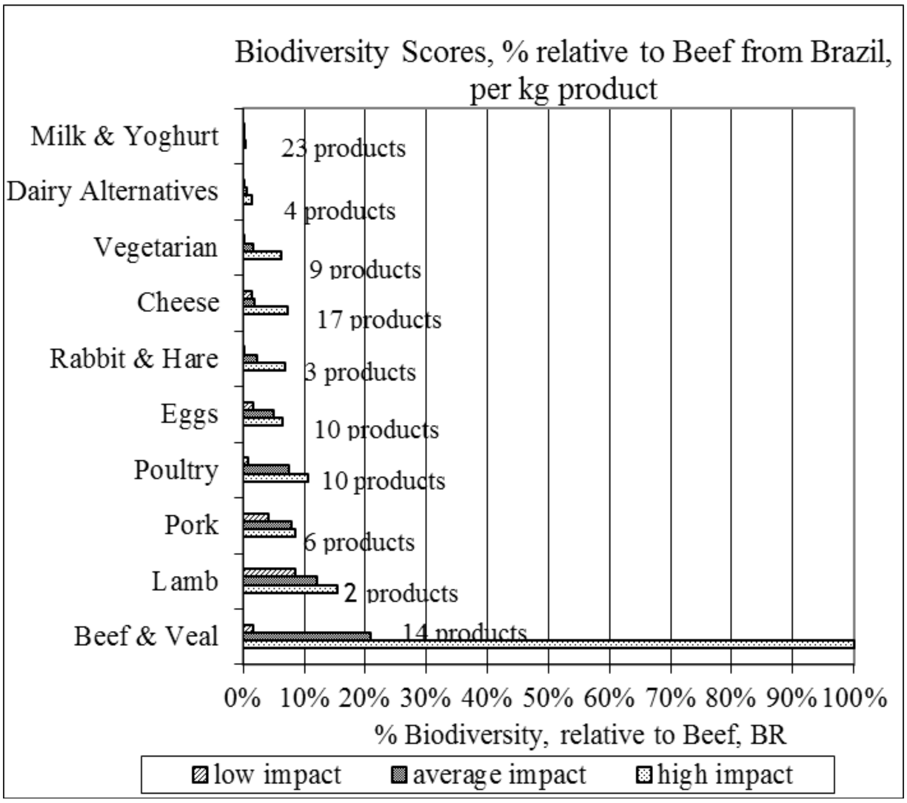


Figure 11 Categorised scores for impact on biodiversity. The bars indicate, from top to bottom, the lowest, the average and the highest score in each category. The number of products represented in each category is given beside the scores.

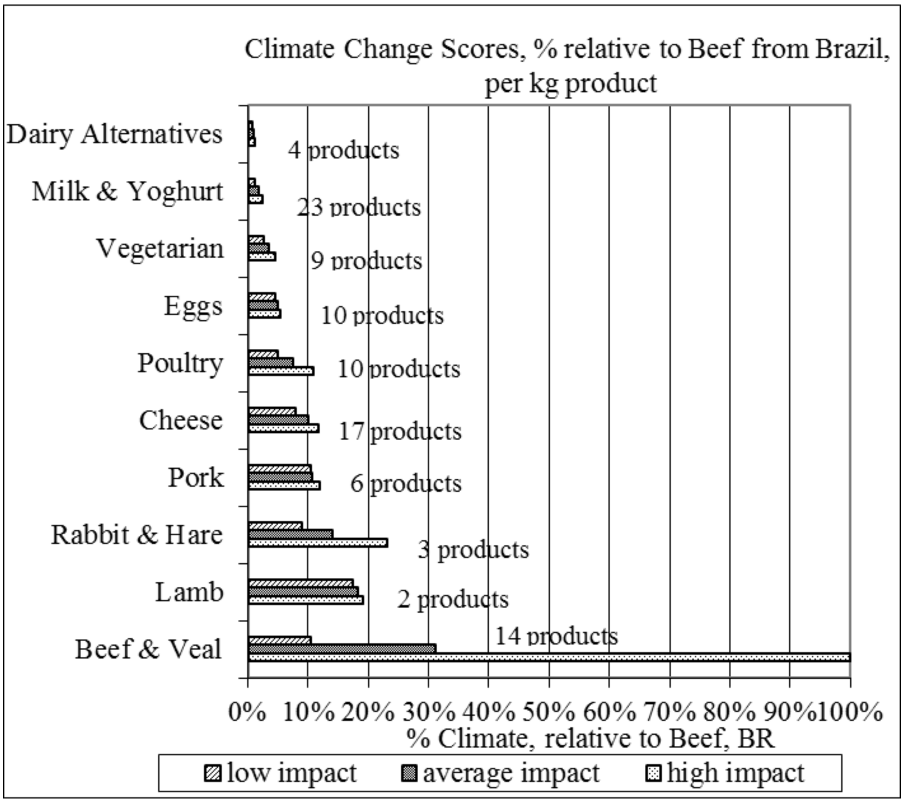


Figure 12 Categorised scores for impact on climate change. The bars indicate, from top to bottom, the lowest, the average and the highest score in each category The number of products represented in each category is given beside the scores.

4. Discussion

The results from the *Superwijzer* show substantial differences between the lowest and the highest scoring products, particularly in terms of the effects on biodiversity. At the extreme, the highest scoring product (Brazilian beef) has a biodiversity score of over 3,000 times that of the lowest product (Dutch hare). Also within the product group ‘Beef and Veal’ there is still a difference of factor 10 between the lowest and highest score. In terms of the effects on climate change, Brazilian beef has a score that is about 36 times higher than Quorn, a meat alternative. Although these scores only illustrate the upper and lower scores, there is a distinct clustering of product types. In terms of an approximate product ranking, beef and veal rank worst, followed by other meat types (the order depends on the impact category used), followed by eggs and cheese, and finally, the meat substitutes (vegetarian), milk and yoghurt and dairy alternatives rank best.

There are also distinct variations in certain product categories. Beef and veal have by far the largest range in scores both for biodiversity and climate change. The lowest scoring products are minced and cut beef originating from spent dairy cows, while the highest scoring product is Brazilian beef. Another product category with a large variation is ‘rabbit and hare’, which has a relatively low scores for Dutch hare and relatively high score for rabbit. Some product groups, such as pork have very little variation in the environmental impact within the group.

As shown in the results section, the differences are large between product groups, but also within product groups, especially the beef and veal group. There are several important factors resulting in the range in this group: e.g. the feed used, allocation assumptions and slaughter-age. Higher consumption of (soy) concentrate by livestock result in higher score, specifically for biodiversity. The lower-scoring beef products are mince beef products from spent dairy cows; the environmental impact is allocated on an economic basis, thus only 5.5% of the environmental impacts are allocated to the beef. Veal scores high because of relatively short live times and relatively high impact of feed due to their special feed mix.

Based on these results we can state that consumers have much more choices in greening their diets then becoming a vegetarian or restricting their diet to less meat. Even a choice between product groups is not necessary, as long as the products with the lowest impact within each product group is selected.

5. Conclusion

This study provides an interesting example of how LCA results can be made available to consumers, by providing easy to understand information in a simple way to help them to make environmentally conscious choices in their diet. The large variations in environmental impact within product groups can have an effect on the ranking of a particular product group, such that general statements regarding the scores of specific groups are difficult to make, as is illustrated by the wide range of scores of beef products. Because of the large variation which exists both within product groups and between product groups, choosing products with a low environmental impact will lead to significant reductions in the environmental impact of an individual’s diet. Therefore, it is important to assess a range of food products, as is done in the *Superwijzer* App in which 15,000 different products are rated, to provide the best available information to the consumer at the location where choices are made; in the supermarket.

6. References

Blonk, H., Kool, A. and Luske, B. (2008). Milieueffecten van Nederlandse consumptie van eiwitrijke producten: Gevolgen van vervanging van dierlijke eiwitten anno 2008, Blonk Milieuadvies, 2008

Broekema, R., Blonk H., with Alverado, C. and Hegger, S. (2009). Milieukundige vergelijkingen van vleesvervangers, Blonk Milieuadvies, 2009

Cederberg, C., Persson, U.M., Neovius, K., Molander, S. and Clift, R. (2011). Including Carbon Emissions from Deforestation in the Carbon Footprint of Brazilian Beef. Environmental Science and Technology, vol. 45, no.5 (2011); p. 1773-1779.

Head, M., Sevenster, M. and Croezen, H. (2011). Life Cycle Impacts of Protein- rich Foods for Superwijzer. Report 11.2329.57, CE Delft.

IDF (2010), A common carbon footprint approach for dairy: the IDF guide to standard lifecycle assessment methodology for the dairy sector, In: Bulletin of the International Dairy Federation no.445, 2010

Environmental impact of different broiler production systems in Malaysia and consumer willingness to pay for reduced impact

Noraisah Spahat*, Jonathan H. Guy, Ilias Kyriazakis, Guy Garrod, Sandra A. Edwards

School of Agriculture, Food and Rural Development, Newcastle University, Newcastle upon Tyne, NE1 7RU UK
*Corresponding author. E-mail: *noraisah.spahat@newcastle.ac.uk*

ABSTRACT

The objectives of this study were to i) estimate the environmental impact of different broiler production systems in Malaysia using a Life Cycle Assessment, with a functional unit of one tonne live-weight of broiler chickens and by taking a cradle to farm-gate approach; and ii) estimate consumers' willingness to pay (WTP) for chicken meat produced with a higher regard for the environment, with consideration of socio-economic characteristics, using a Contingent Valuation Method. The environmental impact analysis using data for two intensive systems and one semi-intensive system showed clear differences in impact between the three systems, with the intensive closed-house system being the least environmentally impacting. The environmental awareness of respondents resulted in a stated WTP value for reduced environmental impact which was significantly higher than the average sale price of chicken meat at both national and regional levels, with half of respondents willing to pay an increment of 10% above the existing market price.

Keywords: broiler chicken, environmental impact, life cycle assessment, willingness to pay, contingent valuation method

1. Introduction

The poultry industry has been the fastest growing of all livestock sectors in both developed and developing countries (Steinfeld et al., 2006; Narrod et al., 2008). It is characterised by a highly dynamic market, consolidated yet still expanding, even though it constantly faces price fluctuations of raw materials and public health concerns. According to FAO (2012), world poultry meat production in 2010 was 98 million tonnes and is expected to reach 122.5 million tonnes by 2020 (Best, 2011). This growth in production is not only due to demand factors, such as increase in population, disposable income and urbanisation, which lead to a changing pattern of consumption, but also due to supply side factors, especially the implementation of large scale vertically and horizontally integrated production chains. This type of production is typically focused on intensive systems which are able to absorb any great fluctuations in input prices, reducing transaction costs and giving control over product quality and safety, besides having a significant impact on overall production performance (Narrod et al., 2008).

Malaysia is self-sufficient in meeting the domestic demands for poultry meat, with a sufficiency of 122% in 2009, representing an annual growth rate of 5.3% for the period 2005-2009. Poultry contributed 53% of the overall value of livestock products of Malaysia in 2009 (DVS, 2011). Most chicken meat is consumed fresh and only 10% is used by the processing industries to manufacture products such as nuggets, burgers and other value-added products, especially for the fast food businesses. The demand for poultry meat from the downstream industries has grown rapidly and generated an increase in demand from the domestic and international market. To achieve the demand target, the expansion of production through vertically and horizontally integrated approaches has been identified as an effective solution. There are many economic advantages of intensive broiler chicken production, particularly the closed-house system that is popular in many countries with a developed broiler industry. However, intensive production might be taxing to the environment because of the release of higher levels of unavoidable waste. Therefore, development of more environmentally-friendly broiler production without compromising economic expectations is crucial. The objectives of this study were to i) estimate the environmental impacts of different broiler production systems in Malaysia; and ii) estimate consumer willingness to pay (WTP) for chicken produced with a higher regard for the environment (chicken-HRE).

Life Cycle Assessment (LCA) is an ISO standard procedure of environmental accounting, a framework which compiles and evaluates the use of resources (i.e. energy inputs, processing of raw materials, disposal and emission of pollutants) for each stage of a production cycle, and interprets the environmental burdens arising from this product system to the specific impact categories (de Vries and de Boer, 2010). Two types of impact categories are available according to their position in the environmental relationship between emissions and impacts, namely midpoint and endpoint indicators. The midpoint indicators are parameters in a cause-effect chain for a particular impact category, whilst endpoint indicators reflect the final effect of impact category and thus often have a higher relevance to society.

The second objective was to evaluate consumer WTP for chicken-HRE, as a proxy to evaluate non-exclusive goods, namely the favourable environmental quality which is intangible and does not have a market price. Based on a utilitarian approach, which refers to the sum of the utility of a certain good to society, a stated preference method was used to estimate the marginal benefits utility, i.e. a contingent valuation method (CVM) (Mitchell and Carson, 2005). In the current study, some improvement in elicitation format was applied to reduce the potential for bias in responses which is always a possible criticism of CVM. Many environmental economists agree that CVM is a highly developed survey approach for non-market valuation, using trained interviewers and involving extensive use of visual aids such as maps, photographs and charts to acquire the closed estimation of WTP and mirror real customer behaviour in the actual market (Hanley et al., 2001; Mitchell and Carson, 2005). In combination, the LCA and CVM approaches may be useful to determine the strategic direction for the Malaysian poultry industry to ensure the effective distribution of financial allocation to support development of food production that is both economically and environmentally sustainable.

2. Methods

Life Cycle Assessment: The assessment of environmental burdens from a total system involved in the life cycle of a product were taken into account, from resources extraction through the processing of raw materials (i.e. crop cultivation and the mechanisation used, feed and feed additives manufactured), utilities consumption (i.e. transportation, water, gas and electricity) and waste disposal (manure, gaseous emissions and substances derived from manure) as well as manure as a fertiliser credit. In this study, poultry manure from the intensive systems could be used as plant nutrients to offset the synthetic fertiliser that would otherwise be required for growing crops. LCA was employed through the use of specialist software, namely SimaPro (version 7.3.2, PRé Consultants, Amersfoort, Netherlands) to evaluate the environmental impacts of three different broiler production systems, namely an intensive closed-house system (CH; typified by confinement of birds indoors in relatively large flocks with environmental control), an intensive open-house system (OH; as for CH but without a controlled environment due to the open sides of the building) and a semi-intensive system (SI; unlike the previous two systems, it uses unimproved genotypes, very small flock size, simple housing and provides access to range). A functional unit (FU) of one tonne live weight of broiler chickens was chosen, with cradle to farm-gate as a system boundary and focusing on a single product output (the model was such that no allocations were necessary) with an attributional approach, since the motivation of the study was based on the technologies provided in different production systems. Four midpoint impact categories were considered, namely energy use, global warming, acidification and eutrophication which contain multiple types of burdens. Most burdens are aggregated into potential for causing impacts using a specific characterisation factor, i.e. Global Warming Potential (GWP), Acidification Potential (AP) and Eutrophication Potential (EP). For example, the characterisation factor of CO_2 is 1, and the GWP value resulting from 1 kg of CH_4 and N_2O are 25 and 298 times respectively higher than CO_2. Time constraints meant that the potential impact of changes in land use was not considered in the current study. Foreground data from nine broiler farms, three farms for each of the three different production systems, and one breeder farm were obtained from sources in the poultry industry and the Government of Malaysia. In addition, background data from the Ecoinvent database (Ecoinvent Centre, Dübendorf, Switzerland), covering the processes and services of products were applied.

Contingent Valuation Method: The structure and essential components of most Contingent Valuation (CV) questionnaires consist of five steps, namely i) setting up the valuation scenario; ii) introducing the payment vehicle; iii) eliciting valuation; iv) validation; and v) collecting socio-demographic and economic characteristics. The valuation scenario contains the critical part of the CV, namely the establishment of the hypothetical scenario which defines and creates the justification to value the good; a process of describing the potential consequences from the changes of the good and the institutional setting which provides the information on the implementation, delivery and monitoring actions to the public. A plausible mixed payment vehicle of a voluntary nature based on a price increase was used in the survey (Pearce et al., 2002). A series of validation follow-up questions were asked in order to understand the motives for the answer given and to help identify any potential bias. The final element of the questionnaire asked for socio-demographic and economic information of the respondents, which was used to confirm the theoretical expectation towards WTP (Mitchell and Carson, 2005). The face-to-face survey was employed in four regions with a total of two hundred and ten respondents recruited, comprising 27, 93, 60 and 30 persons respectively from the northern, central, southern and eastern regions of Peninsular Malaysia. The respondents could be anyone as long as they consumed chicken in their diet and were aged between 21 and 56 years of age. Statistical differences in WTP value between respondents from different regions were explored using appropriate statistical tests including analysis of variance (ANOVA). Various association and correlation tests were performed to identify key socio-demographic parameters for cause and effect relationship tests for WTP option (yes/no) and WTP value.

3. Results

3.1. Environmental impact of different broiler production systems

Table 1. Life cycle impact assessment of energy use, global warming potential, acidification potential and eutrophication potential associated with the production of one tonne live weight of broiler chickens from three different Malaysian housing systems. For the impact of each housing system, the values in a row show the absolute amount followed by, in italics, the percentage of the total impact accounted for by that particular activity

	Broiler feed-related		**Breeder feed-related**		**On-farm inputs/ emission**		**Fertiliser Credit**	**Total**
Energy use (MJ)								
Closed house system	8,946	*74.2%*	828	*6.9%*	2,280	*18.9%*	-3,679	**8,375**
Open house system	10,417	*83.42%*	1,104	*8.9%*	908	*7.3%*	-3,945	**8,484**
Semi-intensive system	12,892	*84.9%*	1,065	*7.0%*	1,236	*8.1%*	-4,807	**10,386**
Global Warming Potential (kg CO_2 eq. on a 100 year timescale)								
Closed house system	1,603	*82.7%*	142	*7.3%*	192	*9.9%*	-680	**1,257**
Open house system	1,866	*86.7%*	190	*8.8%*	95	*4.4%*	-784	**1,367**
Semi-intensive system	2,309	*87.6%*	179	*6.8%*	148	*5.6%*	-889	**1,747**
Acidification Potential (kg SO_2 eq.)								
Closed house system	12.8	*67.8%*	1.2	*6.5%*	4.9	*25.7%*	-2.0	**16.8**
Open house system	14.9	*68.5%*	1.6	*7.5%*	5.2	*24.0%*	-2.3	**19.4**
Semi-intensive system	18.4	*70.1%*	1.4	*5.2%*	6.5	*24.6%*	-2.7	**23.6**
Eutrophication Potential (kg PO_4 eq.)								
Closed house system	7.9	*76.0%*	0.7	*7.0%*	1.8	*17.0%*	-0.3	**10.0**
Open house system	9.2	*76.1%*	1.0	*8.0%*	1.9	*15.9%*	-0.4	**11.7**
Semi-intensive system	11.4	*78.6%*	0.7	*5.0%*	2.4	*16.4%*	-0.4	**14.1**

After considering the offset values from manure as fertiliser, the SI system recorded the largest burdens in the categories of Energy Use, GWP, AP and EP with 10,386 MJ, 1,747 kg of CO_2 eq., 24 kg of SO_2 eq. and 14 kg of PO_4 eq. respectively (see Table 1). This was followed by the OH system, with the CH system being the least environmentally impacting, producing approximately 14 to 28% lower impacts than the SI system.

Broiler feed-related inputs accounted for approximately three quarters of the impact values (with 67.8 to 82.7% for the CH system, 68.5 to 86.7% for the OH system and 70.1 to 87.6% for the SI system), followed by other on-farm inputs and emissions from the manure and chickens, while breeder feed-related inputs contributed the smallest environmental impact. Breeder feed-related inputs contributed approximately 5 to 9% to environmental impacts, with the OH system producing the highest impacts compared to other systems, since this system required four breeder hens to produce one FU. The CH system required an energy input of approximately four times that of the SI and OH systems, largely due to heating and ventilation requirements. However, the SI system produced much higher burdens from emissions, especially the categories of acidification and eutrophication potentials, mainly due to the large amount of manure generated throughout the extended production cycle of 12 weeks.

3.2. Estimation of consumers' WTP for chicken-HRE

For the first objective of the CVM survey, respondents were asked to evaluate the relative environmental impact resulting from the six major economic activities of Malaysia. On average 48.1% of respondents considered the manufacturing sector as being the major contributor of negative impacts on environmental quality, followed by the chemical industry (31.9%), transportation (10.5%), agriculture (5.7%), mining (2.9%) and construction (1.4%). Almost two-thirds of respondents (63.3%) stated that the efforts taken by the both the Government and the industry in these six economic activities to prevent environmental degradation were less than their rate of development. Meanwhile, half of respondents stated that quality was the main factor influencing their decision when purchasing chicken meat, and 40% of respondents believed that safety and price were essential in their decision. Over half of those surveyed believed that environmental problems arising from poultry production were generated from the production stage (including housing system) whereas only 5% considered that production of feed gave the negative impact.

Table 2: Willingness to pay (WTP) for chicken meat produced with a higher regard for the environment based on an absolute value (RM/KG) and percentage increment

	Average sale price	Mean WTP		SE		t-test Value		P Value	
	(RM/KG)	RM	%	Value	%	Value	%	Value	%
National	7.05	7.91	12.2	0.045	0.633	177.1	19.3	<0.001	<0.001
Northern	7.22	7.93	9.9	0.139	1.925	57.2	5.1	<0.001	<0.001
Central	6.86	7.71	12.4	0.065	0.940	119.4	13.2	<0.001	<0.001
Southern	6.53	7.34	12.5	0.071	1.093	102.9	11.4	<0.001	<0.001
Eastern	7.07	7.98	12.8	0.136	1.919	58.7	6.7	<0.001	<0.001

For estimation of the WTP values, there was a significant difference from current average price at both national and regional level as illustrated in Table 2. Half of the sample population at national level and in all regions confirmed that they were willing to pay more than they currently paid per kilogram of chicken meat during the survey period, by at least 10%. An ANOVA test showed that there was a statistically significant difference between groups; the mean value of WTP in the southern region was the lowest compared to other regions, however a Duncan pair-wise comparison of means only indicated that the absolute values, but not the percentage increments, were different between regions. These tests of association showed that variations in WTP were explained by occupation class and the number of persons in the household, with coefficients of determination of 12% and 18% respectively.

4. Discussion

The selection of the FU of one tonne live weight of broiler chickens was based on the function of the product and the choice of system boundary which satisfied economic expectations in terms of production rate and consumption. Three previous LCA studies on broiler production in other countries used one tonne live/dead weight while two studies used one tonne of edible carcass/meat (Williams et al., 2006; Katajajuuri, 2008; Leinonen et al., 2012). According to de Vries et al., (2010), the majority of economic value in live-stock comes from the production of meat, therefore the environmental impact should fully allocate to any form of the edible product. Their study also provided the calculation factor for any conversion between the various forms of edible products.

In the current study, the energy used to produce one chicken in the SI system was the highest of all three systems (26.5 MJ), due to the poorer feed conversion efficiency and longer time to reach finished weight, 2.42 and 84 days respectively. The burdens for GW impact arose mainly from broiler feed and transporta-tion; the comparison between production systems for one chicken showed the SI system released 3.62 kg CO_2 eq. compared to 2.89 kg and 2.74 kg CO_2 eq. for CH and OH systems respectively. The amount of feed consumed and the length of a complete production cycle (5.08, 3.86 and 3.91 kg and 84, 38 and 35 days re-spectively for SI, CH and OH systems) accounted for the differences in impact. Acidification and eutrophica-tion impacts were relatively low in the CH and OH systems compared to the SI system, determined mainly by the amount of ammonia and nitrogen oxide emissions, and the leaching of nitrogenous compounds from the manure. Based on the assumption applied in practice in Malaysia that the substitution ratio was 1:1 be-tween synthetic fertiliser with organic fertiliser, burdens associated with synthetic fertiliser production could be reduced as a result of applying poultry manure to palm oil plantation farms. This practice compensated 44 to 46%, 51 to 57%, 11 to 12% and 3% of total energy use, GWP, AP and EP emissions respectively.

The current study estimated that the energy used to produce one broiler chicken in the CH system in Ma-laysia was 19.24 MJ, which can be compared with standard production systems in the temperate regions requiring 32.63 MJ in the United Kingdom (UK) (Leinonen et al., 2012) and 33.84 MJ in the United States of America (USA) (Pelletier, 2008). Differences in energy use between countries partially reflect the differ-ent lengths of the production cycles in Malaysia, the UK and the USA which were 35, 39 and 48 days re-spectively. The origin of feed played a major role in the different values of GWP in Malaysia, since ap-proximately 90% of feed ingredients used for broiler production were imported, namely maize and soya bean from Argentina and wheat from Japan. Thus in the current study the value of 4.01 kg CO_2 eq. for one chicken in Malaysia compares to only 2.63 and 4.04 kg CO_2 eq. for one chicken in the USA and the UK re-spectively, countries in which a greater percentage of feed ingredients are produced locally.

The survey findings on socio-economic factors were consistent with many of the national economic indi-ces released by the Malaysian government, such as the Population and Housing Census, Gross Domestic Products (GDP), Consumer Price Index (CPI) and Quality Life Index. For example, in the survey 75% of respondents' stated that chicken meat was categorized as either important/very important in their diet with average per-capita consumption of 2.61 kg/person/week. This high priority attached to chicken meat can be explained by the urbanisation taking place in Malaysia; the proportion of population in Malaysia which is urban increased from 62% to 71% during the period 2000 to 2010 (DOS, 2011).

Perhaps one of the most interesting findings of the current study was consumer perception of the effect of poultry production on environmental quality. Respondents believed that the major contribution to environmental impact came from manure management and only 5% considered that the production of poultry feed contributed to environmental problems. In contrast, the LCA results showed that between 74 to 95% of environmental burdens were derived from feed (including the production, transportation and processing) for broiler chickens and breeder hens, and only 2.3 to 24% burdens resulted from manure handling. This contradiction can be explained by the difficulties of consumers in understanding the nature of broiler production in detail, coupled with non-visibility of the negative impacts in the short term. Thus, consumers may have considered the impacts of feed production, which occurred to a significant extent overseas, to be small when compared to other economic activities which produce visual effects, such as smoke from factories which can have a very large influence on public perception.

Even though the mean WTP values showed regional variation, the median values at both national and regional levels confirmed that many respondents were willing to pay 10% more than they currently paid for chicken meat produced with a higher regard for the environment. These two parameters can have quite different interpretations at the macro level of development planning. The mean WTP values have been used to determine the value of the benefit, which reflects the consumer behaviour and demand characteristics, and has high relevance for the purpose of cost-benefit analysis. Therefore if the mean benefit value outweighs the mean costs, this suggests that the proposal should proceed. On the other hand the median value indicates likely public motivation and choices, since it corresponds to the value which represents the majority endorsement. Respondents in the northern and eastern regions of Malaysia already have to pay 0.3 to 2.4% higher than the national average price for chicken, mainly as a result of the transportation and storage costs. Respondents from the northern region with mean WTP at 9.9%, which was the lowest value of all regions, may have been heavily influenced by the fact their incomes are generally lower than those of people in other regions.

5. Conclusions

The results suggest that the CH system was the least environmentally impacting of the three Malaysian broiler production systems considered for all impact categories, producing approximately 12 to 40% less impact than the highest contributor which was the SI system. Broiler and breeder feed-related resources accounted for almost three quarters of the total impact values, followed by other on-farm inputs and emissions produced. In relation to consumers' WTP for favourable environmental quality as a result of more environmental-friendly broiler production, half of the respondents were willing to pay an increment of 10% above market price in all regions, with type of occupation and the number of persons in the household determining the pattern of WTP among consumers. The analysis of the interaction of these agro-ecological and socio-economic findings, together with a comprehensive policy intervention, could be useful to determine best practice to ensure sustainable broiler production in Malaysia.

6. References

Best, P., 2011. Worldwide poultry meat production, consumption forecasts', WATTAgNet.com. Available at:http://www.wattagnet.com/Worldwide_poultry_meat_production,_consumption_forecasts.aspx. (accessed: 02/2012).

de Vries, M. and de Boer, I. J. M., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livestock Science 128, 1-11.

DOS, 2011. Consumer Price Index. Putrajaya, Malaysia: Department of Statistics Malaysia.

DVS, 2011. Annual Statistical Book. Putrajaya, Malaysia: Department of Veterinary Service Malaysia.

FAO, 2012. Food Outlook: Global market analysis. Rome, Italy: Food and Agriculture Organization. Available at: http://www.fao.org/giews/ (accessed: 06/2012).

Hanley, N., Shogren, J. F. and White, B., 2001. Introduction to environmental economics. Oxford University Press Inc. New York.

Katajajuuri, J.M., 2008. Experiences and improvement possiblities - LCA case study of broiler chicken production. Proceedings of 6th International Conference Life Cycle Assessment in the Agri-Food Sector. Zurich, Switzerland.

Leinonen, I., Williams, A. G., Wiseman, J., Guy, J. H. and Kyriazakis, I., 2012. Predicting the environmental impacts of chicken system in the United Kingdom through a life cycle assessment: Broiler production systems. Poultry Science 91, 1-18.

Mitchell, R. C. and Carson, R. T. 2005. Using surveys to value public goods: the contingent valuation method. Washington, D.C.

Narrod, C., Marites Tiongco, M., Costeles, A., 2008. Global poultry sector trends and external drivers of structural change. In Proceedings of the International Conference - Poultry in the Twenty-first Century: Avian Influenza and Beyond. Bangkok, Thailand.

Pearce, D., O¨zdemiroglu, E., Bateman, I., Carson, R.T., Day, B., Hanemann, M., Hanley, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Pearce, D.W., Sugden, R. and Swanson, J. (2002) Economic valuation with stated preference techniques: summary guide. Rotherham: Department for Transport, Local Government and the Region. Publication Sales Centre.

Pelletier, N., 2008. Environmental performance in the US broiler poultry sector: Life cycle energy use and greenhouse gas, ozone depleting, acidifying and eutrophying emissions. Agricultural System 98, 67-73.

Steinfeld, H., Wassenaar, T. and Jutzi, S. 2006. Livestock production systems in developing countries: status, drivers and trends. International Office of Epizootics 25(2), 505-516.

Williams, A. G., Audsley, E. and Sandars, D. L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Available at: http://www.defra.gov.uk (accessed 06/2010)

Environmental impacts accompanying the production of milk, bovine meat and grain under heterogeneous conditions in Norway

Anne-Grete Roer[1,*], Audun Korsaeth[2], Astrid Johansen[3], Anne Kjersti Bakken[3], Kristin Daugstad[4], Gustav Fystro[4], Trond Maukon Henriksen[2], Anders Hammer Strømman[1]

[1] The Industrial Ecology Programme, Department of Energy and Process Engineering, Norwegian University of Science and Technology, 7491 Trondheim, Norway

[2] The Norwegian Institute for Agricultural and Environmental Research, Arable Crops Department, 2849 Kapp, Norway

[3] The Norwegian Institute for Agricultural and Environmental Research, Department of Landscapes, Grassland and Forage Crops, 7500 Stjørdal, Norway

[4] The Norwegian Institute for Agricultural and Environmental Research, Løken Research Centre, 2940 Heggenes, Norway

[*] Corresponding author. E-mail: *anne-grete.roer@ntnu.no*

ABSTRACT

We will present key results from a large, ongoing, three-year project on environmental impact and resource use efficiency of selected food production chains in Norway (from cradle to store). The assessment presented here covers processes from cradle to farm gate, including all activities on the farm, along with the production of machinery, equipment, buildings, diesels, oil, fertiliser, lime, seeds, pesticides and medicines. Net mineralisation of humus, which may play a major role for greenhouse gas emissions under certain conditions, is also considered.

In the ongoing project, the ambition is to account for the outspoken heterogeneity, which characterises Norwegian agricultural systems, as soil type, management, climatic- and topographic conditions may vary largely between regions and between farms within the same region. To do so, we use high resolution inventory data, based partly on interviews with farmers, farm advisors, and agricultural experts, partly on data available in databases (e.g. soil properties, yields), and partly on models used for interpolations where robust data is lacking.

LCA studies of Norwegian agri-food chains are scarce in the peer reviewed literature. A better understanding of the environmental impacts associated with agricultural production in Norway is important for three main reasons: Firstly, such data is required for environmental benchmarking of various food pathways in order to assess the environmental profile of Norwegian food production versus imported products. Secondly, establishing and consolidating knowledge on the environmental profile of current production practices is essential to develop future agriculture policies in an increasing carbon constrained world. Thirdly, a recent political goal set for the agricultural sector in Norway push for an increase in production of 1% annually for the next 20 years, and this increase should be as environmental friendly as possible.

The objective of this study is to perform a life cycle assessment from cradle to farm gate for Norwegian cereal- and dairy production, covering the three most important regions for each production chain, respectively. Functional units are 1 kg of barley, oats, wheat, bovine carcass and energy corrected milk. There will be focused on identifying regional hotspots, and scenarios will be run in order to find possible, region wise improvements to reduce the overall environmental impact.

Assessing carbon, water and land use footprints for beef cattle production in southern Australia

Bradley G Ridoutt[1,*], Girija Page[2], Kimberley Opie[3], Jing Huang[4], William Bellotti[2]

[1] Commonwealth Scientific and Industrial Research Organisation (CSIRO), Sustainable Agriculture National Research Flagship, Private Bag 10, Clayton South, Victoria 3169, Australia
[2] University of Western Sydney, School of Science and Health, Locked Bag 1797, Penrith, NSW 2751, Australia
[3] CSIRO Land and Water, Private Bag 10, Clayton South, Victoria 3169, Australia
[4] China Agricultural University, College of Agriculture and Biotechnology, Beijing 100193, China
[*] Corresponding author. E-mail: *brad.ridoutt@csiro.au*

ABSTRACT

For agri-food products, concurrent assessment of GHG emissions, water use impacts and land use is necessary to communicate meaningfully about environmental performance and to avoid potential negative consequences of narrowly focussed environmental improvement initiatives, such as carbon footprint reduction. In this study, land use footprints were calculated for six diverse beef cattle production systems in southern Australia (cradle to farm gate) using net primary productivity of potential biomass (NPP_0) as a means of describing the intrinsic productive capability of land. The results per kg live weight, ranging from 86 to 172 m^2.yr-e (where 1 m^2.yr-e represents 1 m^2 of land occupation for 1 year at the global average NPP_0) represent between 1.3 and 2.7% of an average global citizen's annual land use footprint, and highlight the importance of land use in cattle production. These results were approximately 10 and 1000 times the normalised carbon and water footprint results. While NPP_0 can be used to improve land use assessment beyond a simple measure of land area, further development of the land use footprint indicator is recommended.

Keywords: livestock, meat, potential net primary productivity, global pressure on land resources, environmental labelling

1. Introduction

For agricultural and food products, potential environmental impacts related to greenhouse gas (GHG) emissions, water use and land use are typically of highest concern (Pfister et al., 2011; Ridoutt et al., 2011). However, there are also frequent tradeoffs between these sources of impact, meaning that the evaluation of alternative agri-food production systems and products is not straightforward. For example, land can be used for biodiversity conservation and carbon sequestration or it can be used for food production, and some forms of agriculture conserve more soil carbon, perennial biomass and biodiversity than others. Alternatively, actions to reduce GHG emissions in agriculture might require greater water use, and interventions to achieve water efficiency and water quality objectives might necessitate greater use of energy and consequently increase GHG emissions. Furthermore, a small area of irrigated agricultural land might produce as much food as a much larger area of non-irrigated land and thereby be considered land use efficient and beneficial in terms of minimising pressure on arable land resources.

This complexity highlights the futility of comparing the environmental performance of food production systems or products using a single stand-alone indicator. An important objective of life cycle assessment (LCA) is the avoidance of burden shifting, not only from one part of the product life cycle to another, but also from one environmental impact category to another. While carbon footprinting of products has been influential in raising awareness about GHG emissions and has even been described as a catalyst for life cycle thinking and management (Weidema et al., 2008), concern has also been raised that the practice violates the core LCA principle of *comprehensiveness*, meaning that consideration should be given to all relevant environmental impacts (Finkbeiner, 2009). Similar concerns could also be raised in relation to water footprints, which consider only water use impacts (Ridoutt, 2011).

In previous research, the carbon footprints (cradle to farm gate) for six diverse beef cattle production systems in southern Australia were assessed and found to range from 10.1 to 12.7 kg CO_2e kg^{-1} live weight (Ridoutt et al., 2011). This compared to LCA-based water footprints of 3.3 to 221 L H_2Oe kg^{-1} live weight for these same systems (Ridoutt et al., 2012), calculated using the Water Stress Index (WSI) of Pfister et al., (2009). Following Ridoutt and Pfister (2010a, 2012), the reference unit 1 L H_2Oe represents the burden on freshwater systems from 1 L of consumptive freshwater use at the global average WSI. The purpose of this paper is to complement these case study findings for beef cattle with novel land use footprint indicator results. The concurrent assessment of GHG emissions, water use and land use is considered a more reliable basis for assessing environmental sustainability than using a single stand-alone indicator, and for products in the agriculture and food sectors, this multi-footprint indicator approach could be considered sufficient to satisfy the comprehensiveness principle in LCA. To assist in the interpretation of the environmental profile for each livestock production system, the indicator results are also presented after normalisation.

In this study, a resource-based approach to land use footprinting has been trialled. This approach recognises that productive land is a scarce resource and that the utilisation of land for the production of any particular goods or services adds incrementally to the global demand for productive land and the associated wide ranging environmental impacts. In describing land as a resource, a simple quantitative measure (e.g. m^2.yr) is insufficient as land is not uniform in its productive capability. Land use footprinting must therefore incorporate the quality dimension of land use. In regards to biodiversity and ecosystem services-based approaches to modelling land use impacts in LCA, much progress has been made and a variety of new characterisation factors have been proposed relating to processes such as freshwater regulation, erosion regulation, water purification and carbon sequestration potential (de Baan et al., 2012; de Souza et al., 2012; Brandão and Milà i Canals, 2012; Bos et al., 2012; Saad et al., 2011, 2012). However, at this point in time, these methods have not generally reached an operational stage of development, lacking normalisation factors and coherence with established impact assessment methods which would allow evaluation of tradeoffs with other well established impact category indicators. The exception is the modelling of climate impacts of land use associated with carbon dioxide transfers between vegetation, soil and the atmosphere (Müller-Wenk and Brandão, 2010). Impact assessment methodologies which address individual ecosystem services and which lead to a profile of impact category indicator results relating to land use will be rich in detail and most beneficial in contexts where the LCA practitioner is reporting within the LCA expert community or where they have the opportunity to provide detailed explanation and interpretation to the decision maker. On the other hand, a simplified resource-based approach to land use footprinting, if it can be shown to be environmentally meaningful, could be beneficial in contexts where a single indicator, reported using an intuitively meaningful unit is required, such as in the situation of Type III eco-labelling.

2. Methods

2.1. System description

This case study concerns six geographically defined beef cattle production systems in the southern Australian state of New South Wales (NSW) where cattle are predominantly raised in mixed (i.e. livestock and cropping) farming systems. The six systems (Table 1) were selected in order to be diverse in farm practice (grass and feedlot finishing), product (12-15 month old yearling cattle to 24-36 month old heavy steers), environment (high-rainfall coastal to semi-arid inland) and local water stress (as defined by the WSI of Pfister et al., 2009). The system boundary was from cradle to farm gate and included all of the direct farming inputs (including replacement heifers and bulls), but excluded capital items such as machinery, buildings and other infrastructure. The functional unit was 1 kg live weight (LW) of beef cattle at the point of sale to the processor. Life cycle inventory data predominantly came from farm enterprise budgets compiled by the NSW government as a planning tool to assist farmers to evaluate business options. These budgets are regarded as being realistic and achievable by most professional farmers with good management practices. Further details are described in Ridoutt et al., (2012).

Table 1. Summary of the six geographically-defined beef cattle production systems[a]

Production system		Main product[b]	Location	Mean max Temp (°C)	Rainfall (mm yr^{-1})	WSI[c]
Japanese ox – grass-fed steers	JOS	24-36 mth old steers, 340 kg DW	Scone	24.1	644	0.032
EU cattle	EUP	24-30 mth old steers, 280-300 kg DW	Parkes	23.4	584	0.815
Inland weaners, grass fattened and feedlot finished	IGF	24 mth old steers, 585 kg LW	Walgett	26.9	477	0.021
			Gunnedah	26.0	619	0.021
			Quirindi	24.6	683	0.021
North coast weaners, grass fattened and feedlot finished	NGF	24 mth old steers, 585 kg LW	Casino	26.7	1096	0.012
			Glen Innes	19.4	849	0.021
			Rangers Valley	19.4	849	0.021
Yearling	YG	12-15 mth old yearling, 185-205 kg DW	Gundagai	22.3	713	0.815
Yearling	YB	12-15 mth old yearling, 185-205 kg DW	Bathurst	19.8	635	0.021

[a] Based on data presented in Ridoutt et al., (2012), [b] DW: dressed weight or dressed carcass weight after removal of hide, head, feet, tail and internal organs; LW: live weight; [c] WSI: Water Stress Index (Pfister et al., 2009).

2.2. Carbon footprint modelling

The carbon footprint modelling is described in detail elsewhere (Ridoutt et al., 2011). In summary, the calculation of GHG emissions from livestock enteric fermentation, manure and urine followed the country specific IPCC Tier 2 approach used in Australia's national GHG inventory, taking into account herd structure on a daily time step, feed quality and growth rate. The Australian national GHG inventory methods were also applied in relation to emissions from agricultural soils as a result of inorganic nitrogen fertiliser application and the residue of cultivated leguminous pastures. Recent land use change (deforestation) was not a feature of any of the systems and possible changes in soil carbon were ignored due to a lack of relevant data. Data relating to GHG emissions associated with electricity, fuels (used on farm and in transportation processes), fertiliser production, supplementary feeds (grain, pasture hay and feedlot ration) and veterinary products were obtained from various Australian database sources. To calculate the carbon footprint, the latest 100-year global warming potentials for GHGs published by the IPCC were used.

2.3. Water footprint modelling

The water footprint modelling was based on consumptive water use only, as a recently developed method integrating consumptive and degradative water use into a single indicator was not available at the time (Ridoutt and Pfister, 2012). A complete description of the water flows quantified is presented in Ridoutt et al., (2012). In summary, the inventory included flows from surface and groundwater into the farming system to irrigate pastures as well as crops used for supplemental feeding on farm and in the feedlot. Secondly, it included the reduction in flows from the farming land base to surface and groundwater as a result of the operation of farm dams used for livestock watering. Thirdly, it included direct water use in feedlot operations. Finally, it included water use associated with the production of inputs to the farming and feedlot operations. The indicator results, in the units L H_2Oe, were calculated by multiplying each spatially differentiated instance of water use by the locally relevant WSI and dividing by the global average WSI (0.602).

2.4. Land use footprint modelling

For each beef cattle production system, an inventory of geographically-defined agricultural land use was compiled. The land use types were unimproved pasture, non-irrigated improved pasture, irrigated improved pasture and cropland. The inventory excluded land use associated with the built environment (e.g. roads, factories) and land use associated with the extraction of resources from nature (e.g. mining of rock phosphate, extraction of oil). Land use was expressed in the unit m^2.yr, taking into consideration the duration of occupation. Land was considered to be occupied if it was unavailable for other productive purposes. As such, single cropping systems, which are prevalent in Australia, were deemed to occupy the land for the complete year, even if production only occurred during part of the year. The inventory did not include situations of multiple land use, such as mixed grazing of livestock, recreation and timber production, meaning that land use was completely attributed to agricultural production. On the basis that biodiversity is largely reduced by intensive agricultural production systems (Pfister et al., 2011), no attempt was made to describe the remaining ecological value of the land.

For land use footprinting, the net primary productivity of potential biomass (NPP_0, g $C.m^{-2}.yr^{-1}$) was used to describe the intrinsic productive quality of land. Our reasoning is that the occupation of high-NPP_0 land exerts more pressure on global land resources than the occupation of low-NPP_0 land. This is obviously a simplification as factors in addition to NPP_0 determine the desirability of land in terms of human development. However, NPP_0 is a useful starting point and it is an objective measure for which global datasets exist. Land use footprint indicator results were calculated by multiplying each spatially-differentiated instance of land use (m^2.yr) by the relevant NPP_0 associated with each land use type in each area, and dividing by the global average NPP_0. The indicator results were expressed in the reference unit m^2.yr-e, where 1 m^2.yr-e represents 1 m^2 of land occupation for 1 year at the global average potential net primary productivity. To perform the assessment, land use maps were obtained from the Australian Government Bureau of Rural Sciences (resolution 1 km) and NPP_0 values at a resolution of 5 arc min were obtained from Haberl et al., (2007).

2.5. Normalisation

To assist in interpretation of the life cycle impact category indicator results, normalisation was performed, which is the step involving comparison of indicator results to a common reference situation. In this study, the chosen reference was the global economic system in the years 1995-2000 and results are reported in person equivalents. The land use footprint of humanity was calculated using data reported in Haberl et al., (2007) and Erb et al., (2007). To be consistent with the method of calculation of land use footprints described above

(Section 2.4), the cropland, infrastructure and grazing land of suitability class 1 (highly productive grazing land) land use classes were deemed to be fully occupied (i.e. effectively unavailable for other productive uses). In contrast, the forestry and grazing lands of poorer suitability were deemed to be multiple-use and occupied in proportion to the fraction of NPP_0 appropriated. As such, humanity's land use footprint was assessed at 3.92E+13 m^2.yr-e, or 6.40E+03 m^2.yr-e per inhabitant, in the year 2000 (Table 2). To avoid possible misunderstanding, it is stressed that our land use footprint is a measure of land occupation, not human appropriation of net primary production.

Table 2. Normalisation factors used in this study.

Indicator	Unit	Factor	Year	Reference
Carbon footprint	kg CO_2-e $person^{-1}$ yr^{-1}	6.83E+03	2000	Sleeswijk et al., 2008
Water footprint	L H_2O-e $person^{-1}$ yr^{-1}	6.73E+05	1995	Ridoutt and Pfister, 2010b
Land use footprint	m^2.yr-e $person^{-1}$ yr^{-1}	6.40E+03	2000	Haberl et al., 2007; Erb et al., 2007

3. Results

3.1. Land use inventory results

The life cycle (cradle to farm gate) agricultural land use associated with the six beef cattle production systems varied from 64.0 to 121.1 m^2.yr/kg LW. However, this land use cannot be directly compared as the proportions of unimproved and improved pasture and cropland varied (Table 3). The land use quality, as described by the net primary productivity of potential biomass, also varied, from a high of 944 g C.m^2.yr^{-1} (high rainfall coastal cropland) to a low of 355 g C.m^2.yr^{-1} (semi-arid unimproved pastureland).

Table 3. Land use inventory results (m^2.yr per kg live weight at farm gate) for 6 diverse beef cattle production systems in southern Australia.

	JOS	EUP	IGF	NGF	YG	YB
Pasture-unimproved	93.6	0	77.3	69.2	0	0
Pasture-improved, non-irrigated	21.1	69.1	17.2	14.2	62.9	64.0
Pasture-improved, irrigated	0.6	0	0.2	<0.1	1.1	0
Cropland	5.8	5.9	9.1	9.6	0	0

3.2. Land use footprint results

The land use footprint indicator results ranged from 86 to 172 m^2.yr-e/kg LW (Table 4). In most cases the numerical value of the land use footprint indicator result (m^2.yr-e) exceeded the inventory result (m^2.yr) because the potential net primary productivity of the land used in production exceeded the global average (502 g C.m^2.yr^{-1}). The exception was the beef cattle production system with weaner production in Walgett (IGF), where the NPP_0 of unimproved and non-irrigated improved pasture were below the global average, i.e. 355 and 454 g C.m^2.yr^{-1} respectively. The largest land use footprint (172 m^2.yr-e/kg LW, Table 4) was associated with the production of heavy steers on mainly unimproved pasture at Scone (JOS). In this region, lands used for unimproved pasture have a relatively high NPP_0 (708 g C.m^2.yr^{-1}), but are mainly hilly and with shallow

Table 4. Comparison of carbon footprint (CF), water footprint (WF) and land use footprint (LUF) results for 6 diverse beef cattle production systems in southern Australia (per kg live weight at farm gate).

	JOS	EUP	IGF	NGF	YG	YB
Indicator results:						
CF kg CO_2-e	10.2	10.8	10.1	12.7	10.4	10.6
WF L H_2O-e	14.4	68.3	9.1	7.7	221	3.3
LUF m^2.yr-e	172	100	88	128	86	88
Normalised results:						
CF person.year equiv (%)	0.15	0.16	0.15	0.19	0.15	0.16
WF person.year equiv (%)	0.0021	0.010	0.0014	0.0011	0.033	0.00049
LUF person.year equiv (%)	2.7	1.6	1.4	2.0	1.3	1.4

soils. The smallest land use footprint (86 m^2.yr-e/kg LW) was associated with yearling production on lower NPP_0 (672 g C.m^2.yr^{-1}) improved pasture at Gundagai (YG) where mixed (livestock and cropping) farming systems are common. It is important to note that there was no apparent correlation between carbon, water and land use footprints. The system with the smallest land use footprint (YG), had the highest water footprint (221 L H_2O-e/kg LW, Table 4) and a mid range carbon footprint (10.4 kg CO_2-e/kg LW). In comparing the normalised results (Table 4), the land use footprint results were approximately 10 times the carbon footprint results (range 9 to 18 times) and approximately 1000 times the water footprint results (range 39 to 2850).

4. Discussion

This study has highlighted the importance of land use in beef cattle production in southern Australia, with the production of 1 kg of animal live weight requiring 1.3 to 2.7% of an average global citizen's annual land use footprint (Table 4). For 1 kg of retail beef, this equates to between 3 and 7% of an average global citizen's annual land use footprint. Globally, the livestock sector is a major land user, estimated to occupy around 30% of the world's land surface (Steinfeld et al., 2006). However, it is important to recognize that the land base supporting livestock production is diverse, including high productivity crop and pasture land as well as large areas of low productivity and non arable land. As such, the land use footprint calculation method demonstrated in this paper, which takes into account the NPP_0 of the land used in production, is considered a better indicator of the pressure on global land resources than an assessment based on land area alone. This is particularly important in evaluating alternative livestock production systems as globally there is expansion in industrialised systems utilizing high nutrition feeds such as grains and oilseeds (de Haan et al., 2010). The land base supporting livestock production is therefore in transition toward greater dedicated use of high productivity cropland.

However, the NPP_0 basis for assessing land quality is not completely satisfying. Firstly, there are additional factors which determine the productive capability of land. For example, a global assessment of land capability by Fischer et al., (2001) found that 12% of land is severely constrained for crop cultivation by slope and 65% by unfavourable soil quality. The NPP_0 values used in this study, taken from Haberl et al., (2007), incorporated climatic factors and a soil-type classification, but the spatial resolution of soil type (0.5^o, approx 60 km at the equator) is too coarse to describe much of the variation at the farm level which influences enterprise decision making for mixed farm systems in Australia. The second issue concerns the potential substitutability of one land use with another, which is incompletely described by NPP_0. Some pasture land may have high NPP_0, but have poor suitability for cropping, and therefore should not be regarded as contributing to global pressure on arable land resources. On a global scale, NPP_0 and crop yield are poorly correlated (West et al., 2010). NPP_0 is highest in the tropics whereas the highest crop yields are generally found in temperate regions. In addition, sequences of legume-based pastures can offer benefits to crops, such as improved soil fertility and disease break for cereal root pathogens (Jensen et al., 2012). Thirdly, a simplified NPP_0 based approach to land use footprinting has the potential to encourage excessive land use intensification and land degradation, which is another way of increasing pressure on the earth's land resources. These and other issues point to the need for further development of the land use footprint indicator.

5. Conclusion

Land resources are currently under stress, and with a world population increasing toward 9 billion inhabitants, the increased demand for food, fibre, and increasingly biofuel, must be met in ways which do not lead to continuing loss of natural ecosystems and expanding land degradation. An LCA-based land use footprint indicator could help in understanding the incremental pressure on land resources of agri-food production systems and consumption patterns, and enable the assessment of tradeoffs with GHG emissions and water use. While NPP_0 is an objective measure of the intrinsic productive capability of land and can be used to improve land use assessment beyond a simple measure of land area alone, further development of the land use footprint indicator is recommended.

6. References

Bos, U., Beck, T., Linder, J.P., Wittstock, B. 2012. Land use impact assessment of ecosystem services according to LANCA. Special Forum on Global Land Use Impacts on Biodiversity and Ecosystem Services in LCA within the framework of the UNEP/SETAC Life Cycle Initiative, Brussels, Feb 17, 2012.

Brandão, M., Milà i Canals, L., 2012. Global characterisation factors to assess land use impacts on biotic production. Int. J. Life Cycle Assess. DOI 10.1007/s11367-012-0381-3.

de Baan, L., Alkemade, R., Koellner, T., 2012. Land use impacts on biodiversity in LCA: a global approach. Int. J. Life Cycle Assess. DOI 10.1007/s11367-012-0412-0.

de Haan, C., Gerber, P., Opio, C., 2010. Structural change in the livestock sector, in: Steinfeld, H., Mooney, H.A., Schneider, F., Neville, L.E. (Eds.), Livestock in a Changing Landscape. Island Press, London, pp. 35-66.

de Souza, D.M., Flynn, D., Koellner, T., 2012. Land use impacts on functional species diversity: proposal of characterization factors to assess effects on ecosystem processes. Special Forum on Global Land Use Impacts on Biodiversity and Ecosystem Services in LCA within the framework of the UNEP/SETAC Life Cycle Initiative, Brussels, 17th February 2012.

Erb, K.H., Gaube, V., Krausmann, F., Plutzar, C., Bondeau, A., Haberl, H., 2007. A comprehensive global 5 min resolution land-use

data set for the year 2000 consistent with national census data. J. Land Use Sci. 2, 191-224.
Finkbeiner, M., 2009. Carbon footprinting—Opportunities and threats. Int. J. Life Cycle Assess. 14, 91-94.
Fischer, G., Shah, M., van Velthuizen, H., Nachtergaele, F.O., 2001. Global Agro-ecological Assessment for Agriculture in the 21st Century. International Institute for Applied Systems Analysis, Laxenburg, Austria.
Haberl, H., Erb, K.H., Krausmann, F., Gaube, V., Bondeau, A., Plutzar, C., Gingrich, S., Lucht, W., Fischer-Kowalski, M., 2007. Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. Proc. Natl. Acad. Sci. USA 104, 12942-12947.
Jensen, E.S., Peoples, M.B., Boddey, R.M., Gresshoff, P.M., Hauggaard-Nielsen, H., Alves, B.J.R., Morrison, M.J., 2012. Legumes for mitigation of climate change and the provision of feedstocks for biofuels and biorefineries: A review. Agron. Sustain. Dev. 32, 329-364.
Müller-Wenk, R., Brandão, M., 2010. Climatic impact of land use in LCA: Carbon transfers between vegetation/soil and air. Int. J. Life Cycle Assess. 15, 172-182.
Pfister, S., Bayer, P., Koehler, A., Hellweg, S., 2011. Environmental impacts of water use in global crop production: Hotspots and trade-offs with land use. Environ. Sci. Technol. 45, 5761-5768.
Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43, 4098-4104.
Ridoutt, B.G., 2011. Development and application of a water footprint metric for agricultural products and the food industry, in: Finkbeiner, M. (Ed.), Towards Life Cycle Sustainability Management. Springer, Dordrecht, pp. 183-192.
Ridoutt, B.G., Pfister, S., 2010a. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Glob. Environ. Change 20, 113-120.
Ridoutt, B.G., Pfister, S., 2010b. Reducing humanity's water footprint. Environ. Sci. Technol. 44, 6019-6021.
Ridoutt, B.G., Pfister, S., 2012. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. Int. J. Life Cycle Assess. (in press).
Ridoutt, B.G., Sanguansri, P., Freer, M., Harper, G., 2012. Water footprint of livestock: Comparison of six geographically-defined beef production systems. Int. J. Life Cycle Assess. 17, 165-175.
Ridoutt, B.G., Sanguansri, P., Harper, G.S., 2011. Comparing carbon and water footprints for beef cattle production in southern Australia. Sustain. 3, 2443-2455.
Saad, R., Margni, M., Koellner, T., Wittstock, B., Deschênes, L., 2011. Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context. Int. J. Life Cycle Assess. 16, 198-211.
Saad, R., Koellner, T, Margni, M., 2012. Land use impacts on freshwater and erosion regulation and water purification: A spatial approach for a global scale level. Int. J. Life Cycle Assess. (in press).
Sleeswijk, A.W., van Oers, L.F.C.M., Guinée, J.B., Struijs, J., Huijbregts, M.A.J., 2008. Normalisation in product life cycle assessment: An LCA of the global and European economic systems in the year 2000. Sci. Total Environ. 390, 227-240.
Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., de Haan, C., 2006. Livestock's Long Shadow: Environmental Issues and Options. FAO, Rome.
Weidema, B.P., Thrane, M., Christensen, P., Schmidt, J., Løkke, S., 2008. Carbon footprint: A catalyst for life cycle assessment? J. Ind. Ecol. 12, 3-6.
West, P.C., Gibbs, H.K., Monfreda, C., Wagner, J., Barford, C.C., Carpenter, S.R., Foley, J.A., 2010. Trading carbon for food: Global comparison of carbon stocks vs. crop yields on agricultural land. Proc. Natl. Acad. Sci. USA 107, 19645-19648.

Effect of farming practices for greenhouse gas mitigation and subsequent alternative land-use on environmental impacts of beef-cattle production systems

T.T.H. Nguyen[1,2,3,4,*], M. Doreau[1], M. Eugène[1], M.S. Corson[2,3], F. Garcia-Launay[1,3,5], G. Chesneau[4], H.M.G. van der Werf[2,3]

[1]INRA / Vetagro Sup, UMR1213 Herbivores, 63122 Saint-Genès-Champanelle, France
[2]INRA, UMR1069 Soil Agro and hydroSystem, 35000 Rennes, France
[3]Agrocampus Ouest, 35000 Rennes, France
[4]Valorex, La Messayais, 35210 Combourtillé, France
[5]INRA, UMR1348 Physiology, Environment and Genetics for Animals and Livestock Systems, 35590 Saint-Gilles, France

ABSTRACT

This study evaluated effects of farming-practice scenarios aiming to reduce greenhouse gas (GHG) emissions and alternative land-use on environmental impacts of a beef-cattle production system. Farming-practice scenarios modified an element of grassland management, herd management or diet, as well as a set of simultaneous compatible practices. The most promising practice was a combination of several scenarios (reduction of impacts by 13-28% per kg of carcass), followed by advancing first calving age from 3 to 2 years (reduction by 8-10%). For other scenarios, impact reduction did not exceed 5%, except for eutrophication (11%) and land occupation (10%). Some scenarios resulted in reduction of permanent grassland area and land occupation per kg of carcass. If this permanent grassland were converted to forest, climate change impact could be reduced by 20-48%. These results illustrate the potential of farming practices and forest as an alternative land use to contribute to GHG mitigation of beef production systems.

Keywords: beef cattle, farming-practices, alternative land-use, environmental impacts, life cycle assessment

1. Introduction

Greenhouse gas (GHG) emissions are a major concern for livestock production worldwide, in particular ruminant production (Steinfeld et al., 2006). Numerous GHG mitigation strategies for ruminant production have focused on a single GHG such as enteric methane (CH_4) or nitrous oxide (N_2O) (Martin et al., 2010; Eckard et al., 2010). Measures to enhance carbon (C) sequestration in the soil have also been identified (Dawson and Smith, 2007). However, it is critical to ensure that there is a net reduction in GHG emissions of the whole production system when such measures are implemented (Beauchemin et al., 2011), i.e. that a reduction in on-farm GHG emissions is not compensated by an increase in off-farm GHG emissions due to imported feed. Therefore, these measures need to be assessed at the scale of the entire production system. Besides GHG emissions, other environmental impacts such as energy use, eutrophication and land-use impacts may be of major importance depending on the local or regional context (Steinfeld et al., 2010).

The present study analysed the effects of farming practices aiming to reduce GHG emissions of beef-cattle production systems on their environmental impacts using the life cycle assessments approach. The baseline beef production scenario, described by Nguyen et al., (2012), reflected the current farm characteristics, management practices by farmers of Charolais beef cattle in France. Alternative land-use was assessed by assuming that any permanent grassland becoming available due to more efficient farming practices was converted to even-aged forest. Eight scenarios were assessed, as well as the sum of all compatible scenarios.

2. Materials and methods

2.1. System boundaries

Life cycle assessments of beef-cattle production systems were conducted from cradle to farm-gate for a one-year period, i.e. including the production and delivery of inputs used for grassland and cereals produced on-farm and for feed produced off-farm, herd management and associated upstream processes, emissions from the animals and manure storage. Environmental burdens from the application of manure for cereals and pasture were included, as were those from buildings. Veterinary medicines were excluded due to lack of data. The impacts, i.e. climate change (CC, excluding and including the effects of land use and land-use change (LULUC)), cumulative energy demand (CED), eutrophication (EP) and land occupation (LO), of different farming-practice scenarios were compared per 1 kg of carcass mass at the farm exit and per 1 ha of (on-farm and off-farm) land occupied. Per 1 kg of carcass mass, if farming practices reduced permanent grassland occupation, this released land was converted to fast-growing conifer even-aged forest as an alternative land-use to increase the amount of C sequestered by the beef system. Planting and main management stages were from Corsican pine (*Pinus nigra* subsp. *laricio*) and amortized over 64 years (Vallet et al., 2009).

2.2. Description of baseline of beef-production system

The baseline beef-production system (corresponding to system St-SM described in Nguyen et al., (2012)) comprised a cow-calf herd and a bull-fattening herd. The cow-calf herd included 70 cows that produced 62 weaned calves each year. These cows had their first calving at 3 years, and each provided a mean of 4.4 calves over their lifetimes. All weaned female calves were reared as heifers (3% mortality) used for replacement cows until 27 months. Of the 30 heifers thus produced, 14 were not selected for replacement and were fattened in pasture complemented with cereals and slaughtered at 33 months. Cull cows were finished for 100 days before being sent to the slaughterhouse. One male calf was selected to replace the breeding bull, and the rest were sent to the bull-fattening herd at 11 months.

The cow-calf herd ration was mainly based on grassland with 1.2 livestock units (LU) per ha of grassland and 7.5 months of grazing. One LU is defined as an animal that consumes 5 t dry matter (DM)/year. We assumed that permanent grassland did not require tilling and sowing operations. Apart from manure excreted on pasture during grazing, permanent grassland was fertilised with mineral and organic N-fertilisers (contributing 28 and 27 kg/ha of N, respectively). Permanent grassland potential yield was 5.6 t DM/ha/year, 23% of which was harvested as conserved forage (hay and/or wrapped grass-silage). Temporary grassland, a combination of grasses and clover, had a higher potential yield (8.3 t DM/ha/year, 75% harvested as conserved forage) and was renewed every 5 years by tillage and seeding. Mineral N-fertiliser for temporary grassland was applied at 33 kg/ha. Grass not harvested as conserved forage was available for ingestion by animals during grazing. For several reasons (selective grazing, trampling of grass, unfavourable weather conditions), some of the grass grown is not ingested; this “loss” corresponded to 31.5% of grass DM available for grazing. Losses during conservation of both hay and wrapped grass-silage were assumed to be 6% of the initial DM. Indoors in winter, the herd was fed hay and concentrates (mainly based on cereals produced on-farm and imported protein supplement containing 30% soybean meal, 40% rapeseed meal and 30% sunflower meal).

Male calves in the standard bull-fattening herd were fed a high-forage diet composed of 58% maize silage, 24% wheat, 15% soybean meal, 2% hay, and 1% minerals (DM basis), resulting in an average daily live weight gain (ADG) of 1.40 kg. All rations were formulated to satisfy beef-cattle nutrient requirements according to animal characteristics and feed-composition values, based on recommendations of INRA beef researchers and data tables (INRA, 2007). The carcass yields of fattened bulls, the breeding bull, finished heifers and finished cull cows were 59%, 57%, 56% and 54%, respectively. Methods used to produce feed ingredients and to estimate emissions from animals were described in Nguyen et al., (2012).

2.3. Farming-practice scenarios

Farming-practice scenarios (denoted S1 to S9) were designed to reduce GHG emissions of the beef-cattle production system. These practices are already applied by some farmers or can be applied without adverse effect on animal performances, based on experimental results. The use of these practices, both individually and simultaneously has been studied. When farming practice affected total feed requirements, the land area needed was adjusted to produce feed. Feed ingredients were produced by the same practices as in the baseline scenarios.

Grassland management (S1-S2)

Scenario S1 assessed effects of decreasing mineral N-fertiliser from 28.0 to 18.5 kg/ha of permanent grassland. The yield of permanent grassland was not affected because current mineral N-fertiliser application levels exceed the optimum level required for grass growth (Devun J., pers. comm.). Estimated nitrate losses were reduced from 20 to 14 kg N/ha.

Scenario S2 evaluated effects of decreasing grass losses (i.e. grass that is not ingested by the cows) on pasture from 31.5 to 16.5% (Devun J., pers. comm.). This reduction can be obtained by better management of grassland, i.e. turn out to pasture as soon as possible, rotational grazing, adjust animal density for grazing during dry season. Estimated nitrate losses were reduced from 20 to 14 kg N/ha.

Herd management (S3-S5)

Scenario S3 evaluated effects of fattening of female calves from 9 to 19 months instead of rearing them as heifers used for replacement and fattening them on pasture for 4 months until slaughter at 33 months. Four-

teen female calves after weaning not selected for replacement were fattened (until 650 kg LW) with a diet based on maize silage, resulting in an ADG of 1.15 kg.
Scenario S4 evaluated effects of increasing longevity of cows from 7 to 9 years to provide a mean of 6.5 calves per lifetime instead of 4.4 calves. As a consequence, the number of culled cows decreased (from 16 to 13), and the number of heifers used for meat production increased (from 14 to 17).
Scenario S5 evaluated effects of decreasing first calving age from 3 to 2 years simulated based on Farrié et al., (2008). All female calves were reared to reach 467 kg LW (instead of 405 kg) at 15 months for the first breeding. Heifers not used for replacement at 15 months were fattened to slaughter at 23 months (about 670 kg LW) instead of 33 months (at 698 kg LW). Replacement rate was slightly lower (21.4%) than in the baseline (23%) scenario; although these cows produced more calves (mean = 4.7 instead of 4.4) per lifetime, they were culled sooner (at 6 years and 780 kg LW instead of at 7 years and 800 kg LW).

Feeding practices (S6-S8)
Scenario S6 evaluated the effects of replacing some protein supplement with lucerne hay during the winter. A portion of temporary grassland was used to produce lucerne hay, and the protein supplement was decreased from 6.8 to 2.3 t. Lucerne hay contributed 12.4% of the total hay production.
Scenario S7 evaluated effects of using rapeseed meal to replace soybean meal in the bull diet.
Extruded linseed was used in scenario S8 to replace a portion of concentrate (cereals and protein supplement) in the cow-calf herd. Lipid content in diets for animals was not to exceed 3% DM. Male calves were sent to the bull-fattening herd after weaning (350 kg LW) and were fed with concentrate-based diet rich in lipids (13% barley straw and 83% concentrate including 46% cereals and 6% extruded linseed) resulting an ADG of 1.71 kg.

Combination of scenarios S1, S2, S4, S5 and S7 (S9)
This scenario (S9) is the addition of five scenarios, which were compatible, and of which effects are expected to be additive: decrease in mineral-N fertiliser (S1), in grass losses on pasture (S2), increase in cow longevity (S4), decrease in age at first calving (S5) and replacement of soybean meal with rapeseed meal (S7).

2.3. Alternative land use: fast-growing conifer even-aged forest

An alternative land-use option was explored to reduce GHG emissions of the farm system. If farming practices reduced permanent grassland occupation per kg of carcass mass, this land area released due to more efficient farming practices was converted to a conifer even-aged forest, which is more attractive to farmers than deciduous hard-wood species, due to its faster growth. Corsican pine was chosen because it is well adapted even on poor sites, provides good wood quality and has been successful in several French regions. It further may enhance on-farm biodiversity. We assumed a 64-year rotation, during which the forest sequesters 11.4 t CO_2/ha/yr into the vegetation (Vallet et al., 2009). The main function of the forest within the beef-farm system being C sequestration, we did not include the harvest of the trees (which would occur 64 years after planting), neither concerning inputs required nor with respect to the products it would yield. We did include inputs required for the plantation of the forest and its management during the establishment phase.

3. Results and discussion

3.1. Effects of farming practices on CC, CC/LULUC, CED, EP and AC

Grassland management (S1-S2)
Decreasing mineral N-fertiliser application to permanent grassland (S1) slightly decreased impacts of the whole system (reduction between 1 and 2%), except for CED and EP (by 2.9 and 10.5%, Table 1), because its use was already low in the baseline. It can, however, reduce production costs.
Decreasing grass loss on pasture (S2) did not affect CC/LULUC (reduction <1%), slightly decreased CC and AC, and decreased CED and EP per kg carcass mass by 2.8 and 10.8%, respectively. The main advantage of S2 is a reduction in grassland occupied per kg of beef produced. However, it requires more work from farmers for grassland management, in particular adapting grazing to grass growth by the systematic use of rotational grazing.

Herd management (S3-S5)

Fattening female calves instead of rearing them as replacement heifers (S3) slightly decreased CED, and decreased other impacts per kg carcass mass by 4-5%. This is due to their faster growth, resulting in less rearing time before slaughter. Also, their enteric CH_4 emissions were lower as they were fed with maize silage and concentrate instead of mainly forage. As maize silage has a higher yield per ha than grass, the area of grassland used for the herd decreased. Even though this practice increased the use of feed-crops, it can be considered as a potential climate change mitigation practice.

Increasing cow longevity (S4) slightly decreased impacts of the whole system, as the annual number of cull-cows decreased but that of finished heifers increased. Using different allocation methods, Nguyen et al., (2012) showed that impacts per kg carcass mass of finished heifers slaughtered at 33 months were higher than those of 7-year-old cull cows (except for mass allocation). In S5, impact reductions obtained by extension of cow lifetime were compensated by high impacts of finished heifers. Beauchemin et al., (2011) observed a similar result for GHG emissions and argued that the additional beef produced had higher per-kg GHG emissions. This practice will mitigate impacts more if impacts of finished heifers could be reduced. It is possible that combining this practice with fattening female calves instead of rearing them as replacement heifers (S3) could reduce impacts of the entire system.

Decreasing calving age (S5) seems one of the most effective impact-mitigation strategies, as impacts decreased by 8-10% due to two effects. First, all heifers were reared at higher growth rates to reach minimum body condition for first breeding at 15 months and first calving at 24 months instead of 27 and 36 months, respectively. In this way, one year of cow rearing (6 instead of 7 years) was saved without reducing reproductive yield per lifetime. Second, heifers not used for replacement also grew faster, thus finishing sooner (23 instead of 33 months), reducing impacts of the whole production system (as explained for S3). First calving at 2 years is the current practice in western Canada (Beauchemin et al., 2011). In France, first calving at 2 years with the Charolais breed was begun in experimental farms and later implemented by some innovative farmers (Farrié et al., 2008). Changing first calving from 3 to 2 years for half of a Charolais herd improved profit when the number of calvings per cow was increased by 5-10% (Farrié et al., 2008).

Feeding practices (S6-S8)

The partial replacement of protein supplement by lucerne hay during the winter (S6) did not affect impacts per kg carcass mass. This is due to the percentage of protein supplement replaced being small (0.8% of total DM intake of the cow-calf herd) and only 30% of it was soybean meal.

The replacement of soybean meal by rapeseed meal in bull diets (S7) had modest effects on the impacts of the whole system, even though it decreased the CC/LULUC and CED impacts of the bull-fattening herd by 9 and 22%, respectively (results not shown).

The use of extruded linseed to increase lipid content in animal diets (S8) decreased CC and CC/LULUC per kg carcass mass by 3-4%, which was lower than the 11% decrease obtained by Beauchemin et al., (2011). This difference is due to including a lower percentage of lipids in the winter cow-calf diet in this study than in that of Beauchemin et al., (2011) (1.2 vs. 4.0%, respectively). However, S8 increased CED and EP by 8.0 and 6.7%, respectively due to an increase in the use of cereals and in energy requirements for linseed production, the extrusion process and concentrate production for finishing and cow-calf diets.

Combination of scenarios S1, S2, S4, S5 and S7 (S9)

A combination of several compatible scenarios (S9) appeared the most promising impact-mitigation strategy, which decreased CC, CC/LULUC and AC per kg carcass mass by 13%, CED by 18% and EP by 28%. Overall, the effects of each farming practice on impact were limited because they only affected one element of the whole system. Combination of several compatible farming practice scenarios approximately results in the sum of the relative effects of the separate scenarios. However, possible interactions between these practices were not considered due to lack of experimental data.

Table 1: Environmental impacts (per kg carcass mass) of baseline for standard beef-cattle production and farming-practice scenarios

Impact	**Unit**	**Baseline**	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**	**S9**
Climate change	kg CO_2 eq.	27.8	27.3	27.2	26.4	27.4	25.5	27.9	27.8	26.9	24.2
Climate change/ LULUC*	kg CO_2 eq.	25.5	25.0	25.3	24.6	25.1	23.5	25.5	25.2	24.4	22.2
Cumulative energy demand	MJ eq.	65.0	63.1	63.2	64.2	64.3	59.8	64.7	62.6	70.2	53.4
Eutrophication	g PO_4^{3-} eq.	98.7	88.3	88.1	94.4	97.3	90.7	98.1	98.0	105.4	71.0

3.2. Effects of farming practices on LO and alternative land use on CC/LULUC

Farming practices such as decreasing grass loss on pasture (S2), fattening female calves instead of rearing them as replacement heifers (S3), decreasing calving age (S5) and combination of S1, S2, S4, S5, S7 (S9) decreased the use of permanent grassland and total land occupation per kg of carcass mass by 12-23% and 9-19%, respectively (Table 2). For these scenarios, if Corsican pine were planted on the released permanent grassland, CC/LULUC both per kg of carcass mass decreased by 19-48% (20.5, 19.2, 18.7 and 13.4 kg CO2 eq./kg carcass mass for S2, S3, S5 and S9, respectively). Corsican pine planted on released permanent grassland did not affect CED. This introduction of alternative land use influenced the CC/LULUC impact of the entire production system when comparing farming practices. Besides forest, there is no other alternative use of permanent grassland that can increase C sequestration in soil and biomass. This option appeared the most promising GHG mitigation strategy of beef production system without altering the farm's productivity. Forest plantation may also enhance biodiversity of the production system.

This study did not include the harvest of the even-aged forest, as a result of which, a part of C sequestered in the biomass will return into the atmosphere. On the other hand this biomass can be used as an energy resource to replace fossil energy which will contribute to GHG and energy use mitigation. In practice, planting even-aged forests is both labour-intensive and regulated at regional levels. Although the introduction of even-aged forest in regions dominated by grassland-based bovine production may not be welcomed by all stakeholders concerned, it certainly has a major potential to contribute to GHG mitigation of worldwide ruminant production. Furthermore, comparing farming practices with identical farm area, i.e. considering alternative land uses on farms, avoids relative changes in impacts according to each functional unit (per unit mass or unit of farm area).

Table 2: Land occupation (m2*year/kg carcass mass) of baseline for standard beef-cattle production and farming-practice scenarios

Land-use type	**Baseline**	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**	**S9**
Permanent pasture	34.0	34.0	29.8	29.3	33.8	29.8	34.4	34.0	33.9	26.3
Temporary pasture	4.7	4.7	4.1	4.1	4.7	4.1	3.7	4.6	4.6	3.6
Arable land on-farm	8.3	8.3	8.3	8.9	8.2	8.2	9.3	8.3	6.4	8.0
Arable land off-farm	1.0	1.0	1.0	1.2	1.0	1.1	0.7	1.0	2.3	1.0
Other land off-farm	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Total	48.2	48.2	43.4	43.7	47.7	43.4	48.3	48.1	47.4	38.9

4. Conclusion

It is difficult to strongly reduce the environmental impacts, and in particular the GHG emissions, of a beef-cattle production system as its impacts result to a very large extent from the suckler cow-calf herd, which offers few options to modify herd management and feeding strategies. Modification of individual farming-practices modestly affected the impacts of the whole beef system; the most promising practice is a radical change in herd management by decreasing calving age from 3 to 2 years. Our results suggest that simultaneous application of several compatible farming-practices can significantly reduce impacts of beef-cattle production. However, our scenario did not consider possible interactions between farming practices. This point should be further explored, an approach combining system experiments and simulation modeling would seem appropriate. The introduction of even-aged forest as an alternative land use in beef cattle farms seems promising and merits further exploration. It illustrates that alternative land use may strongly reduce the climate change impact of the entire production system when comparing farming practices.

5. References

Beauchemin, K.A., Janzen, H.H., Little, S.M., McAllister, T.A., McGinn, S.M., 2011.Mitigation of greenhouse gas emissions from beef production in western Canada - Evaluation using farm-based life cycle assessment. Anim. Feed Sci. Technol. 166-167, 663-677.

Dawson, J.J.C., Smith, P., 2007. Carbon losses from soil and its consequences for land-use management. Sci. Total Environ. 382, 165-190.

Eckard, R.J., Grainger, C., de Klein, C.A.M., 2010. Options for the abatement of methane and nitrous oxide from ruminant production: A review. Livest. Sci. 130, 47-56.

Farrié, J.P., Renon, J., Bourge, C., Gros, J.M., Lahemade, T., Muron, G., Roudier, J., 2008. Conditions et conséquences de la mise en place du vêlage à deux ans dans un troupeau charolais. Renc. Rech. Ruminants 15, 147-150.

INRA, 2007. Alimentation des bovins, ovins et caprins. Besoins des animaux, valeurs des aliments. Tables INRA 2007. Eds. Quae, 307p.

Martin, C., Morgavi, D.P., Doreau, M., 2010. Methane mitigation in ruminants: from the rumen microbes to the farm scale. Animal 4, 351-365.

Nguyen, T.T.H., van der Werf, H.M.G., Eugène, M., Veysset, P., Devun, J., Chesneau, G., Doreau, M., 2012. Effects of type of ration and allocation methods on the environmental impacts of beef-production systems. Livest. Sci. 145, 239-251.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., De Haan, C., 2006. Livestock's Long Shadow: Environmental Issues and Options. FAO, Rome, 390 p.

Steinfeld, H., Mooney, H.A., Schneider, F., Neville, L.E., 2010. Livestock in a Changing Landscape. Drivers, Consequences, and Responses. Washington, D.C.: Island Press.

Vallet, P., Meredieu, C., Seynave, I., Bélouard, T., Dhôte, J.F., 2009. Species substitution for carbon storage: Sessile oak versus Corsican pine in France as a case study. Forest Ecol. Manage. 257, 1314-1323.

An assessment of greenhouse gas emissions and economics of grass based suckler beef production systems

Anne-Marie Clarke[1], Paul Crosson[1,*], Padraig Brennan[2]

[1]Animal & Grassland Research and Innovation Centre, Teagasc, Grange, Dunsany, Co. Meath, Ireland
[2]Bord Bia, Clanwilliam Court, Lower Mount Street, Dublin 2, Ireland
[*] Corresponding author. E-mail: *paul.crosson@teagasc.ie*

ABSTRACT

The objective of this paper was to evaluate the impact of stocking rate (kg of organic nitrogen (N) per hectare (ha)) for steer beef production system on technical and economic performance and greenhouse gas emissions. Carcass output and profitability increased with increasing stocking rate. At a stocking rate of 150 kg organic N/ha, total emissions were lowest per kg of beef carcass (23.4 kg CO_2e /kg beef) and per hectare (9.7 t/CO_2e/ha). The highest output and economic returns were achieved at the highest stocking rate (220 kg organic N/ha). Enteric fermentation was the greatest source of GHG emissions and ranged from 46% to 44% of total emissions with increasing stocking rate. Net margin per tonne of CO_2 e (emission efficiency) increased with increasing stocking rate.

Keywords: LCA, profitability, stocking rate

1. Introduction

In Ireland, agriculture is the largest contributor of greenhouse gas (GHG) emissions accounting for 30.4% of national emissions in 2010 (EPA, 2011a). Agriculture will be required to share the burden of emissions reductions based on the EU target to reduce emissions by 20% by 2020. The primary GHGs from agricultural production are methane (CH_4) and nitrous oxide (N_2O). Of emissions from agriculture, enteric fermentation, manure management and nitrogen additions to agricultural soils account for 47%, 28% and 20% of total emissions, respectively (EPA, 2011b). Life cycle assessment (LCA) is a method that evaluates the environmental impact of products and is regulated by the International Organisation for Standardisation (ISO) standards (ISO, 2006). One of these environmental impact categories is Global Warming Potential which is a relative measure of how much heat a greenhouse gas traps in the atmosphere. The result of an LCA study is reported in terms of relative impact per unit product, known as the functional unit (ISO, 2006).

A number of LCA studies (Nguyen et al., 2010; Peters et al., 2010; Roy et al., 2012) have been carried out on beef cattle production and in most cases the GHG emissions are reported per unit (kilogram) of meat produced in carbon dioxide equivalent, (CO_2e). Roy et al., (2012) reported that GHG emissions of beef (35.6 kg CO_2e/kg-meat) were greater than that of pork (6.9 kg CO_2e/kg-meat) or chicken (6.0 kg CO_2e/kg-meat) in Japan. Peters et al., (2010) examined red meat production in Australia and the results of this study showed that farm level emissions were responsible for the largest proportion of the total burden compared to feedlot and processing stages of its lifecycle. Nguyen et al., (2010) reported that suckler cow calf-beef production system produced 27.3 kg CO_2e/kg-meat (slaughter weight), while dairy calf to beef systems produced on average 17.9 CO_2e/kg-meat. Casey and Holden, (2006), White et al., (2010) and Foley et al., (2011) examined the effect on greenhouse gas emissions of different farm systems. However the effect of increasing stocking rate on greenhouse gas emissions for Irish suckler beef systems has not been investigated.

The objective of this paper was to evaluate the impact of stocking rate (kg of organic nitrogen (N) per hectare (ha)) on steer beef production systems for technical and economic performance and greenhouse gas emissions.

2. Methods

A bioeconomic model of suckler beef production systems, the Grange Beef Systems Model GBSM (Crosson et al., 2006; Crosson, 2008) was used to generate steer beef cattle system scenarios and to quantify the technical and economic performance of these systems. The scenarios were based on spring calving suckler calf to beef research farm systems in Ireland finishing steers at 24 months of age and heifers at 20 months of age. (Drennan and McGee, 2009). The GBSM runs on a monthly time-step and assumes a steady state system over a calendar year. The output data of the GBSM specifies the essential input data for the LCA model that estimates the greenhouse gas emissions associated with grass based suckler beef production systems. This includes animal type and numbers, animal weights, feed intake, feed characteristics (eg DM, DMD, nitrogen concentrations, gross energy), manure application profile, nitrogen fertiliser application (kg/ha), beef produced (kg), lime application (kg/ha) and electricity consumption (kw). This model also provides financial performance data such as farm net margin. Net margin includes direct and overhead costs and all livestock revenues. Land and labour costs are not included as it is assumed that it is a family run farm and all land is owned. The finanical data remains separate from the LCA model.

To model the greenhouse gas emissions, the input data (generated as outputs from GBSM) was imported into the SimaPro software package (Pre Consultants, 2011) in the form of a Microsoft Excel sheet (Microsoft Corporation, 2003). The model simulates production systems involving up to 7 animal groups. The animal groups are cows, male calves and female calves (<1 year old), male and female yearlings (animals between 1 and 2 years of age) and male and female 2 year olds. Direct and indirect greenhouse gas emissions and emissions up stream of the farm gate (those associated with the production of farm inputs) are calculated. Emissions associated with manure management include the storage and spreading of excreta and silage effluent. Excreta deposition at pasture and fertiliser application are included under soils emissions. Sources of indirect N_2O include emissions associated with nitrate leaching and ammonia (NH_3) volatisation while purchased inputs are fertiliser, concentrates and electricity. The production system profile is integrated with emission factors according to IPCC (2006) and all the estimated GHG emissions are converted to their 100-year global warming potential carbon dioxide equivalent (CO_2 e) which on a weight basis, relative to CO_2 was set to a factor of 25 for CH_4 and 298 for N_2O.

The effects of stocking rate on greenhouse gas emissions were modelled. Stocking rate was based on organic nitrogen levels excreted from each animal subject to a maximum organic nitrogen level/ha which is regulated by the EU. This limits the number of animals that can be carried per ha; suckler cows contribute 65kg organic N/head, while animals under 24 months and animals between 24 and 48 months of age contribute 24 and 57 kg of organic N/head, respectively. In the scenarios modelled the animal numbers remained constant and stocking rate was increased by increasing the inorganic fertiliser N application rates on the pasture grazing area of the farm thus, reducing the land area needed to maintain a fixed cow herd and progeny. To take into account the effect of stocking rate on grass utilisation rate (kg grazed grass consumed per kg grass grown) a regression equation was generated based on data available in the literature (MacDonald et al., 2008; French et al., 2010; Baudracco et al., 2011 and Horan et al., 2012).

3. Results

The technical and economic performance of steer suckler beef production systems with stocking rate ranging from 150 to 220 kg of organic Nitrogen (N) per ha is summarised in Table 1. With each incremental increase in stocking rate there was an increase in the application rate of fertiliser N (inorganic). The lower stocking rate of 150 kg N/ha had the lowest output in terms of live weight and carcass weight per hectare. The same stocking rate had the lowest livestock sales, direct costs and net margin per ha per year. The highest output and economic return were achieved at the highest stocking rate (220 kg organic N/ha).

Table 1. Technical and economic performance of steer/heifer production systems of varying stocking rate

Organic Nitrogen (kg/ha)	**150**	**160**	**170**	**180**	**190**	**200**	**210**	**220**
Farm size (ha)	49.9	46.7	44.1	41.6	39.4	37.5	35.6	34.0
Fertiliser nitrogen (kg/ha)	119	133	146	159	173	188	202	216
Live weight output (kg/ha)	746	797	844	895	945	994	1045	1095
Carcass weight output (kg/ha)	395	421	447	473	500	526	553	579
Livestock sales (€/ha)	1475	1598	1714	1839	1961	2081	2206	2328
Direct costs (€/ha)	703	750	794	841	888	935	983	1031
Fixed costs (€/ha)	410	438	464	493	521	548	577	604
Net margin (€/ha)	363	411	456	504	552	598	646	693

At a stocking rate of 150 kg organic N/ha the total emissions were lowest when expressed per kg of beef carcass and per hectare (Table 2). The direct and indirect emissions when expressed per kg of carcass and per hectare increased linearly with increasing stocking rate. The greater the stocking rate, the lower the contribution of enteric fermentation of the total GHG emissions. As stocking rate increased the relative contribution to total GHG emissions was also lower for manure management and diesel use, whereas the contribution was higher for emissions associated with soils, indirect N_2O and purchased inputs. This is due to the greater amount of inorganic fertiliser N applied, resulting in higher soils emissions, N leaching, ammonia volatilisation and fertiliser production emissions. Net margin per tonne of CO_2e was lower at a stocking rate of 150 kg organic N/ha (€39.5/t CO_2e) compared to the stocking rate of 220 kg organic N/ha (€49.5/t CO_2e).

Table 2. Effect of stocking on greenhouse gas emissions per kg of beef carcass and per hectare and the source and percentage of the total contribution to the total greenhouse gas emissions for steer/heifer production systems

Organic Nitrogen (kg/ha)	**150**	**160**	**170**	**180**	**190**	**200**	**210**	**220**
Total emissions per kg beef carcass (kg)	22.2	22.4	22.5	22.6	22.7	22.9	23.0	23.1
Total emissions per hectare (t)	9.2	9.9	10.5	11.2	11.9	12.6	13.3	14.0
Enteric fermentation (%)	48.2	47.9	47.6	47.4	47.2	46.9	46.7	46.5
Manure management (%)	12.6	12.5	12.4	12.4	12.3	12.2	12.2	12.1
Soils (%)	22.6	22.7	22.8	22.9	23.0	23.1	23.2	23.3
Indirect nitrous oxide (%)	4.0	4.0	4.0	3.9	3.9	3.9	3.9	3.9
Diesel use (%)	1.1	1.0	1.0	1.0	0.9	0.9	0.9	0.9
Purchased inputs (%)	11.5	11.9	12.2	12.4	12.6	12.9	13.1	13.3

4. Discussion

In this study, a partial LCA model was developed to evaluate GHG emissions from Irish steer beef production systems. Ireland exports approximately 90% of its beef output (Breen et al., 2010) and due to the increased awareness of carbon footprint by society and consumer preference for 'low carbon-footprint' food (Schulte et al., 2011), it is important to quantify the carbon footprint of agricultural products. Nitrous oxide emissions are generated by the application of organic and inorganic fertilisers and the deposition of faecal and urine nitrogen by livestock. The quantity of CH_4 emissions are determined primarily by the numbers of livestock and its main source is from animal digestion (enteric fermentation), followed by manure management where liquid manure storage systems predominate (Crosson et al., 2011). Emissions per kg of beef carcass in this study were similar to other studies of Irish beef production systems (Casey and Holden, 2006; Foley et al., 2011). The marginally lower values in the study of Foley et al., (2011) are due to the lower fertiliser N applied in their study compared to the present study and differences in emission factors applied.

The developed model was applied to examine the effect of stocking rate on greenhouse gas emissions. Stocking rate is the most important factor affecting production and profitability on farms (Fales et al., 1995; Crosson and McGee, 2011; Horan et al., 2012). Higher stocking rates due to increased fertiliser N/ha applied in this study lead to greater GHG emissions. White et al., (2010) reported simalar findings on New Zealand beef and sheep farms. Neufeldt et al., (2006) found that greenhouse gas emissions on a per hectare basis were highly correlated ($R^2 = 0.85$) to the stocking rates on farms in Germany and suggested that stocking rates are good indicators of overall agricultural GHG emissions. Emissions efficiency in terms of net margin per tonne of CO_2e suggests that there is lower profitability per unit of GHG emission at lower stocking rates.

As a result of EU emissions reductions targets, national governments have must establish respective targets for their own jurisdictions. However, this analysis has shown that production systems which minimize emissions on an area and product based basis also result in lower profitability. Furthermore, the analysis has shown that profitability can be increased substantially by operating at higher stocking rates with only modest increases in GHG emissions per kg beef carcass albeit with substantial increases in GHG emissions per hectare. It is apparent that concomitantly achieving GHG efficient and profitable production systems are possible provided that emissions intensity (GHG emissions per unit product) is the metric used.

5. Conclusion

The developed model is a useful tool in evaluating GHG emissions from beef farms in Ireland This study shows that increasing stocking rate via increased fertiliser nitrogen application rates and higher grass utilisation rates lead to increased profitability on beef farms with only modest increases in GHG emissions.

6. References

Baudracco, J., Lopez-Villalobos, N., Romero, L.A., Scandolo, D., Maceil, M., Comeron, E.A., Holmes, C.W. & Barry, T. N. (2011). Effects of stocking rate on pasture production, milk production and reproduction of supplemented crossbred Holstein-Jersey dairy cows grazing lucerne pasture. Anim. Feed Sci. and Tech. 168, 131-143.

Breen, J.P., Donnellan, T. & Westhoff, P. (2010). Food for Thought: EU Climate Change Policy Presents New Challenges for Agriculture. EuroChoices 9, 24-29.

Casey, J.W. and Holden, N.M. (2006). Greenhouse gas emission from conventional, agri-environmental scheme, and organic Irish suckler-beef units. J. Env. Qual. 35, 231-239.

Crosson, P., O'Kiely, P., O'Mara, F.P. & Wallace, M. (2006). The development of a mathematical model to investigate Irish beef production systems. Agric. Sys. 89, 349-370.

Crosson, P. (2008). The impact of cow genotype on the profitability of grassland-based suckler beef production in Ireland. In: Proceedings of the 22nd Annual Meeting of the European Grassland Federation, Uppsala, Sweden, 9-12 June, p 771.

Crosson, P. & McGee, M. (2011). Suckler beef production in Ireland: challenges and opportunities. Teagasc, National Beef Conference, 5th April, Kilkenny, Ireland. http://www.teagasc.ie/publications/2011/475/PaulCrossonMarkMcGee.pdf

Crosson, P., Shalloo, L., O'Brien, D., Lanigan, G.J., Foley, P.A., & Boland, T.M. (2011). A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. Ani. Feed. Sci. and Tech. 166-167, 29-45.

Drennan, M.J., and McGee, M. (2009). Performance of spring-calving beef suckler cows and their progeny to slaughter on intensive and extensive grassland management systems. Livest. Sci.120, 1-12.

EPA, 2011a. Ireland's Greenhouse Gas emissions in 2010. 9/11/2011. Environmental Protection Agency, Johnstown Castle Estate, Co. Wexford, Ireland.

EPA, 2011b. Ireland's Greenhouse Gas Emissions Projections 2010-2020. Environmental Protection Agency, Johnstown Castle Estate, Co. Wexford, Ireland.

Fales, S.L., Muller, L.D., Ford, S.A., O'Sullivan, M., Hoover, R.J., Holden, L.A., Lanyon, L.E. & Buckmaster, D.R. (1995). Stocking rate affects production and profitability in a rotationally grazed pasture system. J. Prod. Agric. 8, 88-96.

Foley, P.A., Crosson, P., Lovett, D.K., Boland, T.M., O'Mara, F.P., Kenny, D.A. (2011). Whole-farm systems modelling of greenhouse gas emissions from pastoral suckler beef cow production systems. Agric. Ecosyst. Env. 142, 222-230.

French, P., O'Donovan, M., Horan, B. & Shalloo, L. (2010). What is the optimum stocking rate to maximize profitability of grass based milk production? Ir. Grassl. Ass. J. 44, 16-22.

Horan, B., McCarthy, B. & Brennan A. (2012). The effect of stocking rate and calving date on the characteristics of milk production systems post milk quotas. http://www.agresearch.teagasc.ie/moorepark/Articles/Curtins%20A%20Review_V2.pdf

Intergovernmental Panel on Climate Change (IPCC), 2006. IPCC guidelines for National Greenhouse Gas Inventories, In: Eggleston, H., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), The National Greenhouse Gas Inventories Programme, Intergovernmental Panel on Climate Change. IGES, Japan.

International Organisation of Standardisation (ISO), (2006). Environment Management – Life Cycle Assessment – Principles and Framework. EN ISO 14040.

MacDonald, K.A., Penno, J.W., Lancaster, J.A.S. & Roche J.R. (2008). Effect of stocking rate on pasture production, milk production and reproduction of dairy cows in pasture-bases systems. J. Dairy Sci. 91, 2151-2163.

Microsoft Corporation, 2003. http://office.microsoft.com/en-us/excel/ 1 Microsoft Way, Redmond, Washington, USA.

Neufeldt, H. Schafer, M., Angenendt, E., Li, C., Kaltschmitt, M. & Zeddies, J. (2006). Disaggregated greenhouse gasd emission inventories form agriculture via a coupled economic-ecosystem model. Agric., Ecosyst. Env. 112, 233-240.

Nguyen T.L.T., Hermansen, J.E. & Mogensen L. (2010). Environmental consequences of different beef production systems in the EU. J. Clean. Prod. 18, 756-766.

Peters G.M., Rowley, H.V., Wiedemann, S., Tucker, R., Short, M.D. & Schulz M. (2010). Red meat production in Australia: life cycle assessment and comparison with overseas studies. Env. Sci. and Tech.44, 1327-1332.

PRé Consultants (2011). SimaPro 7.2 method. In: Database Manual. Pré Consultants B.V., Amersfoort, The Netherlands.

Roy, P., Orikasa, T., Thammawong, M., Nakamura, N., Xu, Q., Shiina, T. (2012). Life cycle of meats: an opportunity to abate the greenhouse gas emission from meat industry in Japan. J. Env. Man. 93, 218-224.

Schulte R.P.O., Lanigan, G, Gibson, M. (2011). Irish Agriculture, Greenhouse Gas Emissions and Climate Change: Opportunities, obstacles and proposed solutions. Teagasc, Oak Park, Carlow.

White, T.A., Snow, V.O. & McG King, W. (2010). Intensification of New Zealand beef farming systems. Agric. Syst. 103, 21–35.

Could cultured meat reduce environmental impact of agriculture in Europe?

Hanna L. Tuomisto[1,*], Avijit G. Roy[2]

[1] European Commission, Joint Research Centre (JRC), Institute for Environment and Sustainability, Via Enrico Fermi 2749, 21027 Ispra, Italy
[2] Clore Laboratory, University of Buckingham, Buckingham, MK18 1EG, UK
[*] Corresponding author. E-mail: *hanna.tuomisto@jrc.ec.europa.eu*

ABSTRACT
This paper assesses the potential of cultured meat to reduce environmental impacts of livestock production in Europe. Cultured meat (i.e. in vitro meat or lab-grown meat) is produced by cultivating livestock muscle cells in a growth media. The environmental impacts of hypothetical large-scale production of cultured meat were compared to the impacts of livestock production in the EU-27. The results showed that if all meat produced in the EU-27 was replaced by cultured meat, the GHG emissions, land use and water use would be reduced by two orders of magnitude compared to current meat production practices. When the opportunity costs of land use were included, the environmental benefits were even higher. More research and development is required before the product can be commercialised. Further effort is needed to gain public acceptance for this technology.

Keywords: in vitro meat, greenhouse gas emissions, land use, life cycle assessment, water footprint

1. Introduction

Livestock production is one of the major contributors to global environmental degradation. The contribution of livestock production to greenhouse gas (GHG) emissions in the European Union (EU) has been estimated to account for 9.1% of total EU emissions or 12.8% when land use and land use change emissions are taken into account (Weiss and Leip, 2012). Furthermore, livestock production accounts for a large share of land and water use and is the main contributor to the eutrophication of water ways and loss of biodiversity. The main strategies to reduce the negative environmental impact of livestock production include changes in feedstock, improvements of manure management and breeding animals with higher feed-to-food conversion ratios. To achieve more substantial improvements, new approaches to meat production will be required, unless vast majority of people adopt purely vegetarian diets. However, the current trends show that the global meat consumption will increase rather than decrease by 2050 (FAO, 2006).

A novel alternative to conventionally produced meat is to cultivate animal muscle cells *in vitro* without growing the whole animals. Currently, production of *in vitro* meat, also known as cultured meat, is in the research stage, but it has been estimated that the commercial production could start within a decade. It has been shown that the potential environmental impacts of cultured meat are substantially lower than those of meat produced in Europe (Tuomisto and Teixeira de Mattos, 2011). When cyanobacteria hydrolysate is used as the main nutrient and energy source for muscle cell growth, life-cycle-assessment-based GHG emissions, land use, and water use are 78-96%, 99%, and 82-96% lower, respectively, per tonne of meat compared to those of conventionally produced European meat. Energy use for cultured meat production was 38% higher than that of poultry, but lower than those of beef, sheep or pork.

This paper extends previous research by demonstrating the total potential GHG emission reductions and changes in land, water and energy use requirements in the EU-27 when conventional meat production is replaced by cultured meat. The environmental benefits resulting from alternative use of land released from agriculture are also considered. Furthermore, the impacts of cultured meat are compared with plant-based and livestock-based protein sources. Finally, the uncertainties related to the potential of cultured-meat-mediated reductions in environmental impacts of meat production in the EU are discussed.

2. Methods

2.1. Cultured meat production

The data for the environmental impacts of cultured meat production came from Tuomisto and Teixeira de Mattos (2011) (Table 1). The cultured meat production process used in the study is briefly described here. This process produces minced-beef type of product as the production technologies for steak type of products are still under development. Cyanobacteria hydrolysate is used as the source of nutrients and energy for muscle cell production. Cyanobacteria are assumed to be cultivated in an open pond made of concrete. After harvesting, the cyanobacteria biomass is sterilised and hydrolysed to break down the cells. The stem cells are taken from an animal embryo. Embryonic stem cells have almost infinite self-renewal capacity and theoretically a small number of these cells would be sufficient to feed the world. However, the differentiated product of these stem cells, such as muscle cells, has a limited proliferation period. Embryonic stem cells can produce more than 1000 kg of cultured meat, and therefore, the impacts related to the production of the stem cells are not included in this study. Engineered *Escherichia coli* bacteria are used for the production of specific growth factors that induce the stem cells to differentiate into muscle cells. The muscle cells are grown in a bioreactor on a medium composed of the cyanobacteria hydrolysate supplemented with growth factors and vitamins. The system boundaries cover the major processes from input production up to the factory gate, including production of input materials and fuels, production of the feedstock, and growth of muscle cells. The production of growth factors and vitamins are not included in the study as the quantities needed are small (under 0.1% of the DM weight of the media), and therefore, the environmental impacts are negligible. The impacts of the waste management are not allocated to cultured meat as it is assumed that the waste will be used for other commercial processes.

Table 1. Environmental impacts of cultured meat production per 1000 kg of cultured meat (Tuomisto and Teixeira de Mattos, 2011) (in italics the breakdown of the impacts of the main category)

Production stage	Primary energy GJ	GHG kg CO_2-eq	Indirect water use m^3	Direct water use m^3	Land use m^2
CYANOB. CULTIVATION	8.1	611.0	8.6	441.9	232.0
Fertiliser production	*3.2*	*205.9*	*3.4*		
Cultivation of cyanobacteria	*3.7*	*303.2*	*3.9*	*441.9*	*232.0*
Harvesting of cyanobacteria	*0.1*	*10.4*	*0.1*		
Construction and maintenance of the cyanobacteria production plant	*1.1*	*91.4*	*1.2*		
BIOMASS TRANSPORTATION	0.4	25.9	0.4		
STERILISATION	2.9	143.8	7.6	7.2	
MUSCLE CELL CULTIVATION	21.2	1121.8	56.1		
Steel production	*1.0*	*107.9*	*2.6*		
Aeration	*7.9*	*395.6*	*20.9*		
Rotation	*12.3*	*618.2*	*32.6*		
GRAND TOTAL	**32.5**	**1902.4**	**72.6**	**449.1**	**232.0**

2.2. Environmental impacts of livestock production in the European Union

Data about meat production quantities and GHG emissions in the EU-27 were based on Weiss and Leip (2012) (Table 2). GHG emissions were considered both with and without land use and land use change (LULUC) emissions. Weiss and Leip (2012) included three scenarios for estimating the LULUC emissions, and in this study the Scenario II was used, which represents the most likely mix of land use change probabilities. As the production quantities are reported in tonnes of carcass deadweight, the amount of cultured meat needed to replace the conventionally produced meat was calculated by using conversion factors that convert the carcass deadweight to edible meat (Table 3).

The water footprint data of conventionally produced meat was based on datasets of the Water Footprint Network. The total water footprint of meat production in the EU-27 was calculated based on the data about water footprint of each meat type (m^3/t) produced in each EU-27 country (Mekonnen and Hoekstra, 2012) and that was multiplied by the meat production quantities in each country reported by Weiss and Leip (2012).

Table 2. Production quantities and greenhouse gas (GHG) emissions (with and without land use and land use change (LULUC)) of conventionally produced meat in the EU-27 (Weiss and Leip 2012)

Impact	Beef	Sheep	Pork	Poultry	Total
Production (1000 t)	8146	1014	22384	11091	42635
GHG emissions (1000 t CO_2-eq)					
Without LULUC	156814	19920	97431	27405	301570
With LULUC	191000	24425	164780	54360	

Table 3. Conversion factors used for estimating edible meat production quantities.

	% of carcass dead weight			
	Beef[a]	**Sheep**[a]	**Pork**[a]	**Poultry**[b]
Edible meat	38.56	45.55	44.55	51.47

[a] (Garnett, 2007)
[b] (Igri and Ausaji, 2006)

2.3 Opportunity costs of land use

Opportunity costs of land use were estimated by assuming that the land area released from meat production was used for bioenergy production or forestry. It was assumed that grassland was used for forest and arable land for bioenergy crops. As the aim was not to estimate the bioenergy production potential in the EU, but only to demonstrate opportunity costs of land use, generic data for the entire EU was used. Energy production and GHG-emission savings by forest and bioenergy crops were from Tuomisto et al., (2012a), who estimated the yearly net energy production of woodland as 92.6 GJ/ha and its GHG mitigation as 10.2 t CO_2-eq/ha, and the yearly net energy production of a bioenergy crop, *Miscanthus*, as 159.2 GJ/ha and its net GHG mitigation as 15.4 t CO_2-eq/ha.

The calculations of the energy yield and GHG emission mitigation from woodland were based on the following assumptions. It was assumed that 15 m^3/ha/yr wood was harvested and 0.57 t wood chips (75% DM), 2.3 t composite board (90% DM) and 0.39 t sawn timber was produced from the harvested wood. For the wood chips, a heating value of 17.8 MJ/kg was used and the boiler was assumed to operate at 90% efficiency. Life cycle energy use (345 ± 36 MJ/t) and GHG emissions (21 ± 2 kg CO_2-eq/t) were associated with the harvesting of wood (75% DM). It was assumed that the wood chips replaced oil used for heating, and the composite board and timber replaced steel. Production of 1 kg stainless steel requires 30.6 MJ primary energy and emits 3.38 kg CO_2-eq. It was assumed that soil carbon stocks increase during the first 100 years after planting by an average of 0.1 t C/ha/yr (equals 0.37 t CO_2-eq/ha/yr). To avoid double-counting, carbon mitigation by aboveground vegetation was not included, as the wood was harvested and burned or used as materials which will ultimately decompose.

It was assumed that *Miscanthus* bioenergy crop was planted on arable land. An average yield of 10.4 t DM/ha through the whole growing cycle was used in the base calculations. The energy yield of *Miscanthus* was calculated by using the lower heating value of 17.6 MJ/kg DM and a 90%-efficient boiler. The energy inputs required for production of *Miscanthus* through the whole growing cycle was 9.26 GJ/ha/yr.

2.4 Data used for product comparisons

Data for energy use, land use and GWP of crops, livestock products and quorn (meat substitute that is produced from filamentous fungus Fusarium venenatum and egg albumin) was based on Tuomisto (2010). Data for energy use, land use and GWP of tofu and salmon was from Blonk (2008). Data for water use of crops and livestock products came from Mekonnen and Hoekstra (2011) and Mekonnen and Hoekstra (2012).

3. Results

The results showed that if all meat produced in the EU-27 was replaced by cultured meat, the GHG emissions would be reduced by 98.8%, land use 99.7% and water use 94% compared to current meat production practices (Table 4). When the opportunity costs of land use are taken into account the cultured meat system produced 21.1 EJ net energy, which is about 30% of the gross inland primary energy consumption in EU-27 in 2009. The GHG emission mitigation achieved by using the cultured meat system corresponds to 43% of the annual GHG emissions in the EU-27. Also in the EU-27, total water use would be reduced by 21%, and 38% of the total land area would be released from livestock production.

When the environmental impacts of cultured meat were compared with meat products, crops, tofu and quorn per unit of product, protein and energy (Table 5), it was found that cultured meat had lower land use requirements than any other product, regardless of functional unit, except spirulina. The energy use and GHG emissions of cultured meat were higher than those of crops. Cultured meat had also higher energy use than most livestock-based products. GHG emissions of cultured meat were almost always lower than those of livestock-based products and meat substitutes.

Table 4. Estimated impacts for the entire EU-27 of current meat production practices and reduction achieved by using cultured meat technology with and without taking into account land use and land use change (LULUC) emissions (the former includes opportunity costs of land use).

Impact	Unit	Current meat	Cultured meat	Reduction quantity	%
GHG without LULUC	1000 t CO_2-eq	301570	3669	297900	99
GHG with LULUC	1000 t CO_2-eq	434565	-2183663	2618200	603
Water use	1000 m^3	164250	10060	154200	94
Land use	km^2	1650000	4474	1645500	100

Table 5. Land use, energy use, greenhouse gas emissions (GHG), and water use of plant, livestock and meat substitute products per functional unit of mass, protein, or energy in the product.

	per edible (t)				per protein (t)				per energy unit (TJ)			
Product	Land ha	Energy TJ	GHG t CO_2-eq	Water 1000m^3	Land ha	Energy TJ	GHG t CO_2-eq	Water 1000m^3	Land ha	Energy TJ	GHG t CO_2-eq	Water 1000m^3
wheat	0.14	2.5	0.8	3.6	1.1	19	6	28.2	11	0.2	62	27.6
soybean	0.42	3.0	1.3	2.2	1.2	8	4	6.1	27	0.2	84	140.9
maize	0.14	2.4	0.7	1.4	1.1	19	5	10.8	10	0.2	44	92.8
field bean	0.30	2.0	1.0	3.0	1.4	9	5	13.6	22	0.1	74	219.5
spirulina	0.02	10.1	0.8	0.5	0.0	16	1	0.7	2	0.7	54	29.3
beef	4.34	52.5	29.8	11.8	19.3	233	132	99.2	679	8.2	4665	3491.6
pork	0.99	22.3	8.5	4.9	4.5	102	39	29.8	108	2.4	928	711.1
sheep	2.91	48.7	36.9	8.3	14.5	243	184	87.9	297	5.0	3766	1796.3
poultry	0.94	17.6	6.7	3.8	4.2	79	30	25.2	103	1.9	733	612.5
salmon	-	25.4	1.8	-	-	151	11	-	-	3.7	260	-
eggs	0.55	11.8	4.6	3.4	4.4	94	37	30.1	105	2.2	878	716.4
milk	0.12	2.5	1.1	1.1	3.7	79	33	33.6	45	1.0	400	406.0
cheese	0.72	20.0	8.8	5.2	2.8	78	34	20.2	42	1.2	510	299.0
quorn	0.17	38.0	2.3	-	1.0	233	14	-	38	8.5	514	-
tofu	0.30	15.6	2.0	-	3.8	200	26	-	86	4.5	575	-
cultured meat	0.02	31.7	1.9	0.5	0.1	166	10	2.7	5	7.1	423	116.5

4. Discussion

Many technological and social issues have to be resolved before cultured meat can contribute to the reduction of environmental impacts of food production in the EU. Currently, only small quantities of cultured meat have been produced in research laboratories, and more research is required before the production can be scaled up to commercial levels. The main challenges for scaling up the production include development of growth media, optimising the production conditions and making the whole process financially feasible.

As the technology for producing cultured meat in large-scale production plants is currently not well defined, there are many uncertainties about the data of the environmental impacts of cultured meat production presented in this paper. An uncertainty analysis of the environmental impacts of cultured meat production is presented in Tuomisto et al., (2011). More information about the commercial scale cultured meat production system will assist with generating more accurate environmental impact estimates. This study did not take into account the production of scaffolds on which the cells are cultivated. These scaffolds could be made of edible materials or alternatively the cells could be harvested on the surface of the scaffolds. Furthermore, this study did not consider the production of fat cells, and the mechanical and/or electric stretching that would be required for exercising the muscle cells.

Nonetheless, cultured meat would provide substantial environmental benefits, as its land use, GHG emission and water use impacts are only a fraction of those of conventionally produced meat. In particular, when opportunity costs of land use are taken into account, cultured meat could help reduce most environmental impacts of livestock production if the land released from livestock production were used for providing environmental services. However, it has to be noted that the analysis presented in this paper did not take into account the fact that if majority of meat was produced by using cultured meat technology, the co-products of meat production, such as leather and wool, should be produced by alternative ways.

Cultured meat production could also have potential benefits for wildlife conservation for two main reasons: i) it reduces pressure for converting natural habitats to agricultural land, and ii) it provides an alternative way of producing meat from endangered and rare species that are currently over-hunted or –fished for food. However, large-scale replacement of conventional meat production by cultured meat production may have some negative impacts on rural biodiversity due to the reduction in need for grasslands and pastures. In some hilly areas, livestock also has an important role in maintaining the open landscapes that are preferred over forested hills. The overall value of the biodiversity impacts would depend on the indicators used. The conversion of grasslands into forest and arable-lands to *Miscanthus* or other bioenergy crops might benefit some species whilst some others may suffer. *Miscanthus* and wood land were used in this study to demonstrate the opportunity costs of land use, but those options would not be the optimal for each location.

Even though eutrophication impacts were not considered in this study, it can be hypothesised that cultured meat production has substantially lower nutrient losses to waterways compared to conventionally produced meat; since wastewaters from cyanobacteria production can be more efficiently controlled compared to run-offs from agricultural fields.

Large-scale production of cultured meat also requires sufficient demand for the product. This would require consumers to accept cultured meat as a substitute to conventionally produced meat. Therefore, taste and texture should be close to conventionally produced meat, and affordability should be taken into account. Taste of meat is influenced by many factors, such as the source of muscle cell, fat content, texture, colour, and shape. Controlled production conditions may ease the addition, removal or modification of any feature in the final product based on consumer preferences. Fat can be added later, and the content and quality of fatty acids can be controlled. The first cultured meat products will most likely be processed products, such as sausages and/or hamburgers. The development of a steak structure will require more research. The costs of cultured meat production based on the current approach have not yet been quantified.

In this study, it was assumed that 100% of the meat production in the EU would be replaced by cultured meat. However, this choice was made mainly to demonstrate the potential of cultured meat. Like other technologies on the market, cultured meat production and adoption will likely follow the ‘Technology S-curve’ (Sood, 2010). This entails that initially current methods of meat production will outcompete cultured meat on both aspect but within a very short time based on the efficiency categorised in this paper cultured meat will outcompete other methods of production. We also envision that consumer adoption of cultured meat will probably increase as it becomes more available and marketable.

5. Conclusion

Regardless of the uncertainty of the study, the potential environmental benefits of replacing livestock production with cultured meat are substantial. However, more research and development are needed before cultured meat products can be commercialised. Once more knowledge about the processes of the commercial-scale cultured meat production becomes available more detailed estimates about the environmental impacts of cultured meat production can be provided. In order to gain the environmental benefits that cultured meat can offer, the wider acceptance of cultured meat among consumers is required. This could be achieved by improving the public understanding of science.

6. References

Blonk, H., Kool, A., Luske, B., 2008. Milieueffecten van Nederlandse consumptie van eiwitrijke producten. Gevolgen van vervanging van dierlijke eiwitten anno 2008. Blonk Milieu Advies, Gouda.

FAO, 2006. Livestock’s long shadow –environmental issues and options. Food and Agricultural Organization of the United Nations, Rome, p. 390.

Garnett, T., 2007. Meat and dairy production & consumption -Exploring the livestock sector's contribution to the UK's greenhouse gas emissions and assessing what less greenhouse gas intensive systems of production and consumption might look like. Working paper produced a part of the work of the Food Climate Research Network, University of Surrey.

Igri, O., Ausaji, A., 2006. Carcass characteristics and meat quality of broilers fed periwinkle flesh (Pachymelania aurita), palm kernel cake and oil-based rations. Journal of the Science of Food and Agriculture 24, 1021-1030.

Mekonnen, M.M., Hoekstra, A.Y., 2011. The green, blue and grey water footprint of crops and derivered crop products. Hydrology and Earth System Sciences 15, 1577-1600.

Mekonnen, M.M., Hoekstra, A.Y., 2012. A Global Assessment of the Water Footprint of Farm Animal Products. Ecosystems 15, 401-415.

Sood, A., 2010. Technology S-Curve. Wiley International Encyclopedia of Marketing. John Wiley & Sons, Ltd.

Tuomisto, H.L., 2010. Food Security and Protein Supply -Cultured meat a solution? Aspects of Applied Biology 102, 99-104.

Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012a. Comparing energy balances, greenhouse gas balances and biodiversity impacts of contrasting farming systems with alternative land uses. Agricultural Systems 108, 42-49.

Tuomisto, H.L., Teixeira de Mattos, M.J., 2011. Environmental Impacts of Cultured Meat Production. Environmental Science & Technology 45, 6117-6123.

Weiss, F., Leip, A., 2012. Greenhouse gas emissions from the EU livestock sector: A life cycle assessment carried out with the CAPRI model. Agriculture, Ecosystems & Environment 149, 124-134.

The environmental effects of seasonal food purchase: a case study

Chris Foster[1*], Mark Holmes[2], Jeremy Wiltshire[2], Sarah Wynn[2]

[1] EuGeos Limited, 1 Park Street, Macclesfield, UK
[2] ADAS, Pendeford House, Pendeford Business Park, Wobaston Road, Pendeford, Wolverhampton, WV9 5AP, UK
[*] Corresponding author. E-mail: *chrisf@eugeos.co.uk*

ABSTRACT

The environmental effects of seasonal food supply and purchase patterns have been explored using LCA, in conjunction with research into consumer perceptions of seasonal food. An LCA of raspberries supplied in the UK at different times of the year is reported here. The study was designed specifically to examine how impacts changed from one date of supply to another. Supply at different times of the year draws on different production systems and locations. Despite that, the results reveal relatively small differences, except in the case of the water footprint measures. The results are very sensitive to fruit yield. So in this case, yield and agricultural practice appear stronger drivers of the environmental burden of food production than is time of supply. In such situations a strong focus on "seasonality" in sustainable food provisioning is unlikely to deliver large environmental benefits.

Keywords: seasonal, fruit, supply pattern, raspberry, LCA

1. Introduction

As urbanisation progressed in the second half of the 20th Century and the agricultural workforce shrank, so Western European citizens disengaged from food production, losing their connection with its seasonal patterns. Recently, interest in seasonal foods has been resurgent; Dibb et al., (2006) state that 2/3 of people in the UK are now "taking steps to buy seasonally". This trend has various drivers but - as Dibb et al.'s title suggests - some see implications for the environment in it. In line with this, governmental advice on "sustainable diet" often advocates consumption of seasonal food. Seeking additional evidence relevant to such recommendations, the UK's Department of the Environment, Food and Rural Affairs commissioned a research project encompassing investigation of consumers' perceptions and attitudes to "seasonality" alongside exploration of the environmental implications of seasonal food purchasing patterns.

1.1. "Seasonal" Food

A review of literature and consumer research demonstrated that clearly identifying seasonal food is in fact quite difficult. Few commentators take the trouble to define the term "seasonal", while consumer research found that UK consumers "have only a vague definition of seasonal food". In essence "very different definitions and perceptions of what is seasonal are applied by different parties" (Brooks et al., 2011). To inform the project, we used two working definitions of seasonal. The first was a production-oriented or "global" definition: *food that is outdoor grown or produced during the natural growing/production period for the country or region where it is produced. It need not necessarily be consumed locally to where it is grown*[4]. The second was a consumer-oriented, more "local"definition: *food that is produced and consumed in the same climatic zone, e.g. UK, without high energy use for climate modification such as heated glasshouses or high energy use cold storage*. The LCA element of the research concerned a number of food items which were expected to meet one or the other (or indeed both) of these definitions.

1.2. Food, seasonality and the environment

In Brooks et al., (2011) we briefly outline how the timing of agricultural activities in any one place can change the effects of those activities on the wider environment impacts, even if the activities remain the same. But as food production for supply in a certain place is shifted further away in time from the "natural" time of production there, so one or both of two things occurs: either the nature of the producing activity changes (e.g. through the introduction of crop protection) or the place of production changes. Furthermore, preservation and storage allow the time of production and the time of supply to be separated, introducing further flexibility into the supply system. Finally of course, consumers also have access to preservation and storage, so can separate the time of supply from the moment of consumption.

Each of these adjustments changes the interaction between the food system and the natural environment surrounding it: different production systems for the same basic foodstuff have different yields and require different inputs, almost all preservation techniques require energy inputs, as does cold storage. The fact that

[4] This was originally suggested in Defra's project specification

these adjustments can be made at different points in a generic food production-consumption system is a strong indicator that life cycle assessment will be an effective tool to explore their environmental implications. In this project, certain foods were selected as case studies through which the environmental implications of different adjustments could be explored. The foods were lamb, potatoes, raspberry, strawberry and two exotic fruits: melon and pineapple. This paper draws on the raspberry case study to illustrate how environmental impacts vary across the year for one food consumed in the UK. The project did not explore storage and preservation by the consumer, so considered only the effect of changing the times of production and supply in the system as far as delivery to the food retailer.

2. Methods

2.1 Scope

In this research, we equate (reflecting mainstream economics and consumer data) consumption with purchase, and purchase with supply. This embodies a simplification: it is possible that consumers store foods for extended periods after purchasing them. The effect of this, if it occurs, was not considered in the LCA; it would make food consumption less "seasonal" than statistics would lead us to believe it is. Some of the volume captured by this data is supplied to commercial buyers rather than final consumers, of course. This is still purchase, however, and there seems to be no reason to try to exclude it.

Fig. 1 shows UK supply of raspberries changes through the year in volume and by source (data compiled from UK production[5] and import[6] statistics). There is scarcely competition between local production - certainly "seasonal" according to the consumer-oriented definition above - and imports, which are "seasonal" only according to the global definition; rather imports complement local produce in an overall supply pattern. To gain some insight into the environmental implications of this supply pattern an LCA of raspberries was conducted. This covered 3 functional units:

A. 1kg raspberries delivered fresh to a supermarket distribution centre (RDC) in May
B. 1kg raspberries delivered fresh to a supermarket RDC in July
C. 1kg raspberries delivered frozen to a supermarket RDC in November

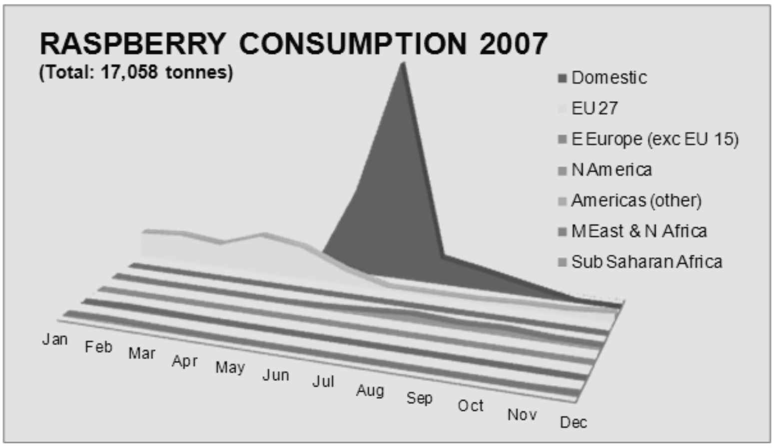


Figure 1. UK Raspberry Supply 2007. Sources: Defra Horticultural Statistics, UK Trade Statistics

Reflecting the sources and production techniques relevant to them, an appropriate product system for A involved production in Southern Spain. In this system, raspberries are grown on an annual basis in fields that are covered for the whole season with "Spanish" tunnels. The ground is prepared each year and beds then formed. The planting material (canes) is produced in the UK or Netherlands and transported to the producing site in chilled lorries. The canes are kept for 3-4 weeks in a cold store prior to planting, then planted directly through plastic into the pre-prepared beds. Fertilisers are applied through drip irrigation: nitrogen as ammonium nitrate or potassium nitrate. Irrigation draws water from an aquifer. Harvesting occurs by hand with the fruit then transported directly to the packhouse; yield is 8 tonnes per ha. In the packhouse the fruit is graded and cleaned, before being placed into punnets with a plastic film lid. Punnets are cooled in a cold store prior to export in refrigerated trucks which travel 2,500km from Spain to the UK.

[5] Department of Environment, Food and Rural Affairs Horticultural Statistics: www.defra.gov.uk/statistics/foodfarm/landuselivestock/bhs/
[6] HM Revenue & Customs Trade Statistics: https://www.uktradeinfo.com/Pages/Home.aspx

For B and C production was in the east of England, on canes grown for seven years in fields covered with polytunnels during fruiting. In this case in the first year the ground is prepared, beds formed and soil sterilised (e.g. with chloropicrin). The canes are produced on a separate farm, cold stored prior to planting, then planted directly through plastic into the pre-prepared beds. On an annual basis (for seven crops) fertilisers are applied through drip irrigation including nitrogen fertiliser as ammonium nitrate or potassium nitrate. Irrigation uses water from an aquifer. Fruit is hand-harvested then transported directly to the packhouse. Yield averages 12t/ha per year for the seven cropping years. In the packhouse the fruit is graded and cleaned, before being placed into punnets with a plastic film lid. Punnets are cooled in a cold store prior to distribution in refrigerated trucks. At the end of the season the soil between the beds is pulverised to reduce compaction. At the end of the seven years the crop is grubbed out, the ground sub-soiled and the plastic rolled up and recycled. For C, the product system included frozen storage from the assumed time of harvest (July) until the time of delivery to the retail distribution centre (RDC). These production techniques are commonly-encountered ones producing for the UK supply pattern, although they are not the only ones.

2.2. Boundaries

The product systems incorporated production of fertilisers, canes, packaging, fuels and all other inputs. Production of material for polytunnels was included, but other capital equipment was excluded so that the calculations of global warming potential were compliant with PAS 2050 (2008). For the production system in B&C, an additional year of operation without any crop production was included as an allowance for cane production, for which direct data were unavailable.

2.3. Data

Primary data from individual operating locations were used to characterise agricultural operations, with expert consultation used to fill data gaps. Background data for inputs such as fertilisers, fuel and polyethylene film were taken from the ecoinvent database v2. Data characterising outdoor production of raspberry canes in the UK were based on the data provided for raspberry fruit production outdoors in the UK and advice from a horticultural expert on likely per ha yields. Methane and nitrous oxide emissions were calculated using the method set out in the IPPC's (2006) Guidelines for National Greenhouse Gas Inventories. This aligned the method with PAS 2050:2008. A single set of data, from one of the operating locations, was used to characterise all packhouse operations and packaging.

A dataset for chilling and short-term cold storage of soft fruit was developed to characterise packhouse operations and storage prior to transhipment either to the RDC or to a long-term cold-storage location. This embodied data provided by one business participating in the study. To create an appropriate dataset for an equivalent operation in Spain, the source of electricity used was changed from the UK grid to the Spanish one; energy consumption was assumed to be the same in both locations – i.e. any additional cooling energy required in the warmer climate of Spain was ignored. A dataset characterising frozen storage was also developed, encompassing energy use and refrigerant loss and account for the burdens of providing a unit volume of cold storage capacity for a unit time; energy use in cold stores was taken from a review of UK facilities (Evans n.d); a mixture of gases was used to represent emissions to air to reflect the range of gases used (R404A and NH_3). It is worth noting that the range of specific energy use (energy use per volume) in cold stores found by Evans is very wide for each temperature regime studied (a factor of 8 between most and least efficient) and this performance variation dwarfs any differences between the energy requirements associated with the different temperature regimes (i.e. "frozen" and "chilled"). Because most LCA results for food products relate to a unit mass of product, figures for mass of product per unit of storage volume were taken from Brunel University (2009) to calculate the cold storage volume needed for 1kg raspberries. Finally, a dataset for the unit operation of road transport in refrigerated trucks was developed using values for fuel consumption, vehicle utilisation and refrigerant loss given by Brunel University (2009); emissions per unit of fuel used were taken from ecoinvent v2.1

2.4. Impact Assessment

Impact assessment was conducted for environmental categories deemed relevant to the project. Category and impact assessment method selection also reflected Defra's desire for the study's results to be as compatible as possible with results from previous LCAs of UK agricultural commodities, particularly those produced by Williams et al., (2009). Therefore, CML midpoint methods were used for the categories global warming, eutrophication, acidification and photochemical oxidation. In addition agricultural land occupancy was reported in units of m^2.yr. (the result obtained by simply adding all land occupation of all classes in the inventory), as were the unweighted water footprint (to the method described by Hoeskstra and Chapagain, (2008); weighted water footprint (using a method very similar to that outlined by Ridoutt and Pfister (2009) whereby a water stress characterisation factor for producing locations was introduced through the use of a water stress index (WSI, Pfister et al., 2009) to "weight" the water use according to the degree of water stress at the place of use) and environmental impact quotient (EIQ, Kovach et al., 1992) of pesticides used.

3. Results

The impact assessment results are shown in Table 1.

Table 1. LCIA results, raspberries in the UK at different times of year.

	Product System		
Impact Category	**A. Raspberries, fresh at UK RDC in May**	**B. Raspberries fresh at UK RDC in July**	**C. Raspberries frozen at UK RDC in November**
GWP100 (kg CO_2 eq)	7.3	7.4	7.7
Water footprint (WF) (m^3 Virtual water)	2.7	1.3	1.3
Weighted WF (m^3 Virtual water)	2.7	0.09	0.09
Agricultural land occupation (m^2.yr)	1.5	1.6	1.6
Pesticide hazard indicator E.I.Q.	0.3	0.3	0.3
Abiotic depletion (kg antimony eq.)	0.01	0.004	0.006
Photochemical oxidation - high NO_x (kg ethylene eq.)	0.0004	0.0001	0.0002
Acidification - (kg SO_2 eq.)	0.01	0.003	0.004
Eutrophication (kg PO_4--- eq.)	0.005	0.004	0.004

Sensitivity analysis was undertaken to assess the effect on the LCIA results of raspberry cane yield (no. Per ha) for case A and of cold-store operating parameters for case C. Selected results are shown in Table 2 (below). Results for impact categories driven strongly by horticultural process parameters (for example, land occupation, GWP) are, as expected, highly sensitive to product yield. Categories to which transport process emissions contribute more (notably acidification) are of course less sensitive to this factor. Sensitivity to other parameters was found to be weaker than to those included in the Table. For cold storage, small variations in energy consumption per unit volume are less significant influences on overall LCA results than the nature of the refrigerant used and assumed loss rates for the more environmentally-significant refrigerants.

4. Discussion

The differences between the LCIA results for the three cases of raspberry supply are not large, except for the water footprint measures. The fact that both canes and fruit are subject to long-distance refrigerated transport is a significant factor behind the higher acidification and abiotic depletion values obtained for raspberries delivered fresh in May (A); for example product transport accounts for 35% of the acidification potential for A but 18% of the acidification potential for B. The close similarity between the EIQ values obtained partly results from the use of expert consultation to fill data gaps; results obtained for strawberries within the same project (for which the pesticide use data was of higher quality) suggest that differences in soil sterilant applications rates and frequencies can have a significant influence on this indicator.

Table 2. Sensitivity analysis results

	Product System						
Impact Category	**A. Raspberries, fresh at UK RDC in May (base)**	**A1. Raspberries, fresh at UK RDC in May, fruit yield +10%**	**A2. Raspberries fresh at UK RDC in May, cane yield +30%**	**B. Raspberries fresh at UK RDC in July**	**B1. Raspberries fresh at UK RDC in July, yield +10%**	**C. Raspberries frozen at UK RDC in November**	**C1. Raspberries frozen at UK RDC in November, maximal cold-store occupancy**
GWP100 (kg CO_2 eq)	7.3	5.7	6.0	7.4	6.7	7.7	7.5
Water footprint (WF) (m^3 Virtual water)	2.7	Not calculated	Not calculated	1.3	Not calculated	1.3	Not calculated
Weighted WF (m^3 Virtual water)	2.7	Not calculated	Not calculated	0.09	Not calculated	0.09	Not calculated
Agricultural land occupation (m^2.yr)	1.5	1.3	1.5	1.6	1.5	1.6	1.6
Pesticide hazard indicator E.I.Q.	0.3	0.3	0.3	0.3	0.3	0.3	0.3
Abiotic depletion (kg antimony eq.)	0.01	0.01	0.01	0.004	0.004	0.006	0.005
Photochemical oxidation - high NO_x (kg ethylene eq.)	0.0004	0.0004	0.0004	0.0001	0.0001	0.0002	0.0002
Acidification - (kg SO_2 eq.)	0.01	0.008	0.009	0.003	0.003	0.004	0.003
Eutrophication (kg PO_4--- eq.)	0.005	0.004	0.005	0.004	0.003	0.004	0.003

For impact categories other than water, likely (e.g. year-to-year) variations in fruit yield and cane yield could rise to variations in the results obtained for one particular case greater than the differences between the different cases shown in Table 1.

N_2O emissions from horticulture contribute a large proportion of the GWP: 75% for A and more than 90% for B and C. However, in this project the calculation of N_2O emissions from soil following the incorporation of crop residues both in the UK or overseas was highly problematic. The IPCC 2006 method using the tier 1 approach is complex and uses many default values for specific crops or crop groups. A large number of crops are not represented in the IPCC method, therefore default data relevant to several of the products considered in this project – including raspberries - were not available. Data for another crop product were used as a proxy, but this introduces a further element of uncertainty.

The production of polyethylene contributes some 25% of the abiotic depletion potential in A. Tunnels account for the majority of the polyethylene in this case. While there is some uncertainty about the fate and longevity of the material used for these tunnels in practice, extending the material's life and recycling it when it is no longer useable are clearly desirable.

In case C (frozen raspberries supplied in November) no allowance was made for loss or spoilage during cold storage. Such losses increase the impacts associated with supplied product but no relevant data for loss rates were available when the work was conducted. Recent work by WRAP (Terry et al., 2011) provides an estimate of 2-3% losses of fresh raspberries in packing and in retail stores, but provides no estimate for losses of packed fruit consigned to frozen storage. The loss rates found for packing are similar to those used in this study.

5. Conclusions

An LCA has been completed of a soft fruit supplied in the UK at three different times of the year. The impact assessment results are quite similar across all three, perhaps surprisingly so in light of the operational differences between the supply systems. The influence on environmental impact of the place of production shows through strongly in the weighted water footprint. This impact assessment method has, of course, location-sensitivity built into it; it may be that if regionalised methods been used for other categories (notably eutrophication), the influence of place would have shown in those too.

Comparing the results obtained for the different cases with the results of the sensitivity analysis suggests that, in this case at least, yield and agricultural practice are stronger drivers of the environmental burden of food production than is time of supply. In such situations a strong focus on "seasonality" in sustainable food provisioning is unlikely to deliver large environmental benefits.

That said, the scoping phase of the work highlighted challenges facing any assessment of the environmental impacts associated with a shift towards "seasonal food consumption". If such a shift occurred at any significant scale, it would presumably involve a complex adjustment of food purchasing and consumption patterns. Given the association of seasonal food provisioning with economic and social benefits, understanding the environmental implications of that remains a desirable aim. Future work should look beyond single food items. In doing so, differences in understandings of seasonal food, modern health norms (e.g. "5-a-day") and 21st-century consumer expectations[7] must be taken into account in order to identify a realistic consumption or purchasing pattern to assess.

6. References

Brooks, M., Foster, C., Holmes M., Wiltshire J., 2011. Does consuming seasonal foods benefit the environment? British Nutrition Bulletin, 36, 449-453.

Brunel University, 2009. Greenhouse Gas Impacts of Food Retailing - report of project FO 0405. Defra, London.

Dibb, S., Collins, J. and Mayo, E., 2006. Season's Promise: an enjoyable way to tackle climate change. National Consumer Council.

Evans, J., n.d. Cold storage of food. Review of available information on energy consumption and energy savings options. FRPERC, University of Bristol.

Foster. C., McMeekin, A., Mylan, J., 2012. The entanglement of consumer expectations and (eco) innovation sequences. Technology Analysis and Strategic Management, 24, 391-405.

Freidberg, S., 2009. Fresh, a Perishable History. Harvard University Press, Cambridge, Mass.

Hoekstra, A.Y. and Chapagain, A.K., 2008. Globalization of water: Sharing the planet's freshwater resources, Blackwell Publishing, Oxford, UK.

Kovach, J., Petzoldt, C., Degnil, J. et al., 1992. A method to measure the environmental impact of pesticides. New York's Food and Life Sci. Bull. 139, 1-8.

Pfister, S., Koehler, A. and Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environmental Science and Technology 43, 4098-4104.

Ridoutt, B.G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environmental Change - Human and Policy Dimensions, 20 (1), 113-120.

Terry,L.A., Mena, C., Williams, A., Jenney, N., Whitehead, P., 2011. Fruit and vegetable resource maps: final report of project RSC008, WRAP, Banbury.

Williams, A.G., Pell, E., Webb, J., Tribe, E., Evans, D., Moorhouse, E. & Watkiss, P., 2009. Comparative Life Cycle Assessment of Food Commodities Procured for UK Consumption through a Diversity of Supply Chains - Final Report for Defra Project FO0103. Defra, London.

[7] For a discussion of the co-evolution of consumer expectations and food products, see Foster *et al.*, (2012) or Freidberg (2009)

Food waste from cheese and yoghurt in a life cycle perspective

Hanne Møller*, Vibeke Schakenda, Ole Jørgen Hanssen

Ostfold Research, Gamle Beddingv. 2b, 1671 Kråkerøy, Norway
* Corresponding author. E-mail: *hanne.moller@ostfoldforskning.no*

ABSTRACT

Food waste is a major contributor to climate change, however many LCA studies do not include it. In this carbon footprint case study of cheese and yoghurt, specific data relating to food waste at the retail stage have been included for both products, and, in the case of cheese, also at the consumer stage. The cheese case study compares sliced cheese with cheese in a whole piece. The waste at the consumer stage is significantly higher for the whole piece of cheese than sliced cheese, but the opposite is found when one considers the use of packaging material. When one looks at the total life cycle, sliced cheese has slightly lower global warming potential when compared with cheese in a whole piece. This reveals how the increased use of packaging materials can result in a lower impact on global warming, where the packaging solution reduces product waste. In the yoghurt case study LCA, four different sizes of product and corresponding packaging systems are compared. The results show that the packaging for the largest product unit had the lowest total global warming potential.

Keywords: LCA, cheese, yoghurt, consumption, food waste, packaging

1. Introduction

Food waste or loss is measured only for products that are directed to human consumption, excluding feed and parts of products which are not edible. In the early life cycle stages (production, postharvest and processing stages) food loss can be defined as loss. At later stages of the life cycle (retail and final consumption) the term food waste is applied and generally relates to behavioural issues (FAO, 2011).

Food waste can make a major contribution to climate change. Food waste creates impact both during production of the wasted product, and from waste treatment. In many LCA studies food waste as an element, has not been included, or included only with data based on assumptions.

Packaging and packaging waste is a major concern, both in the environmental debate and in governmental regulations and policies. Packaging waste is a highly visible problem both domestically and in the retail sector. Other case studies have focused on packaging, and shown that the packaging itself has little significance for the product system as a whole (Busser and Jungbluth, 2009). With regard to packaging, the following factors can affect the amount of food waste relating to the product: improved information about the best before date; packaging size adjusted to the consumer, and use of packaging barrier properties which can increase shelf life. There is no doubt that environmental impact can be significantly reduced with a decrease in food waste, however the extent to which new packaging solutions can influence food waste is not clear (Williams, 2011).

2. Methods and data

The carbon footprint case study on cheese and yoghurt, aims to show comparisons between different product and packaging solutions. The functional unit is the consumption of 1 kg cheese and 1 kg yoghurt respectively, in different packaging solutions. The product system comprises the entire life cycle from cradle to grave. The data for farming and dairy produce is based on a review of literature relating to dairy products (IDF, 2009). This review considered 60 studies on the environmental impact of dairy products, and the averages are used in this case study on cheese and yoghurt. The case study has a particular focus on food waste and packaging. For both products, specific data relating to food waste at the retail stage have been included and, in the case of cheese, food waste at the consumer stage has also been included.

The life cycle of MAP-packed sliced cheese and vacuum packed whole piece cheese have been chosen as examples, in order to highlight the issues relating to optimal packing for cheese. Similarly, in the case of yoghurt, four different packaging solutions have been selected: a 500 ml cup, an 8*125 ml cup, a duo-cup and a drink bottle 0,3 ml..

The data for food waste at the retail stage have been calculated using data from a food waste project (Hanssen and Schakenda, 2011). Data for six retail stores have been selected, based on revenue (high, medium and low) and the type of store. Data for these stores have been sorted and food waste calculated in each packing group for cheese and yoghurt (table 1 and 2).

Table 1. Food waste as a percentage for semi hard cheese

Food waste%	MAP* packed sliced cheese	Vacuum packed whole pieces
Retail	1.5	1.6
Consumer	7	12

*MAP - Modified atmosphere packaging

Food waste at the consumer stage is measured by detailed data from a waste sorting analysis (Syversen, 2010). The data is based on household waste from a limited area. Data is only recorded for cheese, the samples of yoghurt waste being too small to enable the calculation of food waste.

The data for food waste at the retail stage is quite similar with regard to the different cheese packagings. At the consumer stage the food waste is much higher and in addition, it shows a significant difference between the two cheese packaging solutions. In the case of yoghurt the food waste at the retail stage is higher for the 500 ml cup than for the other packaging categories.

Table 2. Food waste as a percentage for yoghurt

Food waste%	500 ml cup	8* 125 ml pack	Duo-cup	Drink bottle
Retail	1.8	0.6	0.7	0.8
Consumer	-	-	-	-

In the LCA methodology, waste is normally shown as an output from one system to other systems (material recycling or energy recovery) or to deposits, where only the impact of waste treatment is included. Product waste in each part of the chain can be included in the reference flow if the functional unit has been defined as food being utilised by the consumer. In that case the food waste will be added to the reference flow of the product, resulting in a higher environmental impact from those earlier stages in the product life cycle. However, if the food losses are generated at the retail and consumer stages, the environmental impact should also be placed at these stages. This is the accounting system used in this case study. The global warming potential from food waste includes both emissions from the production of food as it becomes waste and the emissions from waste treatment.

3. Results

In the LCA of cheese and the corresponding packaging systems, sliced cheese is compared to cheese in whole pieces (Curran, 2012). The product system includes the entire life cycle from cradle to grave (Fig. 1). Since the project focuses on packaging solutions and food waste, the data from farm and dairy produce are average literature data, and thus it has not been possible to refer to specific allocation rules. The principle of including both the impact of the production of food waste on the retail and consumer stage and the impact of waste treatment is also shown in Fig. 1.

The results show that agricultural production has the most important global warming potential (Fig. 2). Consumer waste is much higher for cheese in whole pieces than for sliced cheese. The use of packaging material is 5 times higher for sliced cheese when compared with cheese in whole pieces but overall, sliced cheese has slightly lower global warming than cheese in a whole piece. The packaging system of sliced cheese itself represents approximately 3% of the total impacts on global warming, but as shown above it affects the entire system much more than does the production of packaging and packaging waste. The packaging for cheese in whole pieces represents less than 1% for the total global warming potential. This shows how the increased use of packaging materials can result in a lower impact, if the packaging solution reduces product waste.

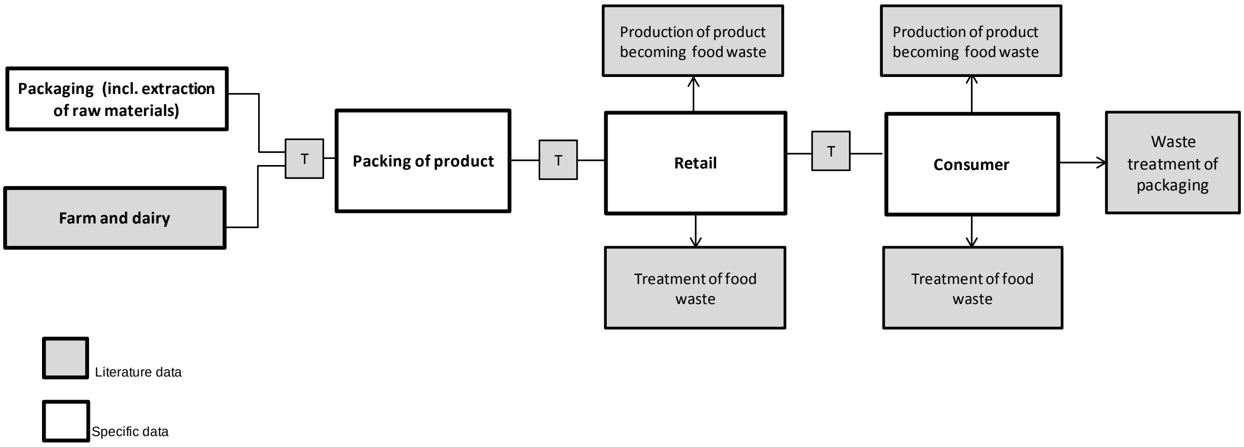


Figure 1. General product system for cheese and yoghurt

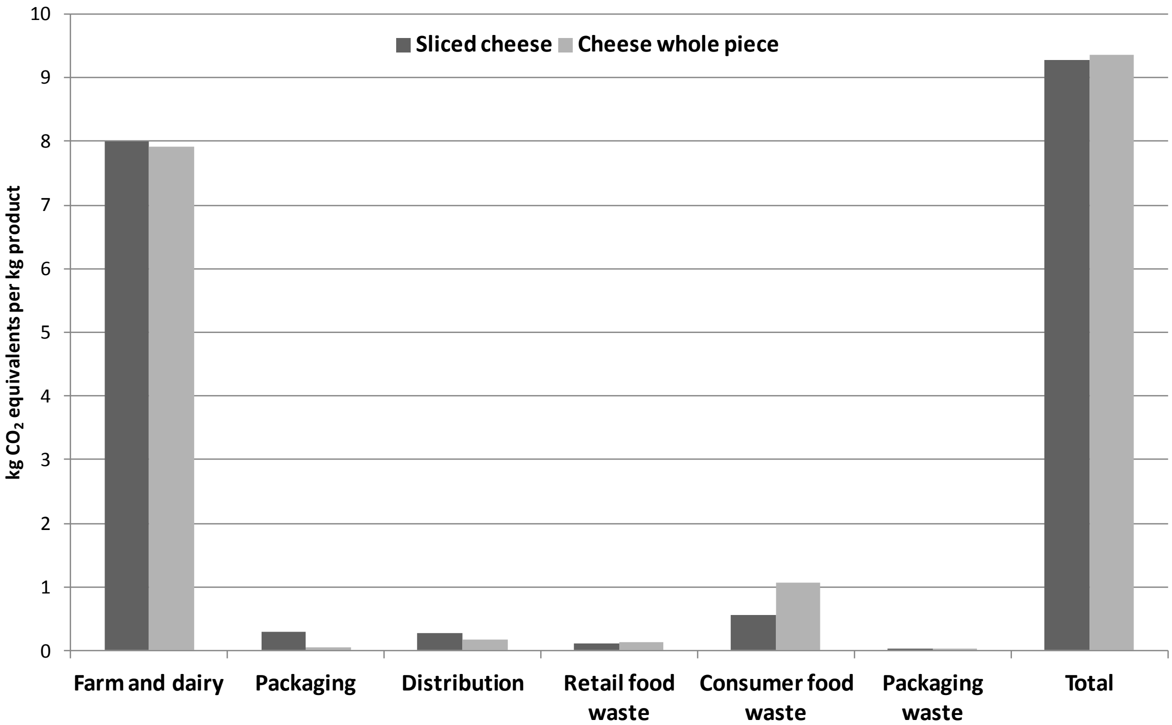


Figure 2. Emission of GHG from 1 kg of cheese throughout its life cycle

In the yoghurt case study, four different sizes of product and packaging systems are compared. The result shows that the packaging for the largest product unit has the lowest total impact on global warming. The yoghurt drink bottle has the highest global warming potential due to heavy packaging per unit and therefore also high packaging waste after consumption. The packaging system represents between 6% and 14% of global warming potential for the four different yoghurt product and packaging systems.

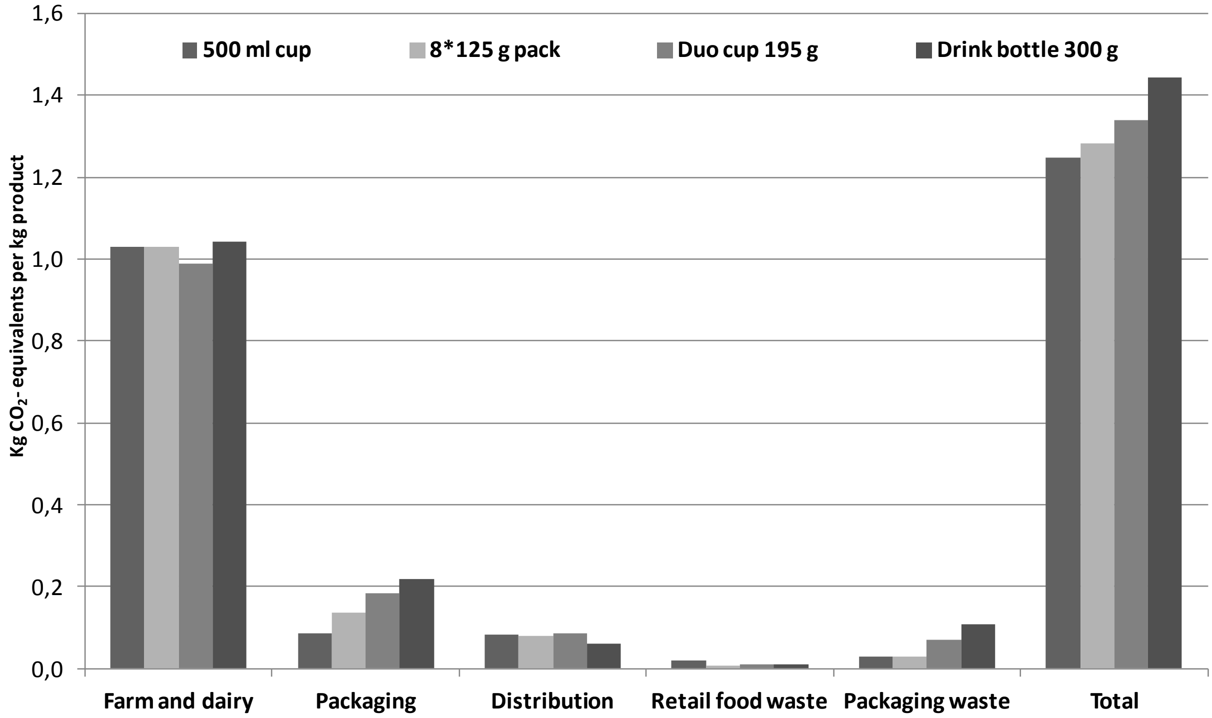


Figure 3. Emission of GHG from 1 kg of yoghurt throughout its life cycle

4. Discussion

The case studies focus on global warming throughout the products' life cycle, the packaging system and loss through product waste. The life cycles of both sliced cheese and a whole piece of cheese are used as examples to show the issues of optimal packing. The largest difference between the two product systems is the production of the packaging and consumer waste. Production of packaging for sliced cheese has about 5 times the global warming potential for that of cheese in a whole piece due to a higher packaging weight per kg of cheese. The waste among consumers is significantly higher for the whole piece of cheese than sliced cheese. In total, sliced cheese has a slightly lower impact on global warming compared with cheese in a whole piece. This shows how the increased use of packaging materials can be compensated if the packaging solution reduces product waste.

The result for yoghurt shows that the largest product unit has the lowest global warming potential. This corresponds to another LCA study of yoghurt and its packaging solutions (Busser and Jungbluth, 2009). The duo cup has lower global warming potential from the farm and dairy, as part of the product is muesli, which in itself has a lower impact on global warming.

Reduction of waste is often the most important action in packaging optimisation. The study also shows that packaging optimisation should be based on the packaging function, and focus on the entire value chain, not just on the packaging itself. This case study is an example of how packaging can affect the amount of food loss. It will not be possible to apply this as a generalisation for all product types, but it provides a good documentation of how a new product and packaging system can, in some cases, reduce the impact on global warming. It is important to use the results of further product development, but also to increase understanding among consumers in order to reduce food waste.

5. Conclusion

The carbon footprint study shows the effects on global warming of products and packaging systems, focusing on packaging and food waste. The results of the study support the existing research (Williams et al., 2008 and Johansson, 2002) but are now made more robust by the use of specific data for product waste. A packaging system with a larger packaging consumption can be justified with reduced global warming potential if it avoids product waste. Reduction of waste is often the most important action in packaging optimisation.

6. References

Büsser, S., Jungbluth, N.,2009, The role of flexible packaging in the life cycle of coffee and butter. Presentasjon på 15th SETAC Case Studies Symposium. www.esu-services.ch

Büsser S. and Jungbluth N., 2009. LCA of Yoghurt Packed in Polystyrene Cup and Aluminium-Based Lidding. ESU-services Ltd. Uster, Switzerland. Commissioned by German Aluminium Association (GDA) in cooperation with European Aluminium Foil Association (EAFA), Düsseldorf, Germany.

IDF, 2009. Environmental/Ecological Impact of the Dairy Sector: Literature review on dairy products for an inventory of key issues. Bulletin of the IDF No. 436/2009. List of environmental initiatives and influences on the dairy sector http://ns6924.ovh.net/Residents/filidf/Public/Publication.php?ID=28664.

Curran, M. A., 2012. “Life Cycle Assessment Handbook: A Guide for Environmentally Sustainable Products, (345–368). Scrivener Publishing. ISBN: 9781118099728

FAO, 2011. Global food losses and food waste. Report fromThe Swedish Institute for Food and Biotechnology (SIK) on request from FAO.

Hanssen, O. J., Schakenda, V., 2011. Analysis of status and trends of food waste in Norway 2010-11. Report from the ForMat Project (in Norwegian). Report from Ostfold Research OR 27.11. ISBN 978-82-7520-655-6.

Johansson, 3. 2002. Förpackningens betydelse för produktförluster i hemmet. Packforsk report nr. 204.

Syversen, F., 2010.Household waste analysis for characterization of usable food waste. Food Waste Prevention. Mepex Consult AS. Februar 2010

Williams, H., Wikström F., Löfgren M., 2008. A life cycle perspective on environmental effects of customer focused packaging development. Karlstad University, SE-651 88 Karlstad, Sweden. Journal of Cleaner Production 16 (2008) 853e859.

Williams, H., Wikström F., 2011. Environmental impact of packaging and food losses in a life cycle perspective: a comparative analysis of five food items. Karlstad University, SE-651 88 Karlstad, Sweden. Journal of Cleaner Production 19 (2011) 43-48.

Food waste in the food chain and related climate impacts

Juha-Matti Katajajuuri*, Kirsi Silvennoinen, Hanna Hartikainen, Lotta Jalkanen, Heta-Kaisa Koivupuro, Anu Reinikainen

MTT Agrifood Research Finland, Responsible Food Chain – Better Consumer Well-being, 00790 Helsinki, Finland
* Corresponding author. E-mail: *juha-matti.katajajuuri@mtt.fi*

ABSTRACT

The Finnish Foodspill study focussed on mapping the volume and composition of avoidable food waste in the Finnish food production-consumption chain and showed that around 130 million kg of food waste is generated each year (23 kg *per capita*/year) from the household sector. Most of the discarded food was fresh and perishable, or leftovers from cooking and dining. Converted into greenhouse gases, the annually discarded food from Finnish households is approximately equal to the annual carbon dioxide emissions of 100,000 cars. In the food service sector the amount of waste ranged from 7-28% for cooked food, depending on restaurant type. In the entire sector it was estimated to be 75-85 million kg per year. Food waste was estimated to be 65-75 million kg per year in the retail sector. The entire food industry was estimated to produce around 75-140 million kg of food waste per year.

Keywords: food waste, avoidable, household, food chain, climate impact

1. Introduction

Food accounts for over a third of the environmental impact of Finnish consumption. When examining the impact on climate alone, food (agriculture, food industry, wholesale and retail, restaurants, and household activities) amounts to about a quarter of the climate impact of Finnish consumption, and the impact on the water system is even more pronounced due to eutrophication (Seppälä et al., 2011).

It is argued that globally roughly one-third of food produced is lost or wasted, which amounts to about 1.3 billion tonnes per year (Gustavsson et al., 2011). It is both ecologically and economically unsustainable to waste edible food rather than consume it because the environmental impacts of producing the raw materials and processing them into food are rendered pointless. Improving efficiency in food production and consumption, as well as changing the general diet in western countries is vital for ensuring the future food supply for up to 9 billion people (e.g. Foley et al., 2011).

The aim of this paper is to ascertain the volume of Finnish food waste and its distribution among all parties involved in the food supply chain. In Finland to date there have been no large-scale food waste studies encompassing the entire food chain and only few limited studies have covered the amount and sources of food waste produced by households (Tarvainen 2009, Koivupuro et al., 2010). We specifically targeted households, food services, industry and the retail sector.

2. Materials and methods

In this study we concentrated on avoidable food waste, i.e. all wasted food and raw material that could have been consumed had it been stored or prepared differently. Other biowaste, such as coffee grounds and bones, was measured only for the food service sector. Of the liquid foodstuffs we included milk, being integral part of Finnish food culture. (Silvennoinen et al., 2012a).

2.1. Household data collection and analysis

We collected data in September 2010 by carrying out a follow-up study mapping the volume and composition of food waste in Finnish households. In addition, we charted the respondents' demographical backgrounds, including age, education and current life stage. Furthermore, we collected background information on, *inter alia*, eating and shopping habits, waste processing, opinions about food packaging, and we also evaluated the influence of these factors on food waste. The respondents were chosen from an online consumer panel. A market research company responsible for arranging the survey also managed the practical arrangements for the study.

A total of 420 households participated in the study and of these 380 households (1054 people) finished the study acceptably. Geographically, the households were situated in and around four cities: Helsinki, Turku, Tampere, and Jyväskylä. Prior to the study, the participants completed an online background questionnaire and they were equipped with electronic kitchen scales, as well as a diary and detailed instructions on how to weigh and record their waste and associated reasons.

The households weighed their food waste daily, each time they disposed of food. The study period was two weeks, and the results were recorded in the diary. The study was carried out in autumn, as summer holidays were over and the school autumn and winter holiday season had not yet started. The diary had separate entries for each time food was disposed of, where the respondents entered not only the weight and type of food disposed of, but also the reason for disposal, such as 'spoiled'; 'past best-before date', etc. Diary entries were easy to make under headings such as 'bread', 'potato and potato products', 'home cooked food', and 'convenience food', so that the respondent needed only to tick the corresponding box in the form. More detailed description of demographic factors is presented in Silvennoinen et al., (2011).

We studied the influences of several socio-demographical, behavioural, and attitudinal factors on generating food waste. Initially we studied the data collected through the background questionnaire, covering the influences of all the socio-demographical, behavioural, and attitudinal factors that we expected to influence the amount of avoidable food waste, based on our assumptions and previously published food waste studies. Subsequently we analysed the influence of several other factors, for which we had collected data, in order to establish whether we could identify some further, more unexpected correlations. To begin with, we analysed most of the data using descriptive statistics, crosstabs, and histograms. Afterwards we applied a linear regression model to the most promising factors to establish statistical significance of results. We formed dichotomous variables to include qualitative information into the model. We also formed dummy variables, so that we could perform regression analysis using a categorical (ordinal and nominal) variable with more than two categories. In addition, for some of the most interesting variants that stood out in the statistical tests, we performed one sample t-tests on the dummy variables (Koivupuro et al., 2011).

2.2 Food services data collection and analysis

Two communal food services and a company responsible for catering for the restaurants of Helsinki University were partners in the study. The three companies had a total of 55 outlets, providing meals for various daycare centres, schools, hospitals, elderly service centres, and workplace restaurants and canteens. The study time for food services lasted one week.

Other restaurant and catering businesses, such as diners, restaurants, hotels, cafes, petrol stations and similar establishments serving meals, participated during a shorter, one-day research period. In total the study covered 17 such businesses and there were 72 participating restaurants. The total number of research days was 292. Most of the outlets were schools and daycare centres (see Silvennoinen et al., 2012a).

In restaurants, diners and food outlets the food waste was measured by establishing the amount of food served, and weighing waste generated during cooking and serving, and customer leftovers. All restaurants participating in the study sorted and weighed leftovers. For communal food services the study was generally carried out at lunch-time, with the exception of elderly service centres and hospitals where dinner was considered. In cafes, petrol stations, diners and restaurants the whole day was usually covered. After the restaurants closed, either the restaurant personnel or the researchers weighed the sorted waste. In addition, the personnel completed forms with daily amounts of food prepared, and amounts of food waste from cooking, service, and leftovers.

Furthermore, the researchers studied the leftover content over 33 days in various outlets, establishing the composition and quantity of leftovers. Of the liquid foodstuffs, we included milk and sour milk used in the kitchen and served to the customers, but only milk was separated from leftovers. As the amount of waste food was compared to the amount of food cooked, all cooked food was weighed during the study period. The personnel in restaurants and food service locations were briefed and instructed on how to define food waste and sort leftovers so that avoidable food waste could be measured. The restaurants were provided with the necessary forms, containers for various types of food waste, boards with guidelines for sorting food waste, and several scales for weighing the produced food and food waste. A detailed description of data collection is reported by Silvennoinen et al., (2012a, 2012b).

2.3. Food waste in the retail sector and industry

The data collection for food waste in the retail sector was done in cooperation with the Nordic Council's Retail food waste project. The project was carried out by interviewing various parties in retail chains, waste management, and other associated actors in Sweden, Norway, Denmark and Finland. In Finland we interviewed four retail chain representatives (covering 90% of food markets), one waste management representative (Helsinki Region Environmental Services Authority) and a member of The Finnish Grocery Trade Association. The research did not include any weighing to determine the actual amount of waste, and consequently there are no public statistical data available (Stenmark et al., 2011).

The food waste data from the food industry were collected mainly from Finnish food companies that participated in the study, and from the literature and some corporate responsibility reports. These data were collected for the first time in Finland and this information represents a preliminary estimate of the food waste in the industry. A suitable definition for food waste in industry turned out to be difficult.

2.4. Environmental and economic analysis of food waste

The environmental impacts of the life cycle of food waste were quantified, not only those linked to the treatment of food waste (such as methane from landfills), but also those generated during earlier stages, i.e. unnecessary emissions from the food production chain. The magnitude of environmental impacts of household food waste were analysed and expressed in terms of climate impacts. However, many international standards and guidelines (ISO 14040/4, 14067, PAS 2050, WRI/WBCSD GHG protocols, ILCD, DHCF, IDF etc.) are published but no commonly approved standards or communication methods evaluating a foodstuff's climate impacts are available. In addition, the standards and guidelines are too generic to provide practical instructions to produce comparable LCA studies. Thus we estimated climate impacts of household food waste by food type categories using numerous data sources (Katajajuuri 2009, Pulkkinen et al., 2011, Usva et al., 2009, Williams et al., 2006) by trying to identify acceptable and relevant CO_2-equivalent values for different food product categories. In parallel we made some approximations for the average GHG emissions per ton of food wasted, similarly as for the European Commission (2010) food waste report. With these two approaches we managed to evaluate the magnitude of climate impacts for households. For the retail and restaurant sector, as well as partly for industrial food waste, the latter approach was taken. The economic values were estimated using statistical data combined with information from Statistics Finland, Tike Agricultural Statistics and the National Consumer Research Centre. Statistics Finland is the Finnish public authority that compiles and reports most official national statistics (OSF 2011).

3. Results

3.1. Household food waste

During the two-week study period the amount of avoidable food waste per person ranged from 0 to 23.4 kg. When extrapolated to describe the food waste over one year, the average annual avoidable food waste ranged from 0 to 160 kg per person, on average corresponding to about 23 kg of food waste per person each year (Silvennoinen et al., 2012a).

Most of the discarded food was fresh and perishable, or leftovers from cooking and dining. Discarded food was diverse: the principal discarded foodstuffs were vegetables 19%, home cooked food 18%, milk products 17%, bakery and grain products 13% and fruits and berries 13%. For meat, fish and eggs the number was 7% and for convenience food 6%. Home cooked food included various foodstuffs prepared at home, such as casseroles, stews, sauces, gravies, porridges, and soups. Convenience food included ready-made casseroles and other meals, but also hamburgers, pizzas and baby food, including infant formula. The waste from tinned goods and other non-perishable foodstuffs, such as snacks, was relatively low, only 2.5% (Silvennoinen et al., 2012b).

The main reasons for disposing of food were spoilage, e.g. mouldy 29%, past 'best before' date 19%, leftovers from dining 14% and preparing food in excess of needs 13%. The reasons behind food waste varied for foodstuffs discarded most often. For example, vegetables were discarded because they were spoiled, whereas home cooking was discarded as leftovers or due to preparing too much food. For milk products, the reasons were most often passing 'best before' or 'use before' dates. Bread, on the other hand, was either mouldy or otherwise undesirable, presumably due to drying out and becoming less appetising (Silvennoinen et al., 2012a).

We also studied possible socio-demographical, behavioural, and possible attitudinal factors that could explain household food waste. Socio-demographical factors were, for example, age, size and type of the household, and number of children. Behavioural and additional factors included shopping habits, waste sorting habits and influence of package size. Unsurprisingly, we found that the size of the household was directly correlated with waste produced – the more people there were in a household, the more waste was produced. When examining waste per person, we found that singles in general produced more waste than others, and single women in particular produced the most food waste (Koivupuro et al., 2011).

Statistically significant factors in household background information were size of household, type of household, gender of person mainly responsible for grocery shopping, opinion on potential to reduce food waste, appreciation of low food prices and opinion on the effect of purchasing the most appropriate packaging sizes. When studying the effects of the packaging, we obtained several interesting statistically significant results, such as households with fewer occupants had a stronger belief in their abilities to reduce food waste if food was sold and bought in smaller packaging. (Hartikainen et al., 2012). Other demographic factors did not explain waste in a clear and consistent way (Koivupuro et al., 2011). The next phase of the research is to include the amount of food bought (in different product categories), to analyse food waste and reasons for it and the effect of socio-demographical factors.

3.2. Food waste in the food services sector

When examining all restaurants, service waste (over production) generally represented the main category of food waste. The main difference between self-service restaurants and restaurants where food was prepared to order was that in the latter the main component of food waste was leftovers. However, the amount of leftovers varied notably from one restaurant to another, depending on the restaurant's business model and type, which in turn determined the portion sizes and the menu. In the restaurant business 25% of all food is served through licensed restaurants, hotels, and catering services. During the study we found that the restaurants representing this sector discarded 19% of all food produced and served. Of that 6% was kitchen waste, 5% service waste, and 7% leftovers. From these results we can deduce that in Finland food waste in licensed restaurants totals about 18-20 million kg per year.

Workplace restaurants and canteens serve 14% of all food in the Finnish restaurant sector. In these establishments 24% of food went to waste, as follows: kitchen waste 3%, service waste 17%, and leftovers 4%. These results would indicate that workplace restaurants and canteens produce 13-16 million kg of food waste annually. In the fast food sector food waste was only 7% of all food handled, 2% of this was attributable to kitchen waste, 3% leftovers and 2% service waste. The results translate into nation-wide annual food waste of roughly 3-4 million kg (Silvennoinen et al., 2012b). The results show that the restaurant sector overall produces 75-85 million kg food waste. This means that about 20% of all produced and handled food in the sector is wasted.

3.3. Food waste in the retail sector and industry

According to the interviews we estimated the total food waste of the Finnish wholesale and retail business to be 65,000-75,000 tonnes annually; 12-14 kg per Finnish citizen. The amount of food waste in households is approximately double that, 20-30 kg per person. The main product groups associated with food waste in stores are fruits, vegetables and bread. Other products resulting in waste include dairy products, fresh meat, fish and convenience food. Pursuant with Finnish Law, perishable products may not be sold after 'best before' or 'use before' dates, when they are to be removed from the shelves. The least food waste was found for tinned goods, dried or frozen food, and other non-perishable goods. These product groups were identical in all Nordic Countries (Stenmark et al., 2010).

The food industry was estimated to produce around 75-140 million kg of food waste per year. This corresponds to around 3% of the total production volume of the Finnish food industry. Altogether, households, restaurants, the food industry and the retail sector produce 62-86 kg of food waste per year *per capita*, corresponding to 335-460 million kg of food waste in Finland per year.

3.4. Climate impacts of food waste

Converting household food losses into greenhouse gases, a rough estimate in Finland for the annually discarded food from households is equal to the annual carbon dioxide emissions of 100,000 cars. Even though pork and beef products amounted to only 4% of all discarded food, their carbon footprints were among the highest compared with other food waste categories. For example, the amount of discarded cheese was less than 2% of total household food waste, but its carbon footprint was almost equal to that of discarded vegetables. At the entire Finnish food system level, including retailers, restaurants and the food industry, the total carbon footprint of food waste was 500-1000 million kg of CO_2-equivalent per year, around one percent of Finnish total annual greenhouse gas emissions.

4. Discussion and conclusion

On average 23 kg of food per person per year was wasted (and which was avoidable) in households based on this Foodspill study. In our study we calculated average household food waste from diary entries, and established that the *per capita* values were significantly lower than for other industrialised countries (e.g. Jones 2005, Knudsen 2009 and KFS 2009). However, the results from other studies are not directly comparable due to differences in methodologies. Furthermore, because the respondents weighed food waste and recorded the reasons for discarding food themselves, it is not possible to evaluate the accuracy and truthfulness of the diaries. The act of weighing may in itself also have reduced waste. In addition, the respondents comprised more families with children and households with multiple people than the Finnish average. The average household size in the sample, 2.8, was markedly higher than that of an average Finnish household, which in 2009 was 2.08 (OSF 2010).

On the other hand, when we studied avoidable food waste volumes in restaurants and workplace restaurants and canteens, the results were about same, and were lower than for other countries. Leftovers discarded by customers ranged from 4% to 8%, which is less than recorded in international studies (Engstrom 2004, The School Food Trust 2009). In addition, a recent Finnish biowaste study reported similar, moderate low levels of food waste (HSY 2011).

Table 1. Avoidable food waste in the Finnish food supply chain (Silvennoinen et al., 2012b).

Sector	Households	Food Services	Retail Sector	Food industry	Total
Total millions kg/year	120-160	75-85	65-75	75-140	335-460
kg per year per capita	22-30	14-16	12-14	14-26	62-86

Altogether, households, restaurants, the food industry and the retail sector produce 62-86 kg of food waste per year *per capita*, corresponding to 335-460 million kg of food waste in Finland per year (Table 1). Comparing our results, for which 10-15% of avoidable food in the entire chain is wasted, to other studies our results seems rather low, while some of the international studies (EU 2010, Gustavsson 2011) reported large food waste percentages. These marked differences are hard to explain based on individual aspects such as primary production not being included in the Foodspill study. Consequently we still need information from other, complementary studies, especially waste-bin composition analysis, to provide a more reliable and comprehensive account of the volume and composition of food waste in different sectors of the food supply chain.

Overall, the production of food that is wasted causes marked, unnecessary negative environmental impacts. A huge amount of resources is used to cultivate, produce, store and distribute food that is not consumed. All these resources, e.g. land, fertilisers, fuel, materials, transportation, water, and electricity, result in significant greenhouse gas emissions and also have other environmental impacts, such as increased water eutrophication.

Examining the economic perspective of food waste, we find every year that the average household uses €4,300 for purchasing food, of which the value of discarded food is €220. Thus, the total sum for food waste from Finnish households is roughly €550 million *per annum* (OSF 2011, Viinisalo et al., 2008).

When studying the contribution for different food categories, the result is quite similar for the carbon footprint: the biggest economic value categories were home cooked food, pork and beef, vegetables and bread. However, the food waste data generated did not turn out to be completely comparable as such. There remains a considerable amount of research to do to explain better the phenomenon of food wastage, which is, in any case, quite a new research field.

5. References

Engstrom, R., Carlsson-Kanyama, A., 2004. Food losses in food service institutions: Examples from Sweden. Food Policy, Volume 29, Issue 3, Pages 203-294.

European Commission, 2010. Preparatory study of food waste across EU 27. Technical Report 2010 – 054. 210 p.

Foley et al., (2011). Solutions for a cultivated planet. Nature 478, 337–342 (20 October 2011)

Gustavsson, J., Cederberg, C., Sonesson, U., Otterdijk, R., Meybeck, A., 2011. Global Food Losses and Food Waste. FAO Rome, Italy. 2011

Hartikainen, H., Koivupuro, H.-K., Katajajuuri, J.-M., Korhonen, V., 2012. Households' perceptions about the influence of food packaging sizes on their food waste levels. Packaging Technology and Science. Submitted. (14 p.)

HSY, 2011. Helsinki Region Environmental Services Authority. Pääkaupunkiseudun biojätteen koostumusAbstact: Properties of Biowaste in Helsinki Metropolitan Area. http://www.hsy.fi/tietoahsy/Documents/Julkaisut/6_2011_Paakaupunkiseudun_biojatteen_koostumus.pdf

Jones, T., 2005. Using Contemporary Archaeology and Applied Anthropology to Understand Food Loss in the American Food System, Bureau of Applied Research in Anthropology, University of Arizona, Tucson.

Katajajuuri, J.-M., 2009. In Valtioneuvoston tulevaisuusselonteko ilmasto- ja energiapolitiikasta. Government Foresight Report on Long-term Climate and Energy Policy: Towards a Low-carbon Finland. In Finnish.

KFS Konsument Föreningen Stockholm, 2009. Rapport från en slaskhink.

Knudsen, M. L. C., 2009. Affaldsforebyggelse i husholdninger – muligheder og barrierer for Danmark. Roskilde University, Specialreport, December 2009.

Koivupuro, H.-K., Jalkanen, L., Katajajuuri, J.-M., Reinikainen, A., Silvennoinen, K., 2010. Food waste in the supply chain. Elintarvikeketjussa syntyvä ruokahävikki, kirjallisuuskatsaus. MTT report 12: 73 p. http://www.mtt.fi/mttraportti/pdf/mttraportti12.pdf

Koivupuro, H., Hartikainen, H., Katajajuuri, J.-M., Silvennoinen, K., Heikintalo, N. Reinikainen, A., Jalkanen , L., 2011. Influence of socio-demographical, behavioural and attitudinal factors on the amount of avoidable food waste generated in Finnish households.

Official Statistics of Finland (OSF), 2011: Annual national accounts [e-publication]. Helsinki: Statistics Finland [referred: 27.11.2011].Access method: http://www.stat.fi/til/vtp/index_en.html.

Pulkkinen, H., Hartikainen, H., Katajajuuri, J.-M., 2011, Elintarvikkeiden hiilijalanjälkien laskenta ja viestintä. MTT raportti 2011. In Finnish.

Seppälä, J., Mäenpää, I., Koskela, S., Mattila, T., Nissinen, A., Katajajuuri, J.-M., Härmä, T., Korhonen, M.-R., Saarinen, M., Virtanen, Y. 2011. An assessment of greenhouse gas emissions and material flows caused by the Finnish economy using the ENVIMAT model. Journal of Cleaner Production 19 (16): 1833-1841 November 2011.

Silvennoinen, K., Katajajuuri, J.-M., Hartikainen, H., Jalkanen, L., Koivupuro, H.-K., Reinikainen, A., 2012a. Food Waste Volume and Composition in Food Chain, Case Study Finland. In: Evans et al., (Eds.) Waste Matters: New Perspectives on Food and Society. Wiley-Blackwell. Submitted. (20 p.)

Silvennoinen, K., Koivupuro, H.-K., Katajajuuri, J.-M., Jalkanen, L., Reinikainen, A., 2012b. Ruokahävikki suomalaisessa ruokaketjussa – Foodspill-hankkeen 2010-2012 loppuraportti. Food Waste Volume and Composition in Finnish Food Chain. MTT Raportti – MTT Report 41. http://www.mtt.fi/mttraportti/pdf/mttraportti41.pdf. 65 p. (in Finnish)

Stenmark, Å., Hanssen, O., Silvennoinen, K., Katajajuuri, J.-M., Werge, M., 2011. Initiatives on Prevention of Food Waste in the Retail and Wholesale Trades. Report. Swedish Environmental Research Institute. Juni 2011.

Tarvainen, M., 2009. Rokka rikassa – Pääkaupunkiseudun lapsiperheiden ruokajätteet. YTV:n julkaisuja 25/2009. Helsinki 2009.

Tike, 2010. Balance Sheet for Food Commodities 2008 and 2009 (preliminary). http://www.maataloustilastot.fi/sites/default/files/ravintotase_2009_ennakko.pdf

The School Food Trust (2009). The Primary School Food Survey. Research Report. http://www.schoolfoodtrust.org.uk/school-cooks-caterers/reports/primary-school-food-survey-2009

Usva, K., Saarinen, M., Katajajuuri, J.-M. & Kurppa, S. (2009). Supply chain integrated LCA approach to assess environmental impacts of food production in Finland. Agricultural and Food Science 18, 3-4: 460-476.

Viinisalo, M., Nikkilä, M., Varjonen, J., 2008. Elintarvikkeiden kulutusmuutokset kotitalouksissa vuosina 1966–2006. Abstract: Changes in the consumption of foods in households during the years 1966–2006. National Consumer Research Centre, publications 7/2008.

Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205. Bedford: Cranfield University and Defra.

Product energy use within the agri-food supply chain

Carla Sarrouy[1,*], Joe Davidson[2], Rob Lillywhite[1]

[1] School of Life Sciences, University of Warwick, Wellesbourne, Warwick, CV35 9EF, UK
[2] Institute of Manufacturing, University of Cambridge, UK
[*] Corresponding author. E-mail: *c.sarrouy@warwick.ac.uk*

ABSTRACT

The global food system consumes very large amounts of energy, a position which is made more challenging as an increasing global population is demanding more food. In order that food production can be made more sustainable, it is important to produce food in a more energy efficient way or to identify food types that require less energy in production. This paper presents the results of a UK study that undertook a supply chain analysis of the embedded energy of selected food products.

A review of the academic literature identified over 50 products for which energy analysis had been undertaken for at least one stage of the food supply chain; the majority of papers related to primary production. Embedded energy values for individual food products across the whole supply chain (primary production to retailer) ranged from 2.4 MJ/kg for potato to 83 MJ/kg for coffee. For most food products, the indirect and direct energy used in primary production remained the dominant use; 45, 45 and 72% respectively for potato, milk and pork. Processing was the second biggest influence for selected products, e.g. the frying stage for oven chips consumed 50% of total energy requirement. Embedded energy values for total product market share was dominated by 'every day' items, e.g. meat, bread, milk and cheese.

The review was supported by new analysis of four multi-ingredient products. Data were collected from multiple sources (farmers, trade associations, food processors, food manufacturers and retailers). The embedded energy of tinned soup, pasta sauce in a glass jar, restaurant pizza and chocolate biscuits was 9, 24, 28 and 20 MJ/kg, respectively. The results show that different products have very different demands for energy and that the 'hotspot' of energy use varied greatly with product; however, in many situations it was possible to identify approaches to reduce energy use or to substitute one product with another. The choice of packaging was very influential for some products; e.g. the glass jar for pasta sauce was responsible for 50% of the total product embedded energy whilst the cardboard for frozen pizza was responsible for only 18% of the total product embedded energy.

Despite the high cost of energy, its contribution to product price remains relatively minor for most products and most food companies do not identify energy as a priority; hygiene and food safety being the dominant focus.

Keywords: energy, LCA, food supply chain, dependency

1. Introduction

There is growing concern regarding the increased demand for energy within the UK agri-food supply chain, which is currently estimated to be 326 petajoules per year (Lillywhite et al., 2012). The trend for more processed foods to feed an increasing population is likely to increase the demand for energy in a time when energy supplies are becoming more insecure.

This paper reports the results of a UK study to identify energy use, at both product and market scale, and energy use hotspots within the agri-food supply chain; the supply chain is assumed to have four main stages (primary production, processing/manufacturing, packaging and logistics/retail). The identification of product and market embedded energy values and supply chain hotspots provides a better understanding of the whole supply chain and has identified areas where improvements could be made.

2. Methods

Initial research was based on a literature review of life cycle assessment (LCA) studies of food items. This review was not limited to the UK but included any country in which these studies had been conducted. The review identified 51 products for which energy analysis had been conducted in at least one stage of the food supply chain (Audsley et al., 2009; Carlsson-Kanyama et al., 2003; Erzinger et al., 2003; Lillywhite et al., 2007; Williams et al., 2006). The studies mainly focused on the primary production stage, *i.e.* primary agricultural production rather than the entire food supply chain. The review identified that very few studies had been undertaken on complex multi-ingredient products. This omission was addressed by analysing four multi-ingredient products, *i.e.* pasta sauce, pizza, soup and chocolate biscuits. These were selected because they are widely-available and highly-consumed in the UK and their ingredients include vegetables, cereals, meat and dairy products; their consumption is forecast to continue increasing in the coming years (Lillywhite et al., 2012). Data were collected from multiple sources: farmers, trade associations, food processors, food and packaging manufacturers and retailers. Two functional units were used: mega joules per kilo (MJ/kg) for individual products and terajoules (TJ) for total market share.

3. Results

3.1. Differences between products and market shares

The results show that individual products have a wide range of embedded energy values which range from 2 MJ/kg for mineral water to 83 MJ/kg for coffee (Table 1). Annual embedded energy values for total product consumption range from 67 TJ for honey to 36,498 TJ for beef.

Fruit and vegetables have the lowest embedded energy values, for example, potato (2.4 MJ/kg), onion (2.9 MJ/kg) and carrot (3.3 MJ/kg). Domestic fruits such as apples (5 MJ/kg) and raspberries (7.5 MJ/kg) require less energy than imported fruit: oranges (8.1 MJ/kg), bananas (8.7 MJ/kg) and grapes (8.8 MJ/kg). There are two exceptions to this: UK strawberries (13.6 MJ/kg) are grown in polytunnels and UK tomatoes (95 MJ/kg) in heated glasshouses. Tomatoes have a wide range of energy requirements which illustrates different production systems (field grown in Mediterranean countries and glasshouse grown in Northern European countries).

A number of dairy products were covered in the analysis. Liquid milk has the lowest embedded energy value (7.3 MJ/kg) but this increases with added processing: cream (12.1 MJ/kg), yoghurt (19.4 MJ/kg) and butter (23.5 MJ/kg) have values corresponding to 166%, 266% and 322%, respectively, of the value for milk. Of the dairy products, cheese (48.8 MJ/kg) has the highest value.

Meat is an energy intensive product with most types having similar average values: pork (33.3 MJ/kg), beef (34.4 MJ/kg) and chicken (39.7 MJ/kg) although the range of values for an individual product can be large and dependent on different production systems, e.g. beef (17.0 to 74.2 MJ/kg). It is interesting to note that these values are smaller than those for different fish species: tinned tuna (44.0 MJ/kg), salmon (57 MJ/kg) and frozen cod (61.9 MJ/kg). However, when total market share is considered, the energy requirement for fish is considerably lower than that of meat. Frozen cod and salmon require 1,581 and 1,819 TJ/year, respectively, compared to chicken and beef that require 23,190 and 36,498 TJ/kg, respectively.

The same product can have different energy requirements depending upon its processing or packaging, for example, there is a 7% difference between fresh peas and tinned peas, due to differences in the processing and logistics stages.

The review revealed that there is a lack of data on processed multi-ingredient products; this omission was rectified by undertaking new research on pasta sauce, soup, pizza and chocolate biscuits. These four products have different energy uses but do not figure as the most energy-intensive of the table below. Vegetable soup is the least energy-intensive product of the sub-group (8.92 MJ/kg) and restaurant pizza requires the most energy (28 MJ/kg).

Table 1. Embedded energy by product and market share

Product	Average product energy use (MJ/kg)	Average product energy use ranking	Range of product energy use (MJ/kg)	Market energy use (TJ)	Market energy use ranking
Apple	5.0	49	2.5 – 11.1	2,585	28
Banana	8.7	42	5.4 – 12.0	6,082	18
Bean (tinned)	18.0	22	16.0 – 20.0	6,205	17
Beef	34.4	9	17.0 – 74.2	36,498	1
Biscuit	25.4	13	23.0 – 27.2	13,296	7
Bread	9.0	39	3.7 – 15.8	18,931	5
Broccoli (fresh)	11.1	34	10.7 – 11.4	877	41
Butter	23.5	15	12.6 – 30.7	3,000	25
Cabbage (white)	4.4	50	3.7 – 5.1	548	45
Cake	16.8	24	11.6 – 21.0	8,151	15
Carrot (fresh)	3.3	53	2.6 – 4.1	2,468	29
Cereal & muesli	13.5	30	10.8 – 17.0	5,602	20
Cheese	48.8	4	35.7 – 65.0	18,175	6
Chicken	24.9	8	20.6 – 29.2	23,190	4
Chips (oven)	10.2	35	-	-	-
Chocolate	43.5	7	43.0 – 44.0	12,219	9
Chocolate biscuits	19.5	20	-	-	-
Cod (frozen)	61.9	2	45.0 – 78.8	1,581	33
Coffee	83.0	1	42.1 – 126.4	4,504	21
Cream	12.1	32	5.1 – 19.0	811	42
Crispbread	20.6	19	14.0 – 27.2	395	46
Crisps (baked)	8.7	43	-	-	-
Crisps (fried)	14.6	27	-	-	-
Eggs	29.2	11	27.2 – 31.3	9,320	12
Flour	3.6	51	1.7 – 5.2	724	44

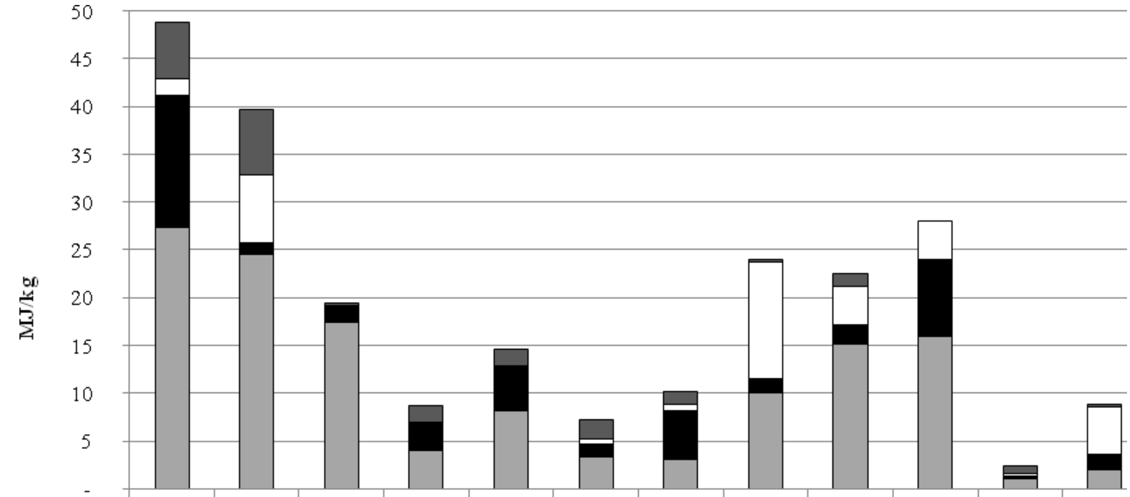
Fruit juice	9.1	38	7.1 – 10.2	9,440	11
Grapes	8.8	41	7.8 – 9.7	1,620	32
Honey	3.5	52	1.3 – 5.6	67	49
Ice cream	16.4	25	14.0 – 20.2	8,637	14
Jam	11.7	33	8.0 – 16.0	933	40
Lettuce	6.3	48	3.5 – 9.1	744	43
Margarine	20.7	18	17.0 – 24.4	1,454	34
Milk	7.3	46	3.4 – 7.0	27,154	3
Oil	21.7	17	14.0 – 35.3	4,156	22
Onion	2.9	54	1.9 – 3.8	1,046	39
Oranges	8.1	44	6.8 – 9.4	1,267	37
Pasta	9.8	36	8.7 – 13.8	2,065	30
Pasta sauce	24.0	14	-	-	-
Peas (fresh)	16.3	26	8.2 – 24.4	260	47
Peas (tinned)	17.4	23	17.0 – 17.7	1,166	38
Pizza (frozen)	22.5	16	-	-	-
Pizza (restaurant)	28.0	12	-	-	-
Pork	33.3	10	25.1 – 48.2	5,846	19
Potato	2.4	55	1.7 – 3.0	3,757	24
Raspberries	7.5	45	-	128	48
Rice	14.2	28	9.8 – 17.8	2,855	27
Salmon	57.0	3	54.5 – 59.4	1,819	31
Soft drinks	6.5	47	5.4 – 7.5	34,897	2
Soup	8.9	40	-	7,332	16
Strawberry	13.6	29	12.7 – 14.5	1,399	36
Sugar	9.8	37	-	2,909	26
Tomato	46.4	5	5.4 – 95.0	13,181	8
Tuna (tinned)	44.0	6	-	3,932	23
Water (mineral)	2.0	56	-	1,436	35
Wine	13.0	31	12.0 – 14.0	9,295	13
Yoghurt	19.4	21	13.7 – 25.1	11,146	10

3.2. Differences within products

Fig. 1 shows the energy requirements, by production stage, for the production of 12 different products. The results show that energy use varies greatly between products and within products. Primary production normally uses the most energy; ranging from 1.05 MJ/kg for potato (45% of total) to 27.4 MJ/kg for cheese (56% of total). However, there are some exceptions: oven chips consume 50% of their energy at the manufacturing stage, and pasta sauce consumes 50% at the packaging stage.

Energy use during the logistics stage is closely linked to product type. Products such as cheese (5.90 MJ/kg) and chicken (6.81 MJ/kg) that require temperature-controlled transport and storage systems have high values but ambient products, such as chocolate biscuits and soup (both requiring only 0.34 MJ/kg for logistics and storage) can be low. The exception is very light products, for example crisps, where transport containers cannot be filled to maximum weight allowances.

The study has highlighted the crucial role that can be played by packaging in the overall energy use of products. For pizza, raw ingredients had the greatest energy requirement but it was the packaging stage that was dominant for pasta sauce and soup, with 50 and 56% respectively, of their total energy used at the packaging stage. Interestingly preparation of pizza required different amounts of energy; restaurant prepared pizza having a higher embedded energy value compared to factory prepared pizza, 28.0 to 22.54 MJ/kg; whether this 5 MJ/kg saving for factory prepared pizza is nullified by home cooking is open to question. Between the four processed multi-ingredient products, the choice of raw ingredients and packaging have the biggest impact on the overall energy content, ranging between 71% (restaurant pizza) to 92% (pasta sauce) of the total.



	Cheese	Chicken	Choc. biscuits	Crisps (baked)	Crisps (fried)	Milk	Oven chips	Pasta sauce	Pizza (frozen)	Pizza (restaur.)	Potato	Soup
Logistics/Retail	5,90	6,81	0,34	1,70	1,70	2,10	1,30	0,30	1,30	-	0,75	0,34
Packaging	1,70	7,07	0,05	0,10	0,10	0,50	0,70	12,10	4,00	4,00	0,25	5,00
Processing	13,85	1,24	1,60	2,80	4,54	1,40	5,11	1,60	2,14	8,00	0,30	1,56
Primary Prod.	27,35	24,58	17,49	4,09	8,25	3,30	3,09	10,00	15,10	16,00	1,05	2,02

Figure 1. Energy content breakdown for 12 different foods

3.3. Differences in packaging

The study examined both primary packaging, used to contain the product, and secondary packaging, used to protect the product during the logistics stage. Both packaging types can have a considerable impact on the total embedded energy of products. Equally the choice of packaging material and the recycled content of that packaging material will have a considerable influence on product embedded energy. This study assumed average UK recycling rates, meaning that plastic, aluminium, glass and steel containers used 100%, 57%, 33% and 56% virgin materials respectively. (SCRIP, 2006; WRAP, not dated; WRAP, 2011).

The results show that energy use for primary packaging is much higher than for secondary packaging (Figure 2). For aluminium and steel cans, the embodied energy can be as high as 10.5 MJ/kg (94% of total energy use) and 2.5 MJ/kg (79% of total energy use) respectively. Glass jars also have a considerable energy use in the manufacturing of their primary packaging which can be as high as 50% of the total embedded energy.

Secondary packaging is much less energy-intensive than primary packaging. The importance of secondary packaging is inversely proportional to the energy use of different types of packaging. The most energy-intensive package types: glass jars and aluminium cans, have the lowest percentage of energy used for secondary packaging (3% each). The least energy-intensive packaging types, stand-up pouches and steel cans, have the highest share of energy required for secondary packaging (63% and 22%, respectively). These values are due to the fact that the less energy-intensive package types analysed tend to have a weaker structure that needs to be compensated by stronger (and more energy-intensive) secondary package types. This highlights the importance of including secondary packaging in the analysis and shows that lighter, less energy-intensive package types require much less energy, even when their secondary packaging is included in the calculation.

The analysis shows that the ratio between the energy use for packaging to product containment can be very different. The choice to package a product in a glass jar rather than in a carton means that more than five times more energy will be required to pack the same amount of containment. This decision, although often overlooked, can have a very strong impact on the overall energy value of the final product. However, this discussion on packaging types should be considered in the context of product safety which is paramount and existing production equipment. Glass is preferred in many situations because it is easy to sterilise the final product and it uses existing production lines.

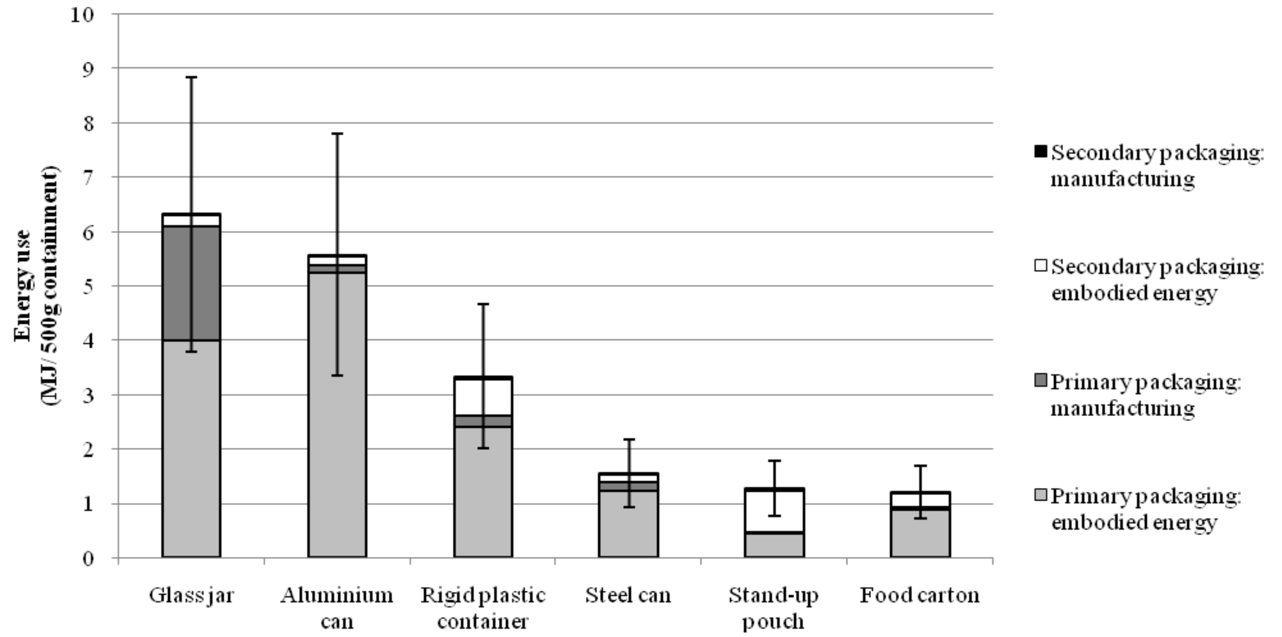


Figure 2. Energy use associated with packaging made of 100% virgin materials (plastic container, pouch and carton) and a typical recycled content (aluminium can, glass jar, steel can) (bars indicate minimum and maximum values for total packaging type) (Oswald, 2011)

4. Discussion

The results reveal the great diversity of energy requirements and product embedded energy values. This diversity suggests that no single approach is possible to reduce energy use in the agri-food supply chain and that the best measure to promote energy efficiency will be directed at specific sectors or products. For most products, primary production requires the most energy; both for simple crops, such as potatoes, and for complex processed multi-ingredient foods such as pizza. Processing is the second most energy-intensive stage of the food supply chain and for some products it is an area in which energy-saving initiatives can be undertaken. The increasing popularity of highly processed multi-ingredient foods, manufactured across different countries and regions, is likely to increase the embedded energy content of these food types. Consequently, it will be more difficult to reduce product energy requirements and successful initiatives will require the cooperation of different stakeholders in multiple countries. It is not the sole responsibility of farmers to improve the energy efficiency of their production; it is also for processors, retailers and consumers to support change and to enable change by designing more energy efficient products.

Meat and dairy products have some of the highest embedded energy values which could be used to drive an increase in consumption of more fruit and vegetables; and changes to diet towards less meat and more fruit and vegetables have been shown to bring health benefits to the consumer and environmental benefits to the planet (UN Human Rights Council, 2011). Additionally, the promotion of local and seasonal food can reduce the energy required in the transport and storage of food. In these areas, consumers have the ability to influence the agri-food supply chain through their purchasing choices.

The results also show that packaging can have an important role in reducing energy use so it is important to understand how different raw materials, manufacturing processes and waste management (the percentage recycled, sent to landfill or incinerated) are related to overall energy use. However, this stage of the agri-food supply chain is very complex because it has several limitations. Firstly, packaging types are restricted by product types (drinks tend to be in aluminium cans rather than cartons whilst nuts can easily be packed in a stand-up pouch rather than a glass jar). It would require changes to processing procedures, shelf life, storage requirements and customer perceptions to enable changes to packaging types to be introduced. However, it is likely that improving technology will still achieve energy efficiencies. Secondly, product packaging is strongly influenced by product safety but it can also be affected by social conceptions. For example, even if switching pasta sauces to soft pouches could lower product embedded energy values, and be acceptable to customers, it is possible that concerns over safety would prevent manufacturers making the switch. Finally, the embedded energy of packaging can be influenced by the end-user through recycling options. Companies that use recycled material and promote recycling, for example, Coca-Cola (Coca-cola, 2012) can increase the use of recycled material in their packaging. A continuous increase in the content of recycled material in their packaging will reduce the use of virgin material and lower the embedded energy of the packaging in the long-term.

5. Conclusion

This study of energy use in the agri-food supply chain shows that there is great diversity in product embedded energy values. Different food types have very different requirements for energy and there is considerable scope for mitigation in energy consumption in the agri-food sector. Primary production has the greatest requirement for energy which is concentrated in the manufacture of nitrogen fertiliser and the use of road fuels; energy efficiency research should concentrate in this area. Processing and manufacturing also use considerable amounts of energy although new technology and competition is bringing about greater energy efficiency. Packaging remains an area where improvements are possible although progress may be restricted by an unwillingness to innovate or to invest in new technologies and equipment. In contrast, the increasing trend towards the consumption of ready-made, chilled and frozen foods has increased product embedded energy values.

Energy is not a priority for the majority of agri-food businesses since hygiene and food safety are always of greater concern; increases in the cost of energy are a long-term but solvable issue whereas a food safety problem has the ability to ruin a business almost over-night. Our study has shown that there is plenty of interest in improving energy efficiency, mainly for economic reasons, but that there is a lack of information on the needs and concerns of the different stakeholders within the supply chain. More research is needed to identify efficient production methods, adapted to the needs of the different stakeholders. There is no single solution for the reduction of energy consumption; instead there is the need for broad spectrum changes towards energy efficiency along the entire agri-food supply chain.

6. Acknowledgements

The authors would like to gratefully acknowledge the valuable input of Julia Oswald on packaging and Defra for funding this project.

7. References

Audsley, E., Brander, M., Chatterton, J., Murphy-Bokern, D., Webster, C., Williams, A., 2009. How low can we go? An assessment of greenhouse gas emissions from the UK food system and the scope to reduce them by 2050, FCRN-WWF-UK.

Carlsson-Kanyama, A., Ekstrom, M., Shanahan, H. 2003. Food and life cycle energy inputs: consequences of diet and ways to increase efficiency. Ecological Economics 44 (2-3): 293-307.

Coca-Cola. http://www.coca-cola.co.uk/environment/ (accessed 14 August 2012)

Erzinger, E., Dux, D., Zimmermann, A., Baderscher, R. 2003. LCA of animal products from different housing systems. In: LCA in the Agri-Food Sector, Horsens.

Lillywhite, R., Chandler, D., Grant, W., Lewis, K., Firth, C., Schmutz, U., Halpin, D. 2007. Environmental footprint and sustainability of horticulture: A comparison with other agricultural sectors. Final report of Defra project WQ0101. University of Warwick and Defra, London.

Lillywhite, R., Sarrouy, C., Davidson, J., 2012. Energy Dependency and Food Chain Security, Main Report. Defra Research Project FO0415. University of Warwick and Defra, Coventry.

Oswald, J., 2011. Evaluation of the Energy Use of Packaging in the Food Industry. MPhil dissertation, University of Cambridge.

SCRIP. 2006. The Steel Lifecycle. Retrieved 28 July 2010, from http://www.scrib.org/recycling_steel/steel_lifecycle.asp.

UN Human Rights Council, 2011. Report of the Special Rapporteur on the Right to Food, The right to an adequate diet: the agriculture-food-health nexus, 26 December 2011, A/HRC/19/59.

Willams, A., Audsley, E., Sandars, D. 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Final report of Defra project IS0205. Cranfield University and Defra, London.

WRAP, (not dated) Lightweighting Beer and Cider Packaging. Retrieved 28 July 2010, from http://www.wrap.org.uk/downloads/Case_Study_-_GlassRite_16_10_08_1230.79587ba6.6085.pdf.

WRAP, 2011. Glass Toolkit – Recycled Content Calculator. Retrieved 28 July 2011, from http://www.wrap.org.uk/retail_supply_chain/research_tools/tools/glass_recycled_content_toolkit/recycled_content.html.

Comparison of the life cycle impacts of ready-made and home-made meals

Ximena Schmidt Rivera, Namy Espinoza Orias, Adisa Azapagic*

School of Chemical Engineering and Analytical Science, Room C16, The Mill, Sackville Street, The University of Manchester, Manchester M13 9PL, UK
*Corresponding author. e-mail: *adisa.azapagic@manchester.ac.uk*

ABSTRACT
This study compares the environmental impacts of a ready-made meal with an equivalent meal prepared at home. The meal considered consists of roast chicken, vegetables and tomato sauce. The results indicate that the home-made meal has on average 2.7 times lower environmental impacts than the ready-made meal. The main hotspots for both meals are the ingredients (chicken and tomato sauce) and in the case of the ready-made meal, refrigerant leakage. The contribution of packaging and meal cooking at home is small, although the latter is more significant for the home-made meal.

Keywords: convenience food, home-made meals, ready-made meals, environmental impacts, LCA

1. Introduction

Convenience food and particularly ready-made meals are becoming increasingly popular due to our modern life style. For example, the UK ready-made meal market was valued at £2.7 billion in 2010 (MINTEL, 2011) with approximately 8.8 kg of chilled and frozen ready-made meals consumed per capita per year (Millstone and Lang, 2008).

Yet, there is currently little information on the life cycle environmental impacts of convenience food, and particularly ready-made meals. Whilst numerous LCA studies exist of single food items, there are few studies of complete meals with most focusing on global warming potential, GWP (e.g. Carlsson-Kanyama, 1998; Wiltshire et al., 2008; Stichnothe et al., 2008; Espinoza-Orias et al., 2010) or on a limited number of environmental impacts (e.g. Sonesson et al., 2005; Davis and Sonesson, 2008; Davis et al., 2009; Berlin and Sund, 2010). To date, only two studies have considered the full LCA impacts of ready-made meals, both based in Spain (Calderon et al., 2010; Zufia and Arana, 2008). As far as the authors are aware, no full LCA studies have been carried out for ready-made meals in the UK.

Therefore, this paper evaluates the life cycle environmental impacts of a 'typical' UK ready-made meal consisting of roast chicken meat, vegetables and tomato sauce and compares it to an equivalent meal made at home.

2. Methodology

The life cycle assessment (LCA) methodology used in this study follows the ISO 14040 standard series (ISO, 2006) and is outlined below. GaBi software (PE International, 2011) has been used to carry out the LCA, applying the CML 2 Baseline 2001 method (Guinée et al., 2001).

2.1. Goal and scope of the study

The goal of the study is to evaluate the environmental impacts of a ready-made meal prepared industrially and compare it to the impacts from an equivalent meal made at home. The results of the study are aimed at both food producers and consumers.

The functional unit is defined as 'preparation and consumption of a meal for one person'. The weight of the meal is 360 g and it consists of roast chicken meat and three vegetables (potatoes, carrots and peas), served with tomato sauce. The meal is consumed at home. The scope is from 'cradle to grave' and the study is based in the UK.

2.2. System definition and assumptions

As shown in Fig. 1, the life cycles of the ready-made meal involves chicken rearing and cultivation of the vegetables, processing of the ingredients at a regional distribution centre (RDC), preparation of the meal at a manufacturing site, its subsequent transport to another RDC, retailer and finally to consumer's home where it is prepared according to manufacturer's instructions. The life cycle of the home-made meal is similar, except that the meal is fully prepared at home, starting from the fresh ingredients. The meal ingredients are given in Table 1.

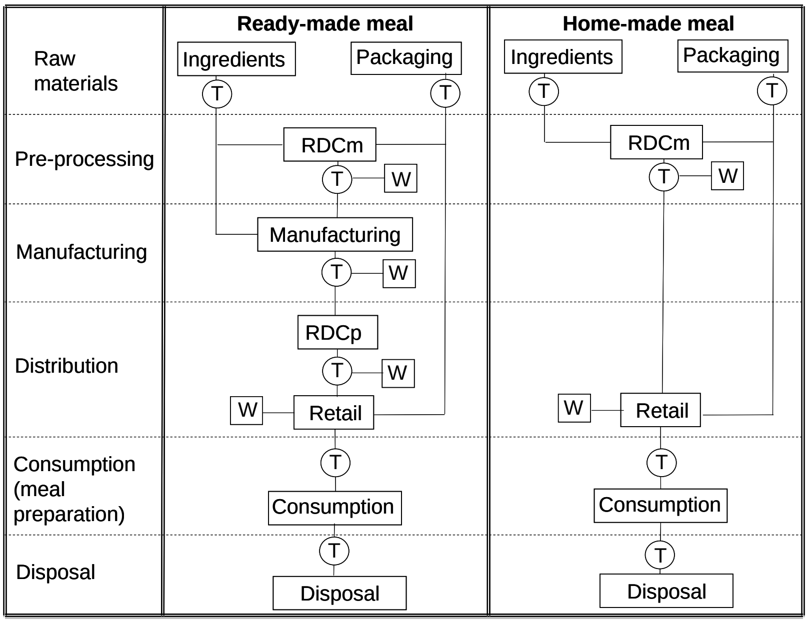


Figure 1. Life cycles of the ready and home-made meals. [RDCm and RDCp: Regional distribution centre for raw materials and products, respectively; T- transport; W - waste]

The raw materials stage involves cultivation of the vegetables and chicken rearing which are then transported from the farm to the regional distribution centre for raw materials (RDCm) to be processed. It is assumed that the raw materials (ingredients) are sourced from conventional farms in the UK, except for the tomatoes used for the tomato paste which are imported from Spain (FAO, 2009). The tomato paste, oil and salt are transported directly from their respective manufacturers to the meal manufacturer.

The pre-processing stage includes cleaning the vegetables and slaughtering the chickens, packaging in plastic bags, crates and pallets and subsequent refrigeration. The manufacture involves cooking of the ready-made meal from fresh ingredients, its packaging and transportation to the regional distribution centre for products (RDCp). Vegetables and tomato paste are cooked together while the chicken meat is cooked separately. The cooked ingredients are then combined, packaged and refrigerated.

The ready-made meals are stored at the RDCp and then distributed to the retailer as chilled. The assumed annual refrigerant leakage is 15%; the assumed average wastage of the meal at retailer is 2% (Brunel, 2008). The ingredients for the home-made meal are chilled at both the RDCp and the retailer. The ready-made meals and the ingredients for the home-made meal are then transported to consumer's home.

In the consumption stage, storage and meal preparation at home are considered. The ready-made meal is cooked in a microwave (according to manufacturers' recommendations). For the home-made meal, the average UK practice is assumed with the chicken roasted in the oven and the tomato sauce and the vegetables cooked on the electric cooker. Note that refrigerated storage at home considers the electricity used but not refrigerant leakage as domestic refrigerators have negligible leakage rates.

All the primary, secondary and tertiary packaging has been considered, including the ingredients and ready-made meal packaging, shopping bags, crates, boxes and pallets. All the waste and packaging are assumed to be landfilled, except for the chicken meat from the RDCm, which is incinerated. These assumptions are in accordance with the prevalent UK waste management practice for food-related products and packaging. All road transport is by diesel vehicles, assuming an empty return trip.

2.3. Data sources

An overview of the data used in the study is given in Table 2. As can be seen from the table, most data for the ingredients correspond to their country of origin considered in this study. The exception is the data for carrots and onions which are not available for the UK, so Danish data have been used instead (Nielsen et al., 2003). Furthermore, data for peas are also not available, so proxy data for green beans have been used following recommendations by Milà i Canals et al., (2011) on dealing with data gaps in the food sector.

Table 1. Meal ingredients

Ingredients	Weight [g]	Contribution [%]
Chicken	98.0	27.2
Potatoes	87.5	24.3
Carrots	35.0	9.7
Peas	35.0	9.7
Tomato sauce:	94.5	26.3
Tomatoes/tomato paste	*66.2*	*70*
Onions	*28.3*	*30*
Salt	1.0	0.3
Vegetable oil	9.0	2.6
Total	360	100

Table 2. Overview of the data sources used in the study

Stage	Detail	Reference	Data origin
Raw materials	British conventional & organic chicken	Williams et al., (2006)	UK
	Brazilian conventional chicken	Prudêncio da Silva et al., (2010)	Brazil
	British conventional & organic tomatoes	Williams et al., (2006)	UK
	Spanish conventional tomatoes	Anton et al., (2005)	Spain
	British conventional & organic carrots	Nielsen et al., (2003)	Denmark
	British conventional onions	Nielsen et al., (2003)	Denmark
	British conventional peas[a]	Milà i Canals et al., (2008)	UK
	Tomato paste	EC (2006); FAO (2009)	Spain
	Slaughterhouse	Nielsen et al., (2003)	Denmark
	Polypropylene crate	Brunel (2008)	UK
	Shopping bags	Brunel (2008)	UK
	Cardboard box	Brunel (2008)	UK
	Pallet	Brunel (2008)	UK
RDCm	Fresh pre-processing	Milà i Canals et al., (2008)	UK
Manufacturing	Ready-made meal	UK manufacturer 2010[b]	UK
	Emissions from food manufacture	EC (2006)	EU
RDCp	Energy consumption	Brunel (2008)	UK
Retail	Supermarket details	Brunel (2008)	UK
Consumption (meal preparation)	Microwave and oven electricity; water consumption	Nielsen et al., (2003); ecoinvent (2009)	UK
Waste	Pre-processing (RDCm)	Milà i Canals et al., (2008)	
	Manufacture	BIS (2011)	
	RDCp	Brunel (2008)	
	Retail	Brunel (2008)	
	Consumption	WRAP (2009)	
Waste management	Food landfilling	ecoinvent (2009)	-
	Incineration of chicken residues	Nielsen et al., (2003)	-
	Landfill of paper and cardboard	PE (2011)	
	Landfill of wood	PE (2011)	-
	Landfill of plastics (PP, HDPE)	PE (2011)	-
Transport	Road transport (diesel vehicles)	ecoinvent (2009)	-
	Bulk sea carrier	ecoinvent (2009)	-

[a] Green beans used as proxy due to lack of data
[b] Confidential data

3. Results and discussion

The life cycle impacts of the ready- and home-made meals are shown in Fig. 2. The results indicate that the impacts of preparing the meal at home are on average 2.7 times lower than that of the ready-made meal, ranging from 10% for GWP to 17 times for ODP in favour of the home-made meal (note that the average is skewed by a high difference in the ODP; if ODP is not considered, the average difference is 1.23 times). The main reason for this is the avoidance of leakage from the refrigerated storage and the lower amount of food waste in the home-made meal system.

The raw materials, distribution and manufacture of the ready-made meal are the most important contributors to most impacts from this system. For example, as illustrated in Fig. 3, the raw materials and distribution each contribute around 25% to the GWP of the ready-made meal, followed by pre-processing and meal manufacturing (~17% each); packaging contributes around 7%. Meal preparation at home has a relatively small impact (2%). Among the ingredients, chicken and tomato are the main hotspots, contributing 90% to GWP from the raw materials stage.

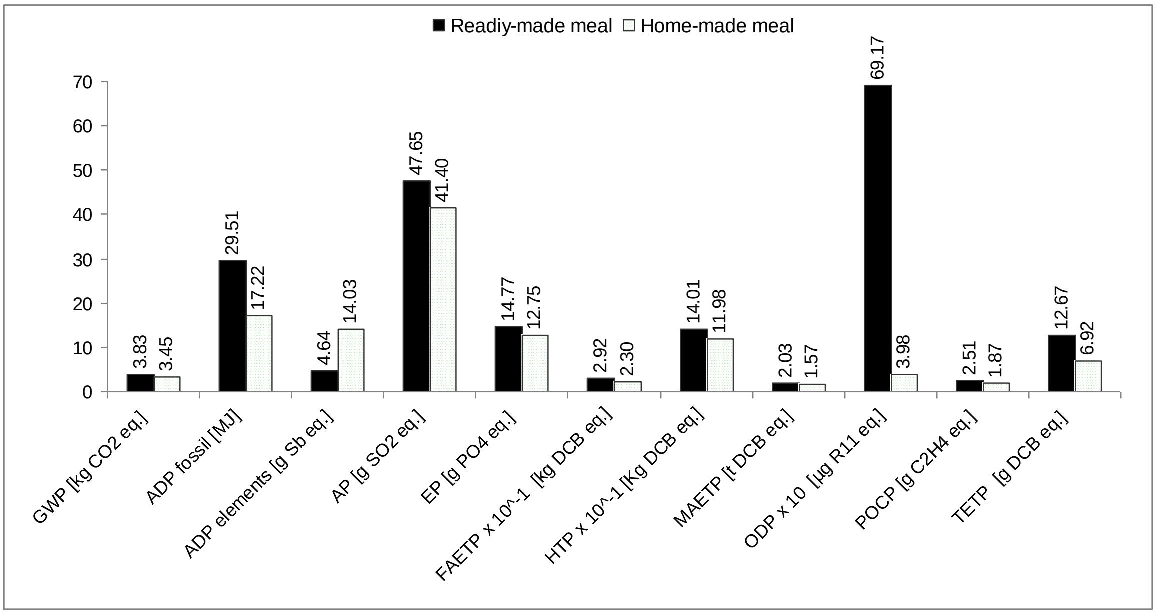


Figure 2. Environmental impacts of the ready- and home-made meals

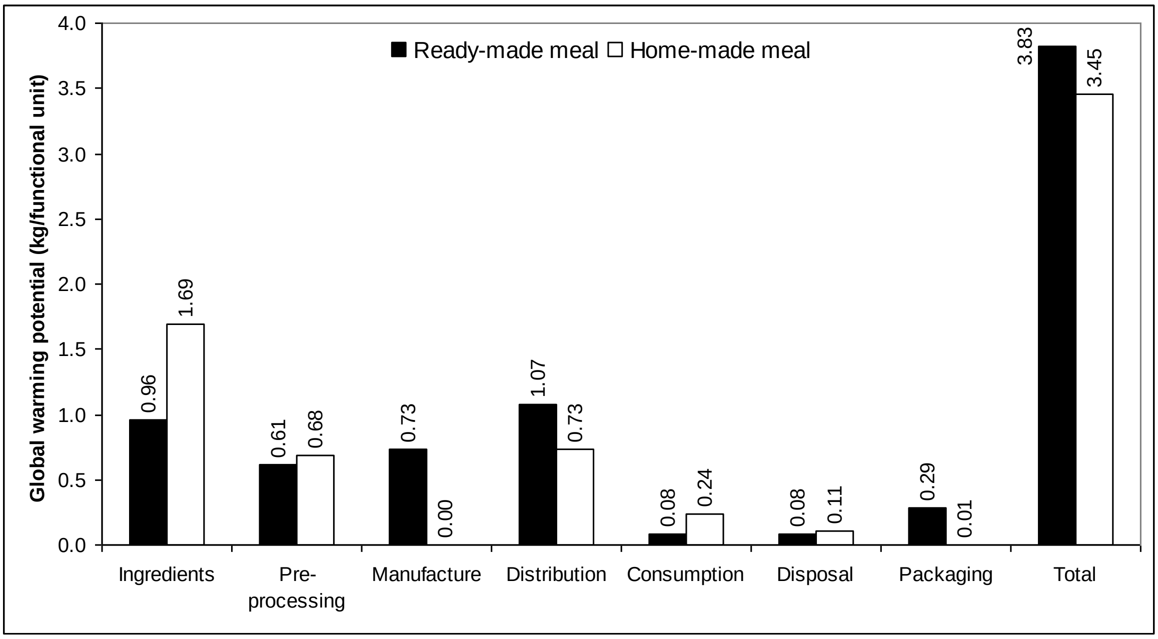


Figure 3. GWP hotspots for the ready- and home-made meals

4. Conclusions

This study has examined the life cycle environmental impacts of a typical ready-made meal produced and consumed in the UK. The results suggest that preparing the same meal at home is environmentally more sustainable than the ready-made meal when the same ingredients are used: the difference in the impacts is on average 2.7 times in favour of the home-made meal (or 1.23 times if the difference in ozone layer depletion is not considered). The main reason for this is the avoidance of refrigerant leakage and lower amount of food waste for the home-made meal system.

The main hotspots for both types of meal are the ingredients and for the ready-made meal, refrigeration. The contribution of packaging and meal preparation at home is relatively small, although the latter is more significant for home- rather than ready-made meals.

5. References

Anton, A., Montero J. I., and Muñoz, P., 2005. LCA and Tomato Production in Mediterranean Greenhouses. Int. J. Agriculture Resources Governance and Ecology 4(2).

Berlin, J., and. Sund, V, 2010. Environmental Life Cycle Assessment (LCA) of Ready Meals. SIK Report No. 804 2010. http://www.sik.se/archive/pdf-filer-katalog/SR804.pdf.

BIS, 2011. Global Food and Farming Futures, Synthesis Report C7: Reducing Waste. The Government Office for Science, Department for Business, Innovation and Skills, London. http://www.bis.gov.uk/assets/bispartners/foresight/docs/food-and-farming/synthesis/11-627-c7-reducing-waste.pdf.

Brunel, U., 2008. Greenhouse Gas Impacts of Food Retailing. Report F0405, DEFRA, London.

Calderón, L. A., Iglesias, L., Laca, A., Herrero, M. And Diaz., M., 2010. The Utility of Life Cycle Assessment in the Ready Meal Food Industry. Resources Conservation and Recycling 54(12): 1196-1207.

Carlsson-Kanyama, A. 1998. Climate Change and Dietary Choices – How Can Emissions of Greenhouse Gases from Food Consumption be Reduced? Food Policy, 23(3/4): 277-293.

Davis, J. and U. Sonesson (2008). Life Cycle Assessment of Integrated Food Chains - A Swedish Case Study of Two Chicken Meals. Int. J. LCA 13(7): 574-584.

Davis, J, Sonesson, U, Baumgartner, D, and Nemecek, T. (2009). Environmental Impact of Four Meals with Different Protein Sources: Case Studies in Spain and Sweden. Food Research Int. 43(7): 1874-1884. http://archive.defra.gov.uk/foodfarm/growing/organic/standards/pdf/guidance-document-jan2010.pdf.

EC 2006. Reference Document on Best Available Techniques in the Food, Drink and Milk Industries, European Commission. pp163-165. August 2006. http://ftp.jrc.es/eippcb/doc/fdm_bref_0806.pdf.

Ecoinvent 2009. Ecoinvent Database. Swiss Centre for Life Cycle Inventories, http://www.ecoinvent.ch.

Espinoza-Orias, N., Stichnothe, H., and Azapagic, A., 2010. Carbon Footprint of Ready-Made Chilled Meals. Proc. 1st IIR International Cold Chain Conference Sustainability and the Cold Chain, 29-31 March 2010, Cambridge.

FAO 2009. Agribusiness Handbook: Fruit and Vegetable Processing. FAO Investment Centre Division and FAO's Rural Infrastructure and Agro-Industries Division. Rome, Italy, http://www.fao.org/docrep/012/al177e/al177e.pdf.

Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, De Koning A. Life cycle assessment: an Operational Guide to the ISO Standards; Part 2a. Institute Oo Environmental Sciences, University of Leiden; 2001.

ISO 2006. ISO 14040, Environmental Management – Life Cycle Assessment: Principles and Framework. International Standards Organisation, Geneva.

Milà i Canals, Ll., Muñoz, I., Hospido, A., Plassmann, A. and McLaren, S., 2008. Life Cycle Assessment (LCA) of Domestic vs Imported Vegetables: Case studies on Broccoli, Salad Crops and Green Beans. Centre for Environmental Strategy, University of Surrey. http://www.surrey.ac.uk/ces/files/pdf/0108_CES_WP_RELU_Integ_LCA_local_vs_global_vegs.pdf.

Milà i Canals, L., Azapagic, A., Doka, G., Jefferies, D., King, H., Mutel, C., Nemecek, T., Roches, A., Sim, S., Stichnothe, H., Thoma, G., and Williams, A., 2011. Approaches for Addressing LCA Data Gaps for Bio-based Products. *J Industrial Ecology*, 15(5): 707-725.

Millstone, E. and Lang, T., 2008. The Atlas of Food: Who Eats What, Where, and Why. Earthscan, London.

Nielsen, P. H., Nielsen, A. M., Weidema, B. P, Frederiksen, R. H., Dalgaard, R., Halberg, N., 2003. LCA Food Database. http://www.lcafood.dk.

Prudêncio da Silva, V., H.M.G. van der Werf and S. R. Soares (2010). LCA of French and Brazilian Broiler Poultry Production Systems. Proceedings of LCA Food 2010 (Vol.1): 475-480.

PE International 2011. Gabi Software and Database. http://www.gabi-software.com.

Sonesson, U., Mattsson, B., Nybrant, T., and Ohlsson, T., 2005. Industrial Processing Versus Home Cooking: An Environmental Comparison Between Three Ways to Prepare a Meal. Ambio 34(4-5): 414-421.

Stichnothe, H., Hospido, A., and Azapagic, A., 2008. Carbon Footprint Estimation of Food Production Systems: The Importance of Considering Methane and Nitrous Oxide.

Williams, A. G., Audsley, E., and Sandars, D.L., 2006. Determining the Environmental Burdens and Resource Use in the Production of Agricultural and Horticultural Commodities. Research Project ISO205. http://www.cranfield.ac.uk/sas/naturalresources/research/projects/is0205.html.

Wiltshire, J., Wynn, S., Clarke, J., Chambers, B., Cottrill, B., Drakes, D., Gittins, J., Nicholson, C., Phillips, K., Thoman, R., Tiffin, D., Walker, O., Tucker, G., Thorn, R., Green, A., Fendler, A., Williams, A., Bellamy, P., Audsley, E., Chatterton, J., Chadwick, D., Foster, C., 2008. Scenario Building to Test and Inform the Development of a BSI Method for Assessing GHG Emissions from Food. Research Project Final Report (DEFRA project Code FO0404). Department for Environment, Food and Rural Affairs, London. http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=15650.

WRAP 2009. Household Food and Drink Waste in the UK. WRAP, London. http://www.wrap.org.uk/downloads/Household_Food_and_Drink_Waste_in_the_UK_Nov_2011.106b4a18.8048.pdf.

Zufia, J. and Arana, L. 2008. Life Cycle Assessment To Eco-Design Food Products: Industrial Cooked Dish Case Study. J Cleaner Prod. 16(17): 1915-1921.

POSTER SESSIONS

1. Coupling LCA and GIS for the assessment of greenhouse gas emissions from global livestock production

Alessandra Falcucci[1,*], Giuseppe Tempio[1], Michael Macleod[1], Carolyn Opio[1], Theun Vellinga[2], Pierre Gerber[1]

[1] Food and Agriculture Organization of the United Nations, [2] Wageningen Livestock Research, The Netherlands, [*] Corresponding author. E-mail: *alessandra.falcucci@fao.org*

Usually life cycle assessments (LCA) are produced outside of any spatially explicit context even though the integration with a Geographical Information Systems (GIS) would provide the necessary tools to fully implement a spatially explicit LCA. The few examples available provide only a partial integration among the two, with the GIS used only for specific aspects of the LCA (e.g. land use). We provide an example of a complete integration between LCA and GIS with the general aim of assessing GHG emissions from livestock at the global level. In particular, using a process based approach, we estimated GHG emissions from different compartments, namely: feed production (including cultivation, induced land use change, manufacture of fertiliser and processing and transport), manure management, , enteric fermentation, energy use (embedded and direct), and post-farm emissions to the point of retail. The entire LCA was implemented in GIS using as inputs spatially explicit layers available at the global level from different sources, and representing the different variables included in the model (e.g. climate, agro-ecological zones, etc.). The approach that we propose has many advantages. It allows for: (1) a global analysis that still maintains a reasonable spatial resolution compared to more traditional national and/or regional analyses; (2) the inclusion of the many spatially explicit variables developed in the last few years by a wealth of international research centres, and; (3) a better integration of variables that are naturally highly variable in time and space (e.g. temperatures, yields, etc.) and that represent the main drivers of important GHG emission sources (such as feed production, manure management). In addition, using a spatially explicit database it is possible to combine, aggregate and/or extract the data depending on the particular question at hand, considering different spatial scales and different administrative regions (or other spatial aggregations). Outputs from the model can also be represented as GHG emissions maps, with a details that is by far greater than a simple country or regional result. Future developments of our approach are possible refining the number of variables and processes to be considered (e.g. including better estimates of transportation distances), improving the resolution and the accuracy of the data considered, and investigating livestock related impacts on nutrient balances, water consumption and biodiversity.

2. Combined mass and economic allocation

Erik Svanes

Department for Preventive Environmental Research, Ostfold Research, Norway, E-mail: *erik@ostfoldforskning.no*

Allocation is a common problem in Life Cycle Assessments. A common allocation situation in food production occurs when a production line produce one or several co-product(s) and by-product(s). Furthermore the distinction between waste and by-product is often unclear because although a waste fraction might be given for free, the transportation of the waste entails costs and the waste can be raw material for products with a commercial value.

Avoidance of allocation using the decision three given in ISO 14044 can be difficult because the processes cannot be split, systems expansion is not possible and no causality can be found. Systems expansion is often difficult because incorporating more functions in a Functional unit is not interesting if the aim is to study one product and systems expansion through substitution is impractical because a single alternative product cannot be identified.

Economic and mass allocation have in the past been the most commonly used allocation methods in food systems but both have their disadvantages. The use of economic allocation can be confusing for consumers, e.g. because the fact that a product of a high commercial value should have a higher environmental impact than a low value product coming from the same raw material and the same process is hard to understand. Mass allocation reflects only physical relationships thus avoiding this problem- One disadvantage of mass allocation is the fact that byproducts of a low value, e.g. fish skin and bones, carries the same environmental burden as the main products. The consequence might be that buyers of these by-products might have less incentive to use this resource. Another problem with mass allocation occurs when a by-product that was previously given for free to users is being sold instead. The result could be a significant shift in the environmental burdens from one assessment to another while no physical changes have been made.

One way to avoid the problems associated with these allocation methods is to combine mass and economic allocation. In the first step economic allocation can be used to distribute environmental impacts between products of very different usage. In the second step mass allocation is used to distribute impacts between products of the same usage. The concept has been investigated using two examples. In one example an animal gives several products going to human consumption, one product going to fertiliser and one to energy production. In another example carrots are sorted into four main products going to human consumption and one waste fraction given for free to be used for animal feed. In these examples the above mentioned disadvantages were avoided. Thus combined allocation proved to a good solution.

ISO 14044 does not seem to preclude such an approach to allocation except maybe the passage that states that the same allocation procedure should be uniformly applied in the assessment. This could, however, be interpreted to mean that consistency is required in similar situations, not that one single allocation method is applied.

References

Schau EM and Fet AM (2008). LCA Studies of Food Products as Background for Environmental Product Declaration. Int J LCA 13 (3) 255–264.

Ayer N, Tyedmers PH, Pelletier NL, Sonesson U and Scholz A (2007): Co-Product Allocation in Life Cycle Assessments of Seafood Production Systems: Review of Problems and Strategies. Int J LCA 12 (7) 480–487.

3. Animal- and environmentally friendly beef production: a conflict?

Martina Alig*, Florian Grandl, Johanna Mieleitner, Thomas Nemecek, Gérard Gaillard

Agroscope Reckenholz-Tänikon Research Station ART, Reckenholzstrasse 191, 8046 Zürich, Switzerland, * Martina Alig. E-mail: *martina.alig@art.admin.ch*

Meat production faces many challenges today, like consumer demand for animal friendly production systems, the food supply of a growing world population and the diminishment of the environmental impacts of meat production. In order to provide a solid basis for decisions, the impacts of different production systems have to be quantified. This contribution presents a study of different beef production systems in Switzerland and in countries exporting to Switzerland as basis for the design of purchasing strategies. The data were derived from model farms of the project LCA-FADN (Hersener et al., 2011). For Switzerland, three production systems were analysed: A conventional bull fattening system and two animal friendly suckler cow systems (a conventional and an organic one). Additionally, a conventional bull fattening system in Germany and a very extensive suckler cow system in Brazil were analysed. The functional unit was kg meat ready for sale at point of sale.

The results show that for all systems the agricultural stage dominates the environmental impacts (Fig. 1). The most important contributions are the application of fertilisers and from field emissions, animal emissions and for the bull fattening systems the purchase of concentrates. Comparing the Swiss systems, the most intensive production system (conventional bull fattening) has the lowest impacts per kg live weight (LW) for most categories analysed (see also Alig et al., 2011). Exceptions are the categories deforestation, ecotoxicity and resource use potassium, where the conventional bull fattening system has the highest results, as well as resource use phosphorus (2nd highest results). This is due to the use of concentrate feeds with soy beans from Brazil compared to the mainly grass-based feeding in the suckler cow systems. The reason for the higher environmental impacts of the suckler cow systems is the general design of the production system itself: in a suckler cow system, the mother cow only serves to produce meat, whereas in a conventional bull fattening system the parent animal produces milk and meat. Therefore, its environmental impacts are allocated between these two products, whereas in the suckler cow systems, the full environmental load of the mother cow is allocated to meat production. This is especially apparent with methane emissions, which are more than 60% higher in the suckler cow systems than in the conventional bull fattening system (Fig. 2).

The German bull fattening system is mostly similar to or a little bit lower than the Swiss conventional system. It is more intensive than the Swiss system, i.e. more based on concentrates and maize silage. The Brazilian production system is a special case: it is very extensive, uses almost no external inputs but huge land area. This influences the results: in categories linked with the use of external inputs as energy demand, resource use or ecotoxicity the Brazilian system has very low impacts. On the other hand, it has very high results for land competition and deforestation and therefore also high values for eutrophication. Due to the long fattening period (over two years), water use and methane emissions of this system are also high.

In summary, there are no clear advantages for a certain system. The animal friendly suckler cow systems stand out due to their low use of arable land and their ability to mitigate the competition for food between man and animals. In order to have advantages in categories like energy demand and resource use, a suckler cow system has to be really extensive (e.g. Brazil). In Switzerland, the animal friendly suckler cow systems had overall higher environmental impacts than the conventional bull fattening system. This is to a great deal due to the system design, where the full environmental impact of the mother cow is allocated to the meat production. In order to develop animal and environmental friendly production systems, alternative systems have to be contemplated, e.g. a combined milk and meat production with dual purpose cows.

References

Alig et al., 2011. Ökologische Bewertung von Rind-, Schweine- und Geflügelfleisch. Schlussbericht Forschungsinitiative COOP-ART. In preparation.

Hersener, J.-L., Baumgartner, D.U., Dux, D. (Eds.), 2011. Zentrale Auswertung von Ökobilanzen landwirtschaftlicher Betriebe (ZA-ÖB). Forschungsanstalt Agroscope Reckenholz-Tänikon ART, Zürich/Ettenhausen.

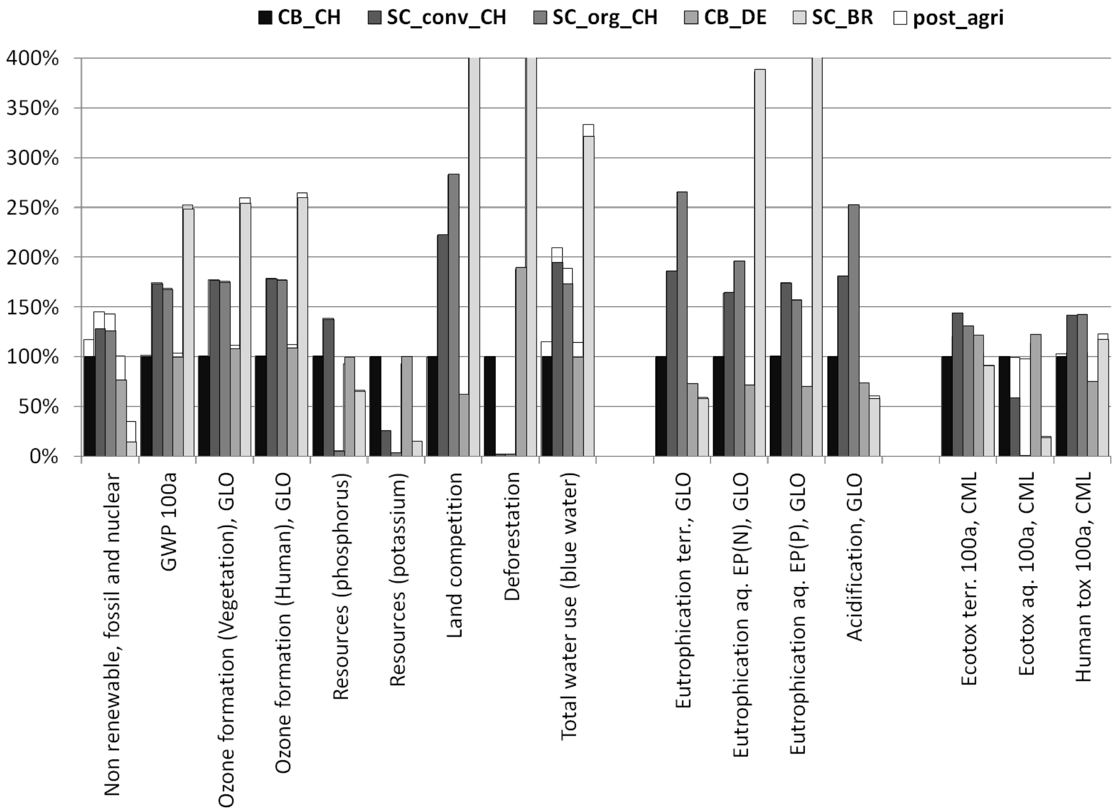


Figure 1. Overview of the environmental impacts per kg meat ready for sale of the five analysed beef production systems. CB_CH: agricultural stage conventional bull fattening Switzerland; SC_conv_CH: agricultural stage conventional suckler cow system Switzerland; SC_org_CH: agricultural stage organic suckler cow system Switzerland; CB_DE: agricultural stage conventional bull fattening Germany; SC_BR: agricultural stage suckler cow system Brazil; post_agri: post-agricultural phase (slaughtering and transports to point of sale).

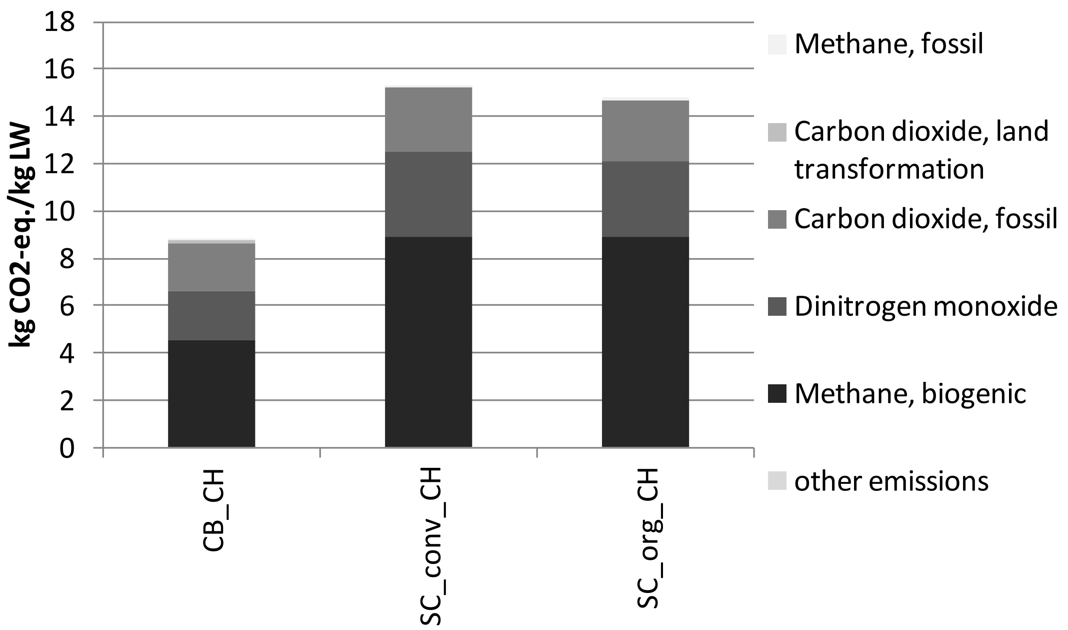


Figure 2. Global warming potential per kg live weight (LW) of the three Swiss beef production systems.

Acknowledgement: These research results were developed by ART with the support of COOP.

4. Allocation procedures in the beef life cycle assessment

Claudio Mazzini[1], Nicola Brina[1], Sara Daga[1], Fabrizio Boeri[2], Massimo Marino[2],Sonia Pignatelli[2,*]

[1] Coop Italia, [2] Life Cycle Engineering, [*] Corresponding author. E-mail: *pignatelli@studiolce.it*

The beef production is one of the food sectors with the highest environmental impact: this is mainly due to the feed production, the manure management and the methane emissions from enteric digestion processes. Even if the high impact is well known among all the LCA practitioners, it is quite important the definition of the hypotheses because some of them are quite relevant for the final result such the functional unit (meat boneless or not), the system boundaries (from where to where), the data quality requirements, etc. Probably, the most important hypothesis needed for the beef LCA is related to the definition of the allocation rules between the many by-products generated along the chain.
Considering the whole chain, for example, aspects that shall be considered are:

- how to consider the impact of reproductive cow used for the generation of calves;
- the allocation rules when the reproductive cow is mainly bred for the milk production; in that case it is necessary to define the portion of impact to be allocated to the veal (by-product of farms that produce milk).
- how to deal when the meat comes directly from milk cow or reproductive cow for calves production at the end of their life;
- how to consider the leather.

After the definition of the rules related to the system analysed and the eventual by-products generated along the chain, the other issue concern the approach used to allocate the impact: economic allocation, mass allocation or other alternative approaches (i.e. biological causality defined as the physiological feed requirements of the animal to produce milk, meat or other by-product).
The aim of this paper is to examine some allocation procedures and to present a sensitivity analysis of the chosen procedure on the final results; for example in Fig. 1 are illustrated the difference, in terms of Carbon Footprint, related to veal production chain considering different allocations rules for the impact of reproductive cow mainly finalised to produce milk.

References

Cederberg C., Stadig M., 2003. System Expansion and Allocation in Life Cycle Assessment of Milk and Beef Production, International Journal of LCA 8, 350-356.

Ponsioen T., Kool A., December 2010. Carbon footprint assessment of calf milk replacer, calves and veal, Blonk Milieuadvies Study.

Blonk H., Ponsioen T., Kool A., Marinussen M., April 2011. The Agri-Footprint method - Methodological LCA framework, assumptions and applied data - Version 1.0, Blonk Milieuadvies Study.

Barilla Center for Food and Nutrition (BCFN), 2011. 2011 Double Pyramid: Healthy Food for People, Sustainable for the planet, Supporting technical paper Version 2 of 14 July 2011.

Casey J. W., Holden N. M., 2006. Quantification of GHG emissions from sucker-beef production in Ireland, Agricultural Systems 90, 79-98.

Ecoinvent Swiss database, v.2.0, www.ecoinvent.ch.

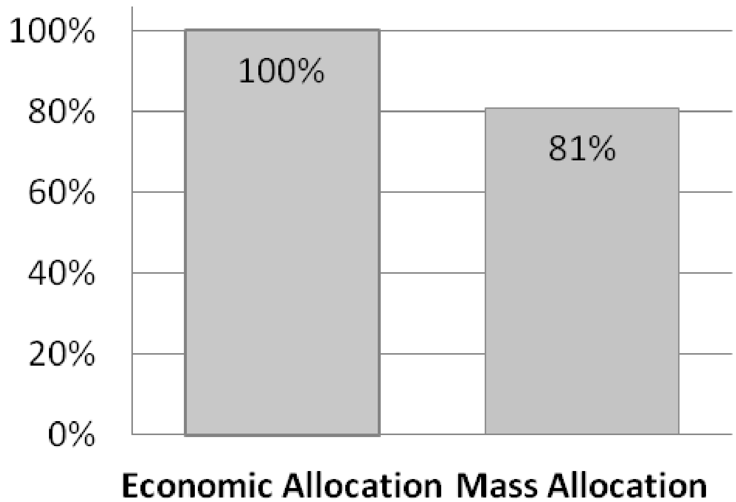


Figure 1. Carbon Footprint of 1kg of veal meat with different allocations rules for the reproductive cow impact.

5. Evaluation of the environmental impact of Belgian beef production systems through life cycle assessment methodology

Loriers Astrid[1,*], Mathot Michael[2], Stilmant Didier[2]

[1] Agence cooperative de service et conseils (ACOSEC scrl), Centre Wallon de Recherches Agronomiques (CRA-W), [2] CRA-W, [*] Corresponding author. E-mail: *a.loriers@cra.wallonie.be*

Environmental issues like GHG emissions, process of eutrophication and acidification, fossils fuels depletion, etc. received increasing attention over the last years by the European politic. In this context, the livestock sector is often pointed out. At the same time, consumers search more “green” products and require more information on food production process and origin.
The aim of this study is to highlight and adapt, based on literature review and on the characteristics of the main beef production systems identified in Belgium, the emission parameters which could be, in LCA (Life Cycle Assessment) methodology, adjusted to our national conditions. Indeed, beef livestock systems are based, in Belgium, on Belgian Blue breed characterised by a very high carcass yield associated to a very good feed conversion ratio.
The impacts taken into account for these processes will be the global warming, eutrophication, acidification, fossils fuels depletion, land use and occupation, and water use. In term of boundaries, the system that will be presented will add, to the classical cradle to farm gate approach, the slaughter house, carcass transformation and packaging processes. We expect that this analysis will help identify the best practices to improve environmental performances of this sector in order to advise its different actors.

6. Life cycle assessment on dairy and beef cattle farms in France

Jean Baptiste Dollé[1,*], Armelle Gac[2], Vincent Manneville[3], Sindy Moreau[4], Elise Lorinquer[2]

[1] Institut de l'Elevage, Building and Environment Department, F-62051 St Laurent Blangy, France, [2] Institut de l'Elevage, Building and Environment Department, F-35652 Le Rheu, France, [3] Institut de l'Elevage, Building and Environment Department, F-63170 Aubière, France, [4] Institut de l'Elevage, Building and Environment Department, F-69364 Lyon, France, [*] Corresponding author. E-mail: *jean-baptiste.dolle@idele.fr*

In the current environmental context, as much as political (reduction of greenhouse gas emissions, preservation of biodiversity) and social (consumer demands for information concerning food products), a need to determine the influence of ruminant livestock on environment is incontestable. It is now crucial to quantify precisely the environmental impacts for different ruminant livestock systems by using Life Cycle Assessment (LCA). The French Livestock Institute has launched a work program to determine the environmental assessment of dairy and beef cattle systems at farm scale. In this context, a methodology based on life cycle assessment method has been built to assess many impacts which concern climate change, eutrophication, acidification and energy consumption. This methodology has been applied to several beef and dairy cattle systems from the French Breeding Network database (208 dairy farms and 268 beef farms) representative of the French cattle. Five types of dairy system, defined by farm typology based on part of maize in farm and location area, have been studied (Table 1). Milk gross carbon footprint varies from 0.8 to 1.5 kg CO_2eq/kg of milk produced and net carbon footprint vary from 0.1 to 1.4 kg CO_2eq including carbon sequestration. Concerning acidification, eutrophication and energy consumption for the five systems studied, the variation goes respectively from 0.005 to 0.010 kg SO_2eq, 0.001 to 0.0010 kg PO_4eq and 1.2 to 4.0 MJ, all expressed in kg of milk produced. Differences between systems are not very high, however the systems located in plain area contain more than 30% maize in the diet and present a higher productivity per cow, that result in a higher risk of eutrophication and a higher net carbon footprint than the other systems. In beef production, three specialised suckler-cattle systems have been studied (table 2), calf-to-weanling system producing weaners (9-10 months old), calf-to-beef system producing beef steers (over 30 months old) and calf-to-beef system producing young bulls (17 months old). French suckler cattle farm systems produce from 8.7 to 26.0 kg CO_2eq/kg of live weight. Calf-to-beef system producing beef steers, fattened on pasture, has the lower net carbon footprint (5.9 kg CO_2eq/kg of live weight) considering carbon sequestration and a lower risk of eutrophication. The energy used is quite similar for the three systems. In most cases, the intra-system variability of environmental footprints is higher than inter-system variability. The intra-system variability is related to technical and practices efficiency on farms. At equivalent systems, an important difference can be observed on the final impact between optimised and non-optimised farms. These differences are due to herd management, cultural practices, feed and fertiliser strategies, etc. For example, in relation with the nitrous oxide emissions, the most optimised farms, which consume less feed, less fertiliser, etc. have better nitrogen balance and lower environmental impacts. This study show link between environmental issues and highlight the relation between environmental issues and practices on farm, which propose some ways of mitigation adapted to the production systems. Finally, these investigations demonstrate that numerous mitigation actions can be identified in the livestock systems to reduce the environmental footprint of milk and beef meat at the farm gate. Some of them concern management practices (adjustment of dietary intake, fertilisation management, etc.) which result in substantial savings in agricultural expenses. Others require installation of new technologies which would require additional funds to improve the production processes.

References

De Vries M., De Boer I.J.M., 2010. Comparing environmental impacts for livestock products: A review of cycle assessments. Livestock Science, 120, 1-11.

Gerber P., Vellinga T., Opio C., Steinfeld H., 2011. Productivity gains and greenhouse gas emissions intensity in dairy systems. Livestock Science, doi: 10.1016.j.livsci.2011.03.012.

Veysset P., Lherm M., Bébin D., 2010. Energy consumption, greenhouse gas emissions and economic performance assessments in French Charolais suckler cattle farms: Model-based analysis and forecasts. Agricultural Systems, 103, 41-50.

Table 1. Environmental impact of milk on dairy farms

	Production system				
	Plain > 30% maize	Plain 10-30% maize	Plain < 10% maize	Mountain maize	Mountain grass
Number of farms	37	36	50	26	59
	Average *<min-max>*	Average *<min-max>*	Average *<min-max>*	Average *<min-max>*	Average *<min-max>*
Gross carbon footprint kg CO_2eq/ kg milk	1.06 *0.8-1,3*	1.1 *0.8-1.4*	1.1 *0.7-1.4*	1.1 *0.8-1.5*	1.2 0.*9–1.5*
Net carbon foot-print kg CO_2eq / kg milk	1.0 *0.7-1.3*	0.8 *0.6-1.4*	0.7 *0.4-1.1*	0.9 *0.6-1.3*	0.7 *0.1-1.0*
Acidification kg SO_2eq / kg milk	0.007 *0.005-0.009*	0.007 *0.005-0.011*	0.009 *0.005-0.012*	0.008 *0.005-0.013*	0.010 *0.006-0.013*
Eutrophication kg PO_4eq / kg milk	0.006 *0.003-0.011*	0.004 *0.002-0.009*	0.003 *0.001-0.010*	0.005 *0.002-0.010*	0.004 *0.002-0.008*
Energy con-sumption MJ / kg milk	2.5 *1.8-3.8*	2.2 *1.2-3.3*	2.4 *1.2-3.9*	2.6 *1.7-3.8*	2.7 *1.8-4.0*

Table 2. Environmental impact of meat on beef farms

	Production system		
	Calf-to-weanling system producing weaners (9-10 months old)	Calf-to-beef system producing beef steers (> 30 months old)	Calf-to-beef system pro-ducing young bulls (17 months old)
Number of farms	163	13	72
	Average *<min-max>*	Average *<min-max>*	Average *<min-max>*
Gross carbon footprint kg CO_2eq / kg meat	14.7 *10.9-25.4*	13.2 *9.5-18.8*	13.5 *8.7-18.4*
Net carbon foot-print kg CO_2eq / kg meat	7.7 *0.3-17.3*	5.9 *3.8-8.5*	9.9 *5.6-15.0*
Acidification kg SO_2eq / kg meat	0.120 *0.078-0.217*	0.109 *0.069-0.156*	0.115 *0.062-0.194*
Eutrophication kg PO_4eq / kg meat	0.039 *0.019-0.113*	0.028 *0.016-0.048*	0.052 *0.021-0.130*
Energy con-sumption MJ / kg meat	20.4 *5.6-35.6*	21.0 *10.8-38.7*	20.6 6.9-34.1

7. Life cycle assessment of four fattening calves systems in Spain

Sandra Lartategui Arias[1,*], José Soler Rovira[2]

[1]Global Food Quality, Campus Montegancedo Parque Científico y tecnológico, Centro empresas Local 12 28223 Pozuelo de Alarcón, Madrid, Spain, [2]Escuela de Ingeniería Técnica Agrícola. Universidad Politécnica de Madrid. Ciudad Universitaria, s/n 28040 Madrid, Spain, [*]Corresponding author. E-mail address: *slartategui@globalfq.com*

The livestock sector is a very important element of stress for many ecosystems and for the entire planet (FAO, 2009). In Spain, livestock production contributes in a proportion of about 5% of the total equivalent CO_2 (De Blas, et al. 2008). Although intensive feedlot system is the most widespread in Spain, there are other systems (extensive and semi-extensive) whose environmental impact is interesting to assess.

We used the LCA methodology to asses four fattening systems of calves used in Spain. Two of these systems were of intensive fattening, where calves fed on concentrate of cereals; other system was of extensive fattening, calves fed on grass, and the last one was of semi-extensive fattening, with calves fatted in two phases, at the first one fed with grass and at the second one (named "finishing") with grain concentrate.

The objectives of this LCA study were firstly identify which unit processes generate more environmental impacts, and secondly compare the environmental impact of the those four systems of fattening. For each system of fattening were analysed: i) production of raw materials (soya, palm oil, grass, etc) used in the feedstuff, including fertilisation and tillage; ii) transport of raw materials to the farm where calves are fattened; iii) the fattening of calves on the farm, including enteric methane emissions and emissions from manure. Impact categories analysed were global warming potential (GWP), acidification potential (ACP) and eutrophication potential (EUP). Literature data of specific on site systems, transport calculation and different databases and references were used (Lartategui-Arias, 2010). The functional unit considered was ton of meat for consumption, once removed the entrails, skin, head, legs and fat coverage. The system limits of the study are shown in Fig. 1.

According to data from the agricultural process, fertilising and tilling, is the most polluting in GWP category in intensive farms (Table 1). At extensive and semi-extensive farms, fattening of calves are the most polluting in this category. In EUP category, fattening activity is the most polluting in three of four systems analysed, the two intensive and semi-extensive. In ACP, fattening is the most polluting activity in the four systems analysed. Adding all the unitary processes, semi-extensive system is the most polluting in EUP and ACP categories, primarily due to the process of fertilisation of the raw materials forming the feedstuff of the finish phase and fattening process. In GWP, the extensive system is the most polluting, primarily due to the fattening process, in which enteric methane emissions play an important role.

References

De Blas, C., García-Rebollar, P., Cambra-López, M. y Torres, A.G. 2008. El cebo de terneros en España, una actividad respetuosa con el medioambiente. Estudio de la repercusión de la producción de vacuno de carne en la emisión de gases con efecto invernadero. ASOPROVAC. Madrid, Spain.

FAO, 2009. La larga sombra del ganado, problemas ambientales y opciones. Roma. ISBN: 978-92-5-305571-5.

Lartategui-Arias S. 2010. Aplicación de la metodología de Análisis de Ciclo de Vida para la evaluación ambiental del cebo de terneros en España. Graduate Thesis, Universidad Politécnica de Madrid, Spain.

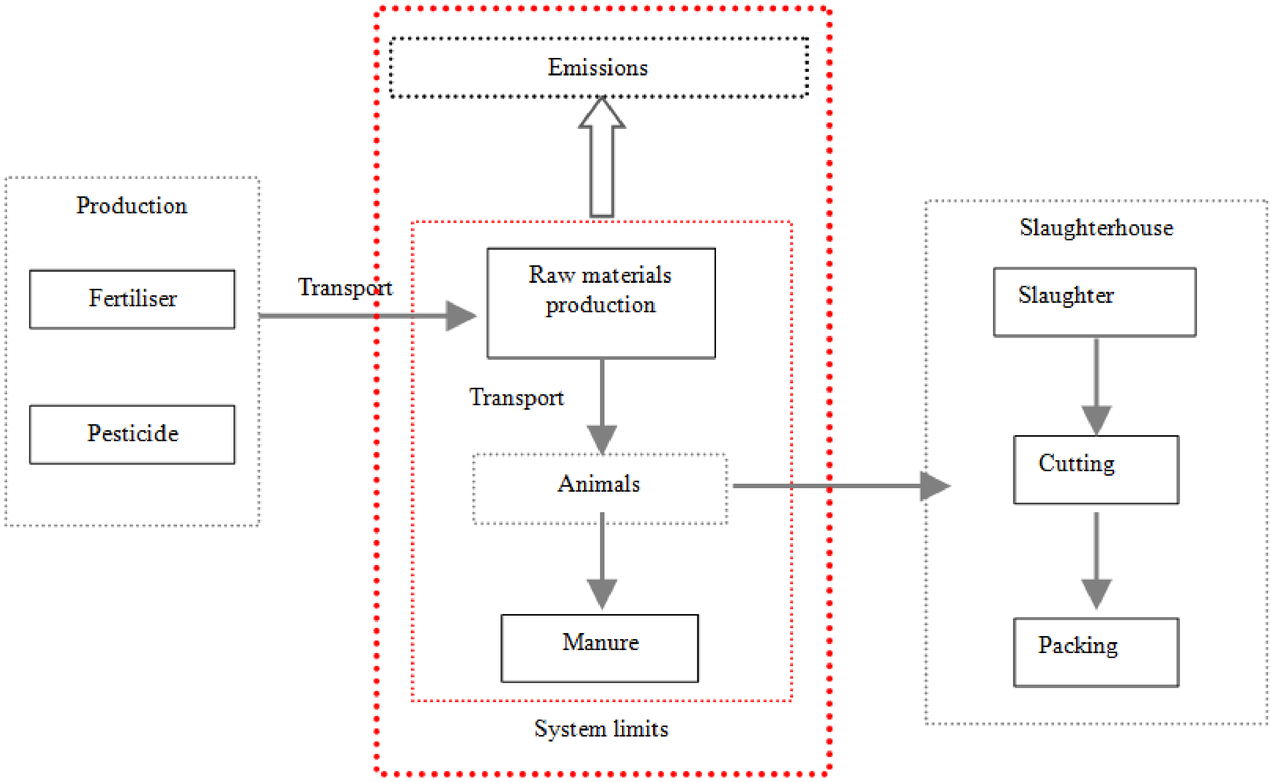


Figure 1. System limits of the LCA.

Table 1. Results of the LCA for the four fattening systems in GWP, EUP and ACP impact categories.

Unitary process	Fattening system	GWP (CO_2-eq)	EUP (PO_4-eq)	ACP (SO_2-eq)
		Emissions kg/t meat		
Fertilisation	Intensive 1	**3504.20**	**25.16**	4.44
	Intensive 2	2962.00	7.18	3.87
	Extensive	1444.00	6.33	3.39
	Semi-extensive	2810.00	23.53	**12.42**
Tillage	Intensive 1	53.70	**0.44**	**2.27**
	Intensive 2	107.00	0.33	1.85
	Extensive	41.80	0.13	0.72
	Semi-extensive	**109.00**	0.34	1.88
Transport	Intensive 1	**254.60**	**0.81**	**4.34**
	Intensive 2	153.00	0.45	2.42
	Extensive	30.60	0.05	0.25
	Semi-extensive	151.00	0.23	1.24
Fattening	Intensive 1	2372.10	21.55	11.13
	Intensive 2	1838.00	18.68	9.64
	Extensive	**6499.30**	20.26	10.47
	Semi-extensive	4319.00	**28.51**	**14.72**
Total	Intensive 1	6184.60	47.96	22.18
	Intensive 2	5060.00	26.64	17.78
	Extensive	**8015.70**	26.77	14.83
	Semi-extensive	7389.00	**52.61**	**30.26**

8. Environmental impact of beef – role of slaughtering, meat processing and transport

Johanna Mieleitner*, Martina Alig, Florian Grandl, Thomas Nemecek, Gérard Gaillard

Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland, * Corresponding author. E-mail: *johanna.mieleitner@art.admin.ch*

Environmental impact of meat production is widely discussed and of increasing importance to customers and stakeholders. To find mitigation strategies it is important to identify important processes in the whole production chain from the agricultural stage up to the retail store. Several studies found that the environmental impact of meat is dominated by animal production (Roy et al. 2012, Foster et al. 2006). But is this also the case if meat products are transported over longer distances? Knowledge about the role of the processes after the animal production stage is important, especially when comparing domestic production with imports.
The aim of this study is to assess the environmental impact of beef produced in Switzerland and to compare it with beef imported from Germany and Brazil. Emphasis is set on the role of the processes after the animal production stage: slaughtering, meat processing and transport. The life cycle of beef is analysed from cradle to the sales point and the functional unit is 1 kg of meat ready for sale (packed). The agricultural phase is described by Alig et al. (2012). For slaughtering, meat processing and transport data from industry (meat production and retail business) and from literature are used.
The environmental impact of beef produced and sold in Switzerland is dominated by animal production, which is responsible for over 80% of all investigated environmental impacts. The stages after animal production account for around 15% of the impact categories 'non-renewable energy demand' and 'blue water use' and for less than 5% of all other environmental impacts (Fig. 1). The most important process within these post-agricultural stages is slaughtering and meat processing which contributes up to 15% to the total impact. The transport of living animals and of processed meat and the distribution centres contribute less than 2% to all impact categories (Fig. 1).
Beef imported from Germany is also dominated by animal production. Due to longer distances, transport has a slightly higher impact but still contributes less than 5% to all impacts (Fig. 2).
Beef imported from Brazil by ship is dominated by the agricultural production for most environmental impacts, despite the higher impact of transport, as the impact of transport by freight ship is relatively low. This is different for beef imported by aircraft. Here the transport from Brazil to Europe has an important impact (Fig. 2). E.g. it accounts for around 15% of the global warming potential and over 80% of the non-renewable energy demand.
Looking at the environmental impact of slaughtering and meat processing, the main contribution to most impact categories comes from direct energy use. Also important factors are packaging film, water use, sewage treatment and waste disposal.
In conclusion, animal production dominates the production chain for beef produced in Switzerland and imported from Germany and Brazil. An exception is the import of beef by aircraft from Brazil. Animal production is therefore the most important starting-point for mitigation strategies. However it is also important to reduce the environmental impact of the post-agricultural stages. All stakeholders along the whole production chain have to contribute to maximise the overall mitigation potential. Transport by airplane can have a high impact and should be avoided. Another important factor is the use of non-renewable energy during slaughtering and meat processing which could for example be reduced by replacing fossil fuels with renewable energy sources. *Acknowledgement: These research results were developed by ART with the support of COOP.*

References

Alig, M., Grandl, F., Mieleitner, J., Nemecek, T., 2012. Animal and environmental friendly beef production: a conflict? Abstract, 8th International conference on LCA in the Agri-Food sector.

Foster, C., Green, K., Bleda, M., Dewick, P., Evans, B., Flynn, A., Mylan, J., 2006. Environmental impacts of food production and consumption. A Final Report to the Department for Environment, Food and Rural Affairs, Manchester Business School, Defra, London.

Roy, P., Orikasa, T., Thammawong, M., Nakamura, N., Xu, Q., Shiina, T., 2012. Life cycle of meats: An opportunity to abate the greenhouse gas emission from meat industry in Japan. Journal of Environmental Management 93, 218-224.

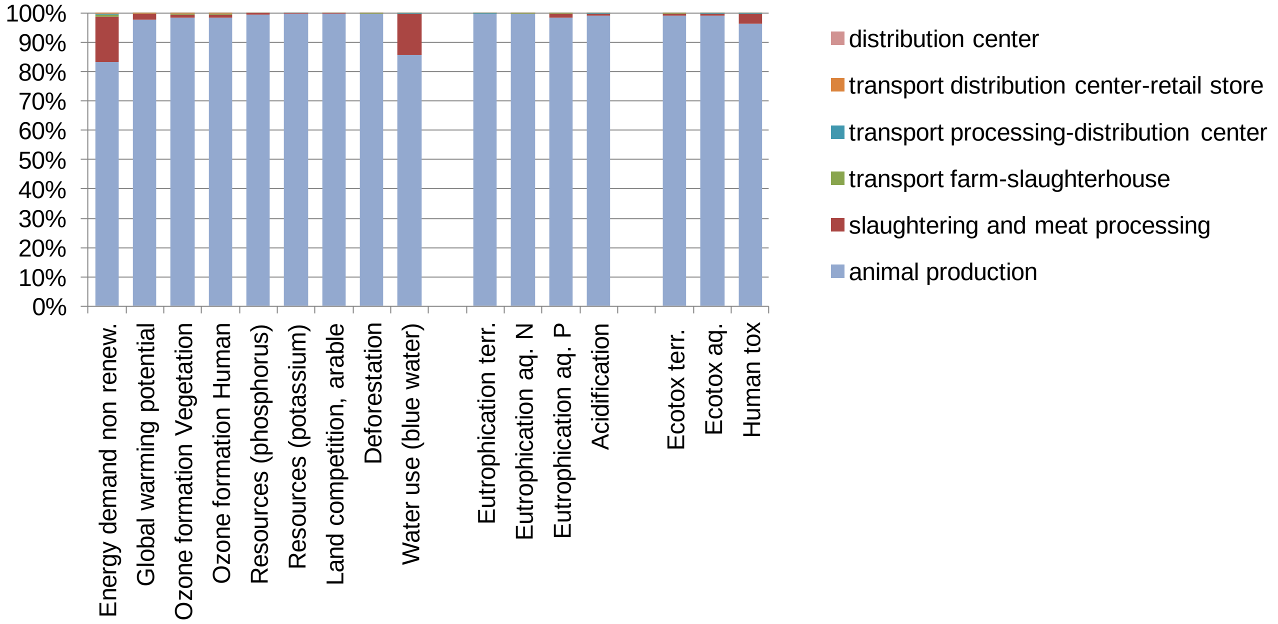


Figure 1. Environmental impacts per kg meat ready for sale for beef produced and sold in Switzerland.

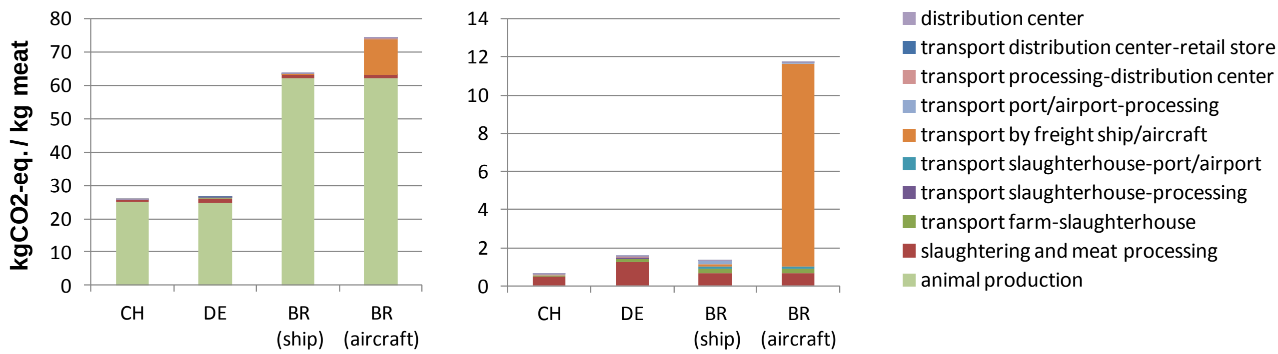


Figure 2. Global warming potential (GWP) per kg meat ready for sale at the point of sale in Switzerland. Left: Whole Chain. Right: slaughtering, meat processing and transports. CH: animal production in Switzerland, transport by lorry, DE: animal production in Germany, transport by lorry, BR: animal production in Brazil, transport by freight ship (ship) and transport by aircraft (aircraft).

9. Life cycle assessment of Mediterranean buffalo milk

Giacomo Pirlo[1], Sara Carè[1], Valentina Fantin[2,*], Francesca Falconi[3], Patrizia Buttol[2], Giuseppina Terzano[4], Paolo Masoni[2]

[1]CRA-FLC Agricultural Research Council-Fodder and Dairy Productions Research Centre, Via Porcellasco 7, 2610, Cremona, Italy, [2]ENEA - Italian National Agency for New Technologies, Energy and Sustainable Economic Development, Via Martiri di Monte Sole 4, 40129 Bologna, Italy, [3] LCA-lab SRL, Via Martiri di Monte Sole 4, 40129 Bologna, Italy, [4]CRA-PCM Agricultural Research Council, Research Centre for Meat Production and Genetic Improvement, Via Salaria 31, 00016 Monterotondo, Italy, [*] Corresponding author. E-mail: *valentina.fantin@enea.it*

This study is part of a broader project carried out by CRA (Agricultural Research Council), in collaboration with ENEA, aimed at evaluating and identifying environmentally friendly livestock models, which could be applied to Italian agri-food production systems, in order to improve their environmental sustainability.

In Italy there are about 365 thousands Mediterranean Buffaloes. This population increased considerably in recent years, as consequence of the strong worldwide demand of "Mozzarella di bufala campana – DOP" (AIA, 2011).

At present no LCA study on buffalo milk production exists in literature, whereas many LCA studies have been performed on cattle milk production, both on farming systems and on the entire life cycle. In this study the standard ISO Life Cycle Assessment and ILCD Handbook methodology (ILCD, 2010) have been applied to the production of two buffalo dairy farms located in Southern Italy. The goal is to evaluate their environmental performance and to identify the hotspots in the production chain. The functional unit is 1 kg of Normalised Buffalo Milk at farm gate. An attributional approach has been applied according to the stated goal of the study.

System boundaries (Fig. 1) comprise crop production, as well as the activities related to buffalo feeding, breeding and milking. Specific primary data, referred to 2010, have been collected from the two buffalo farms for each of the above phases. In particular, the following items have been included in the system boundaries: number of producing buffaloes and replacement heifers; production and transport of purchased feeds; production and transport of seeds, fertilisers, and detergents; energy consumption related both to cropping, feeding and milking; disposal and treatment of waste produced at farms. Buildings, infrastructures and equipments have not been included in the system boundaries, but they are included in some database's processes. The production of medicines and the milk-processing phase have not been included. Databases (mainly Ecoinvent) and literature have been used for the background data. Manure and slurry produced by buffaloes are spread as fertilisers on agricultural farms' land. As regards emissions related to the use of chemical and organic fertilisers, N_2O airborne emissions and NO_3^- waterborne emissions have been calculated according to ISPRA (2008). NH_3 airborne emissions have been calculated according to ISPRA (2011a). Phosphorus waterborne emissions have been estimated according to the budget farm gate methodology proposed by Dalgaard et al. (2006). Methane emissions on farms due to enteric fermentation and manure management have been estimated referring to ISPRA (2008). As buffalo milk production at farms is a multifunctional process, the environmental impacts have been allocated between the main product (milk) and co-products (calves and culled buffaloes) on the basis of their economic value.

At the Conference the preliminary results, including the following impact categories, will be presented and discussed: Global warming, Photochemical Oxidation, Acidification, Eutrophication. The assessment of Land Use and Ecotoxicity due to use of pesticides and antibiotics will be performed in a second step of the study.

References

Associazione Italiana Allevatori (AIA), 2011. Quadro economico e relazione attività 2010.

Dalgaard, R., Halberg, N., Kristensen, Ib.,S., Larsen, I., 2006. Modelling representative and coherent Danish farm types based on farm accountancy data for use in environmental assessment. Agriculture, Ecosystems and Environment 117, 223-237.

International Reference Life Cycle Data System (ILCD), 2010. General guide for Life Cycle Assessment-Detailed guidance. European Commission, Joint Research Centre (DG JRC), Available at: http://lct.jrc.ec.europa.eu/assessment/assessment/projects#d (visited in February 2012).

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories: volume 4: Agriculture, Forestry and other Land Use. Intergovernmental Panel on Climate Change. Available at: www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html (visited in February 2012).
ISPRA, 2008. Agricoltura-Inventario nazionale delle emissioni e disaggregazione provinciale. Institute for Environmental Protection and Research. Rome, Italy. Available at www.isprambiente.it.
ISPRA, 2011. Agricoltura-Emissioni nazionali in atmosfera dal 1990 al 2009. Institute for Environmental Protection and Research. Rome, Italy. Available at www.isprambiente.it.

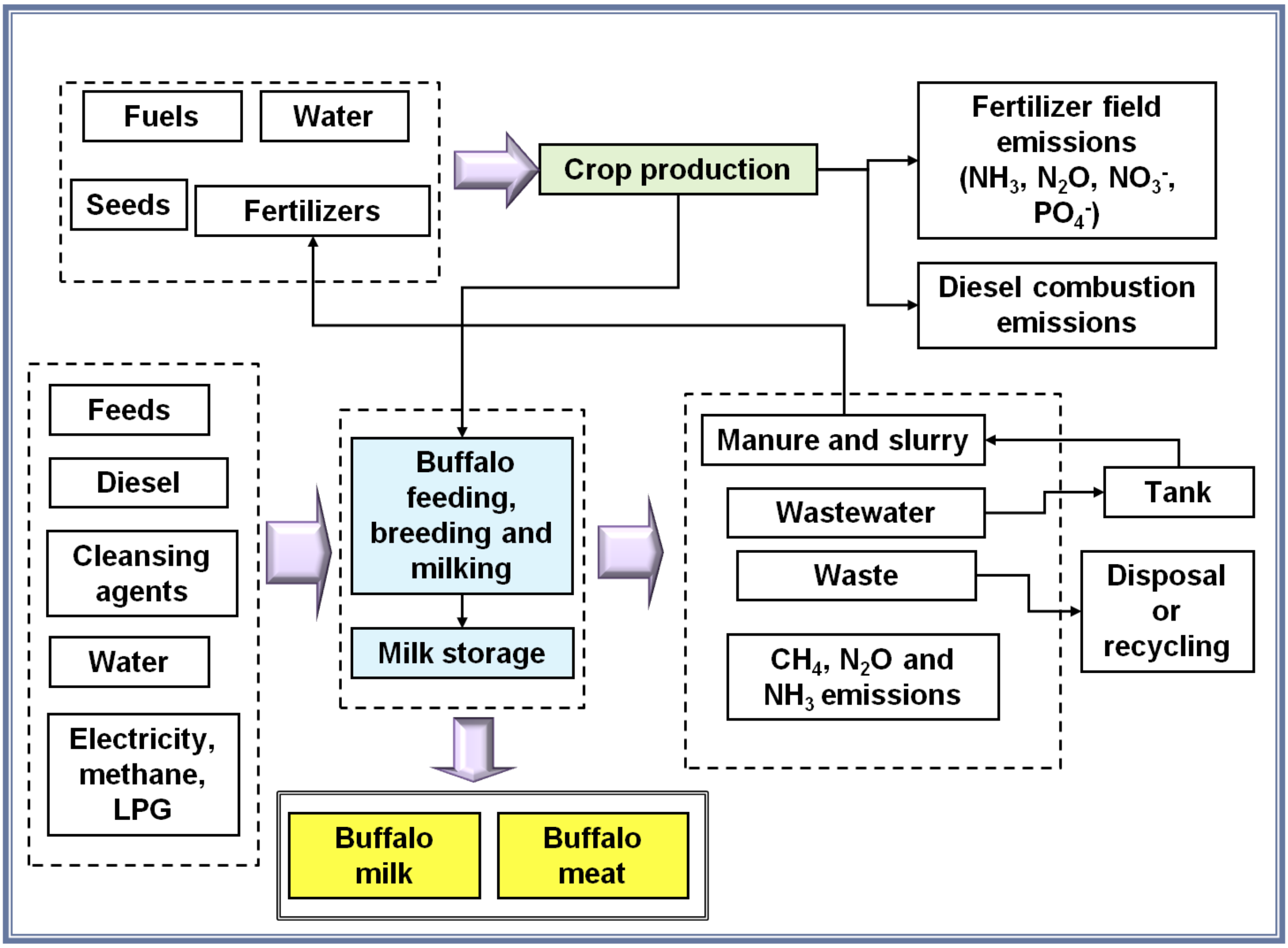


Figure 1. System boundaries.

10. Life cycle assessment of milk production in Italian intensive dairy farms

Matteo Guerci, Maddalena Zucali*, Anna Sandrucci, Alberto Tamburini, Chiara Penati, Luciana Bava

Dipartimento di Scienze Agrarie e Ambientali, Università degli Studi di Milano, Italy, * Corresponding author. E-mail: *maddalena.zucali@unimi.it*

Environmental concerns are having increasing priority upon political, social, and economic agendas, in particular when related to agriculture. Food production has an environmental impact, and as the global populations continue to increase, it is critical to produce sufficient high-quality food from a finite resource supply in order to mitigate the effects upon the environment (Capper et al., 2009). In the North of Italy favorable climatic and infrastructural conditions promoted a great concentration of livestock farms with intensive utilisation of natural resources (i.e. land, air, water). The objective of this study was to assess the environmental impact of milk production in intensive dairy systems, in order to identify farm characteristics that guarantee at the same time low environmental impact and economic sustainability. A cradle to farm gate Life Cycle Assessment (LCA) was performed on 41 intensive dairy farms in Northern Italy. In addition to the evaluation of greenhouse potential, impact categories as acidification, eutrophication, land use and energy use were considered. The functional unit was 1 kg Fat and Protein Corrected Milk (FPCM). Farm key parameters about crop production for feeding, livestock, manure management, purchased feed, fertilisers, pesticides, electricity and fuels, milk and meat sold were collected by personal interview to the farmers. LCA was carried out using LCA software package, SimaPro 7.3.2 (Pré Consultants, 2011). Gross margin, i.e revenues minus the direct production costs, excluding labour costs (expressed in €/t FPCM), was used as economic indicator. Database was analysed using the CLUSTER procedure (SAS, 2000). In order to identify different farming systems the following variables were considered: gross margin, feed self-sufficiency, dairy efficiency and stocking density. Two main clusters of farms were identified (A and B); moreover in each of the two clusters two subgroups of farms were defined (Table 1). Farms from cluster B were slightly less intensive than farms from cluster A: they had significantly larger farm land (ha), lower stocking density (LU/ha) and higher feed self-sufficiency (%). Economic results were similar between the two main clusters but ecological performances were better for farms from cluster B: nitrogen and phosphorus balances at farm gate (kg/ha) and the off-farm components of total climate change, acidification, eutrophication, energy use and land use per kg FPCM were significantly lower in cluster B than in A. In fact farms from cluster B had higher feed self-sufficiency and purchased less feed, reducing off-farms fraction of all impacts but increasing on-farm component of eutrophication, acidification and land use. Farms from cluster B impacted more in term of total eutrophication (on- and off-farm components) in comparison with farms from cluster A. On-farm crop production weighted for the 50% on eutrophication, because of the use of fertiliser which could determine nitrate leaching in the water and ammonia emission in the air. Considering the subgroups, farms from cluster 4 had better economic performances than cluster 3; they were characterised by low stocking density, high feed self-sufficiency and balanced partition of farm land among different crop production (lucerne: 15.0% ; grass: 14.9%; maize for silage 21.2% of farm land). Farm included in cluster 4 had lower nitrogen and phosphorus balances than cluster 3; they probably paid more attention in using fertilisers (134 *vs* 178 kg of N input from artificial fertilisers) and sold feed (446 *vs* 0 kg of N output from sold feed). The energy use of cluster 4 was lower than cluster 3 (P=0.10) as a consequence of reduced use of off-farm products, especially feeds. In the context of intensive dairy farming of Northern Italy cluster 4 identifies a type of farming system that can produce good economic performances without increasing environmental impact.

References

Capper, J. L., Cady, R. A., Bauman, D. E., 2009. The environmental impact of dairy production: 1944 compared with 2007. J. Anim. Sci. 87, 2160-2167.

Pré Consultants, 2011. SimaPro 7.3.2 PhD, LCA software. Printerweg, Amersfoort, Netherlands.

SAS II, 2000. The SAS System for Windows. In Book The SAS System for Windows, Vol. Relase 8.01.

Table 1. Characteristics of clusters (least square means)

	Cluster A		**Cluster B**						
	Cluster 1	**Cluster 2**	**Cluster 3**	**Cluster 4**					
n farms	**6**	**11**	**8**	**16**	**SE**	**P**	**A vs B**	**1 vs 2**	**3 vs 4**
Farm land, ha	22.8	38.1	57.5	52.9	11.1	0.07	0.01	0.28	0.70
Livestock Unit	175	224	326	189	50.2	0.07	0.17	0.43	0.01
Stocking density, LU/ha	9.05	5.84	5.53	3.79	0.87	<0.001	<0.001	0.01	0.07
Feed self-sufficiency,%	32.0	49.8	61.8	73.1	2.42	<0.001	<0.001	<0.001	<0.001
Milk production, kg FPCM/cow/day	26.8	29.4	29.0	27.6	1.35	0.30	0.86	0.12	0.31
N balance, kg/ha	853	587	614	350	78.8	<0.001	<0.001	0.01	<0.01
P balance, kg/ha	125	70.4	60.6	31.2	11.8	<0.001	<0.001	<0.001	0.02
Gross margin, euro/t FPCM	128.5	206.3	143.2	218.4	20.3	<0.001	0.42	<0.001	<0.001

Table 2. The effect of cluster on total climate change, acidification, eutrophication, energy use and land use per kg fat-and-protein-corrected milk (least square means)

	Cluster A		**Cluster B**						
	Cluster 1	**Cluster 2**	**Cluster 3**	**Cluster 4**					
n farms	**6**	**11**	**8**	**16**	**SE**	**P**	**A *vs* B**	**1 *vs* 2**	**3 *vs* 4**
Total climate change, kg CO_2-eq.	1.43	1.24	1.28	1.31	0.08	0.29	0.55	0.06	0.72
On farm	0.92	0.82	0.90	0.98	0.06	0.07	0.15	0.19	0.23
Off farm	0.49	0.40	0.37	0.33	0.03	<0.001	<0.001	0.03	0.24
Acidification, g SO_2-eq.	20.0	18.2	18.7	21.1	1.44	0.17	0.49	0.33	0.12
On farm	14.7	14.1	16.2	19.1	1.19	<0.001	<0.001	0.68	0.03
Off farm	5.15	4.03	2.50	2.01	0.61	<0.001	<0.001	0.15	0.45
Eutrophication, g PO_4^{3-} eq.	7.9	8.2	8.7	10.1	0.64	<0.001	0.02	0.76	0.04
On farm	4.83	5.78	7.07	8.76	0.56	<0.001	<0.001	0.18	0.01
Off farm	3.07	2.38	1.59	1.35	0.31	<0.001	<0.001	0.09	0.48
Energy use, MJ	7.13	6.07	5.62	5.65	0.51	0.10	0.03	0.11	0.96
On farm	2.09	1.83	1.85	2.55	0.28	0.04	0.31	0.48	0.03
Off farm	4.20	3.59	3.50	3.03	0.29	0.01	0.01	0.10	0.13
Land use, m^2	1.50	1.46	1.46	1.57	0.10	0.62	0.65	0.76	0.31
On farm	0.37	0.57	0.69	0.93	0.07	<0.001	<0.001	0.03	<0.01
Off farm	1.12	0.88	0.77	0.64	0.08	<0.001	<0.001	0.02	0.12

11. Milk and meat biophysical allocation in dairy farms

Jean Baptiste Dollé[1,*], Armelle Gac[2]

[1] Institut de l'Elevage, Building and Environment Department, F-62051 St Laurent Blangy, France, [2] Institut de l'Elevage, Building and Environment Department, F-35652 Le Rheu, France, [*] Corresponding author. E-mail: *jean-baptiste.dolle@idele.fr*

For the dairy farming systems, where the main focus is to produce milk, the meat generated from surplus calves and culled dairy cows is an important co-product. In Life Cycle Assessments (LCA), the environmental burdens (GHG, etc.) must be distributed between these outputs. It is therefore necessary to determine total GHG emissions of the production system, which include dairy cows and heifers, and to allocate them between milk and meat. This issue has already been addressed in several studies (Cederberg & Stadig, 2003, Flysjö et al., 2011). However, the ISO 14044 suggests that the allocation should be avoided as soon as the system allows it, by subdivision of the multifunction process by sub-processes. This implies each sub-process has to be precisely defined as dedicated to the production of one of the co-products and the input and output fluxes of the whole system have to be attributed to each sub-process by a separated data collection or by the use of a technical distribution rule. Our investigation on French dairy systems is based on the causal relationship between the energy needed by animals on dairy farms and the milk and meat production. Then, the biophysical allocation rule proposed is based on the technical functioning of the production system and consists to separate energy needed for dairy cows and heifers. It is considered that the total energy of the feed intake by cows is needed to produce milk (except pregnancy energy affected to the calf) and the total energy needed by heifers for their growth is to produce meat (final live weight before calving) in relation with meat avoided from suckler beef systems. In accordance to the IPCC guidelines 2006 to determine methane emissions, energy demand for each category of animals (dairy cows and heifers) is evaluated by distinguishing energy for maintenance, activity, growth, pregnancy and milk production.

This biophysical allocation has been tested on French dairy systems. The assessments highlight that energy affected to milk (maintenance, activity, growth and milk production) represent 73% of the total energy needed by the dairy herd (dairy cow + heifers) and the energy affected to meat (calving+ heifers) correspond to 27%. This ratio, applied to allocate GHG from dairy system to milk and meat, has been compared to milk/meat ratios obtained with other allocation approaches: protein allocation used by FAO, IDF allocation (IDF - 2010), system expansion and economic allocation (Table 1). The distribution of environmental burdens to milk and meat varies in a range of 72-88%. The ratio obtained with biophysical allocation is close to the one with system expansion (meat from beef production system) but far from values observed with protein and IDF allocations. Applied in French dairy systems, these different allocation rules have an important incidence on carbon footprint. For milk and meat, carbon footprints at farm gate range respectively from 0.79 to 0.97 kg CO_2eq/kg of milk and from 4.4 and 9.5 kg CO_2eq/kg of live weight (Fig. 2).

The allocation choice for handling by-product is crucial for the outcome of the final carbon footprint of both milk and meat. This choice is often taken from a dairy production point of view but it should also consider that culled dairy cows represent a significant share of the total cattle meat production (40-50% in France). Consistency concerning allocation in LCA studies on dairy and beef system, but also on all animal production (pigs, poultry) should then be found. This is allowed by biophysical allocation, which also has the advantage of being related to breeding practices (feed intake, forage, etc.) and showing the environmental gain allowed by mitigating techniques.

References

Cederberg C., Stadig M., 2003. System expansion and allocation in life cycle assessment of milk and beef production, Int J LCA, 350-356.

Flysjö A., Cederberg C., Henriksson M, Ledgard S., 2011. How does co-product handling affect the carbon footprint of milk? Case study of milk production in New Zealand and Sweden, J Life cycle assessments, 16, 420-430.

IDF, 2010. Bulletin of the IDF n°445/2010.

IPCC 2006, 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National

Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan.

ISO 14044: 2006. Environmental Management - Life Cycle Assessment - Requirements and guidelines

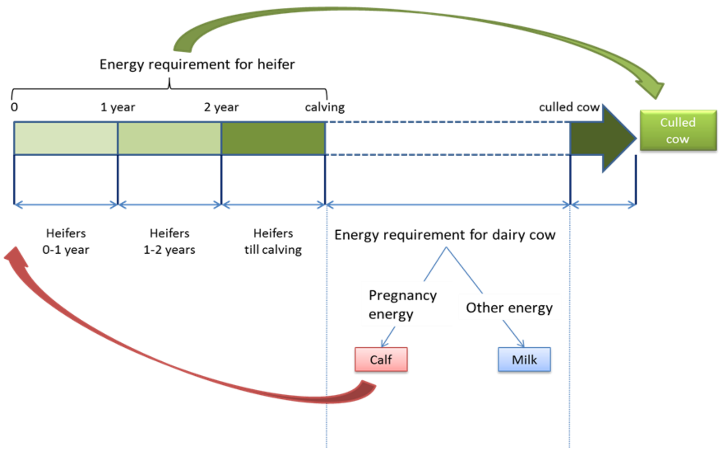


Figure 1. Energy calculation in dairy system

Table 1. Milk and meat ratios according to different allocation rules

Allocation rules	Milk	Meat
Protein	88%	12%
IDF	82%	18%
Economic	79%	21%
Biophysical	**73%**	**27%**
System expansion	72%	28%

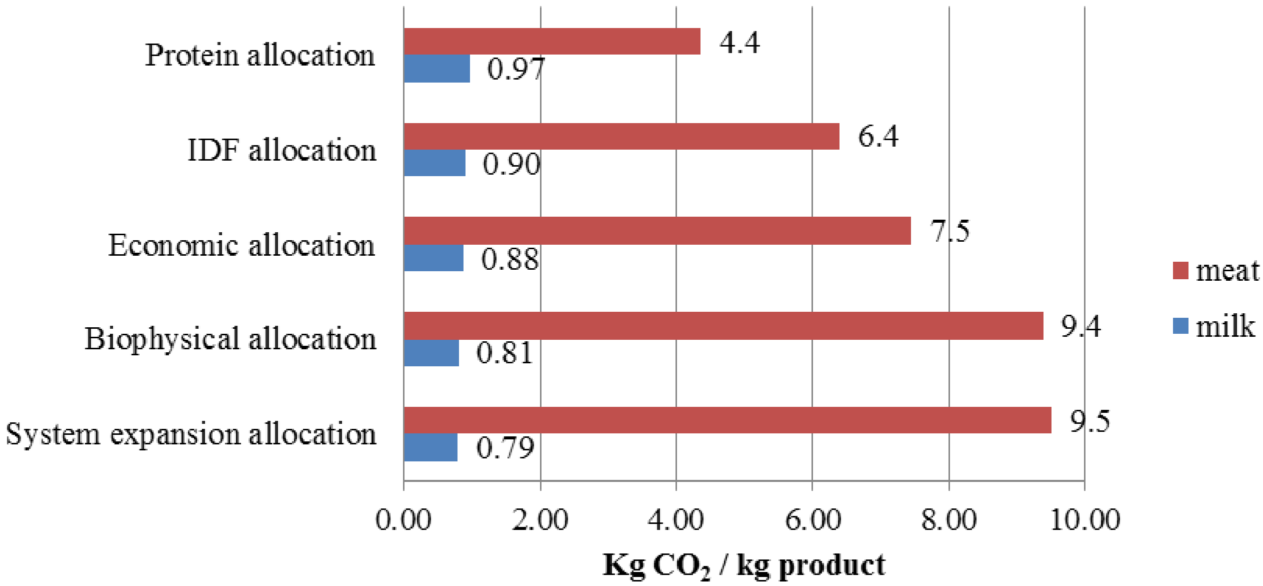


Figure 2. Milk and meat carbon footprint at dairy farm gate according to different allocation rules

12. Functional unit and reference flow in the dairy cow LCA

Chun-Youl Baek[1], Kun-Mo Lee[1,*], Kyu-Hyun Park[2]

[1] Department of Environmental Engineering, Eco-product Research Institute, Ajou University, Korea, [2] National Institute of Animal Science, Suwon, Korea, [*] Corresponding author. E-mail: *kunlee@ajou.ac.kr*

Functional unit in LCA defines the functional performance characteristics delivered by the product system, here livestock. It serves as denominator in normalising the input and output data (environmental load: EL) of a product system. Normalised EL is intended implicitly and explicitly for comparison between processes and activities in a given product system, or between different product systems. Defining functional unit of an industrial product is less complicated compared with that of a livestock. This is mainly because of the difference in functional performance characteristics of the two, where the former possesses fixed attribute while the latter variable attributes. Reference flow is an amount of product needed to fulfil the function of a product system.
The objective of this paper is to investigate the effect of product attributes on the functional performance characteristics in defining functional unit and reference flow. A dairy cow was chosen as the target product for the livestock LCA and a typical size dairy farm of 65 heads of cows with different stages in growth in Korea was chosen to gather input and output data. A dairy cow undergoes different stages in its life cycle including calf, heifer, lactating cow and dry cow with a life span of average five (5) years. Different life cycle stages of a cow means composition of a cow in a farm during the life span of a cow of five years may influence the variability of the EL data from a cow. Thus, composition of a cow in a farm should be clearly defined in the reference flow.
Key findings of this paper include: functional unit should be defined based on the functional performance characteristics delivered by the product, reference flow should be one cow with known composition and life span, and composition of cows in a farm may vary depending on the number of heads the farm houses which will affect LCA results. A stable composition should be used for defining reference flow in the dairy cow LCA.

References

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., De Haan, C., 2006. Livestock's long shadow: environmental issues and options, FAO, Rome, Italy.

De Vries, M., De Boer, I.J.M., 2010. Comparing environmental impacts for livestock products - A review of life cycle assessments, Livestock Science, 128, 1-11.

13. Linking environmental impact of milk production to the territory: the Qualaiter project

Michaël Mathot[1,*], Michael S. Corson[2,3], Didier Stilmant[1]

[1] Farming Systems, Territory and Information Technologies Unit, Walloon Agricultural Research Center, rue du Serpont 100, B-6800 Libramont, Belgium, [2] INRA, UMR 1069, Sol, Agro- et hydrosystème Spatialisation, F-35000, Rennes, France, [3] Agrocampus Ouest, F-35000, Rennes, France, [*] Corresponding author. E-mail: *m.mathot@cra.wallonie.be*

For the last decade, animal productions have been strongly criticised for their environmental impact. However, within a given food chain large impact variations between different systems of production are highlighted at world (FAO, 2010) or at regional scale (Casey et al.,2006). Variations that are worthwhile to explore. Therefore the Qualaiter project aims to quantify environmental impact of milk production using LCA approach, through the adaptation of EDEN methodology (Van der Werf et al., 2009) to the Belgian (Walloon) context, in link to territorial diversity recorded within Walloon area. This diversity, related to climate, geological and historical conditions, leads to associated agricultural practices diversity that potentially influence the environmental impact of the production systems. In order to discriminate the territorial from the management role on the environmental impact of dairy farms we identify three farms types that could be found in three contrasted area to compare their environmental performances.

References

Casey, J.W., Holden, M.N., 2006. Quantification of GHG emissions from sucker-beef production in Ireland. Agricultural Systems 90, 79-98.

FAO, 2010. Greenhouse Gas Emissions from the dairy Sector. A Life Cycle Assessment. Food and Agricultural Organization of the United Nations. Animal Production and Health Division. 94 p.

Van der Werf, H. M. G., Kanyarushoki C., Corson M. S., 2009. An operational method for the evaluation of resource use and environmental impacts of dairy farms by life cycle assessment. Journal of Environmental Management 90, 3643-3652.

14. A matrix approach to spatialise impacts of U.S. milk production

Anne Asselin-Balençon*, Andrew Henderson, Olivier Jolliet

School of Public Health (SPH), University of Michigan, 6622 SPH Tower, 1415 Washington Heights, Ann Arbor, Michigan 48109-2029, USA. * Corresponding author. E-mail: *aasselin@umich.edu*

In many cases in production, goods are not consumed in the same place they are produced. This is particularly the case in agriculture. Crops are produced for many purposes, such as milk or meat production or processing for human consumption. Downstream production (e.g., milk or meat) does not necessarily take place in the same location where the crops were grown; some of these crops can be transported over long distances before being consumed or processed. This is especially the case in the case of United States milk production, where corn grain, for example, is mainly produced in the Corn Belt but consumed all over the nation.

On the other end, and for some impact categories, the impacts of crop production are intensely local. Such is the case of water stress, for example. Crop production is a water intensive activity, and water use and impacts vary greatly depending on region, crop irrigation and type of crop. In some regions, irrigation accounts for up to 90% of water withdrawn from available sources, while in others with plentiful rainwater, irrigation is barely necessary.

This presentation introduces a generic model to properly account for the attribution of a flow or impact to milk-producing locations and grain-producing locations, through a matrix approach at the state level. This approach allows an analysis in which an inventory flow in milk-producing state j can be decomposed into inventory flows in grain-producing states i. It also structures the data inputs from feed and dairy farms in a consistent way and enables adaptation and refinement to new data sets or to different resolutions. The original matrix approach is key to correctly assess local impacts of a given industry by accounting for the origin of its supplies and can be used for different spatial resolutions and industries.

15. Environmental and socioeconomic references of French conventional pig systems

Sandrine Espagnol[1,*], Alexandre Rugani[1], Christine Baratte[2], Christine Roguet[1], Michel Marcon[1], Aurélie Tailleur[5], Cyrille Rigolot[2], Jean-Yves Dourmad[3,4]

[1]IFIP Institut du porc, La Motte au vicomte BP35104, 35651 Le Rheu, France, [2]INRA UMR1080 Production du Lait, 35590 Saint Gilles, France, [3]INRA UMR1079 Système d'Elevage Nutrition Animale et Humaine, 35590 Saint Gilles, France, [4] Agrocampus Ouest, UMR 1079 Systèmes d'Élevage Nutrition Animale et Humaine, 35000 Rennes, France, [5]Arvalis Institut du Végétal, 91720 Boigneville, France, [*] Corresponding author. E-mail: *sandrine.espagnol@ifip.asso.fr*

This study aims to produce multicriteria environmental references (using Life Cycle Assessment, LCA) associated with socioeconomic indicators for different types of pig units representative of the dominant French standard production systems. Eight systems are assessed, discriminated according to the size, the degree of specialisation and the location of the pig unit, the slurry management and the pig feeding strategy. The results are expressed per kilogram live pig produced and the LCA boundaries include the production and the supply of inputs, the production of buildings, the pig breeding and the management of slurry. The references bring a socioeconomic and environmental photography of the performance of the existing pig production systems and their variability between and within systems. The environmental results allow identification of the most strategic and easily attainable options of improvement. The effect of different improvement strategies are indentified in connection with feed formulation, improvement of animal performance, and the implementation of the recommended good environmental practices. The socioeconomic indicators of the systems show the various levels of access to the control levers of action.

References

Basset-Mens, C., Van Der Werf, H., 2004. Evaluation environnementale de systèmes de production de porcs contrastés. Journées Rech. Porcine, 36, 47-52.

Chardon, X., Rigolot, C., Baratte, C., Le Gall, A., Espagnol, S., Martin-Clouaire, R., Rellier, J.P., Raison, C., Poupa, J.C., Faverdin, P., 2007. MELODIE: A whole-farm model to study the dynamics of nutrients in integrated dairy and pig farms. In Oxley, L. and Kulasari, D. (eds) MODSIM 2007 International Congress on Modelling and Simulation. December 2007, 1638-1645.

CORPEN, 2003. Estimation des rejets d'azote-phosphore-potassium-cuivre et zinc des porcs. Influence de la conduite alimentaire et du mode de logement des animaux sur la nature et la gestion des déjections produites. Ministère de l'écologie, 41p.

Gac, A., Deltour, L., Cariolle, M., Dollé, J-B., Espagnol S., Flénet, F., Guingand, N., Lagadec, S., Le Gall, A., Lellahi, A. Malaval, C. Ponchant, P., Tailleur, A., 2010. GES'TIM: Guide méthodologique pour l'estimation des impacts des activités agricoles sur l'effet de serre. Version 1.2, 155p.

GIEC, 2006. Lignes directrices 2006 du GIEC pour les inventaires nationaux de gaz à effet de serre, Volume 4: Agriculture, foresterie et autres affectations des terres. www.ipcc.ch

Guingand, N., Aubert, C., Dollé, J.-B., 2010.Guide des bonnes pratiques environnementales d'élevage. Institut de l'élevage, IFIP, ITAVI, 303p.

Ilari, E., 2007. De 2000 à 2005, comment ont évolué les exploitations porcines françaises? Techniporc, 30 (6), 19-25.

Loyon, L., Guiziou, F., Beline, F., Peu, P., 2007. Gaseous emissions (NH_3, N_2O, CH_4 and CO_2) from the aerobic treatment of piggery slurry – Comparison with a conventional storage system. Biosystems engineering, 97, 472-480.

Marcon, M., 2008. Consommations d'énergie des bâtiments porcins: comment les réduire?. IFIP, France.

Massabie, P., 2001. L'abreuvement des porcs. Techniporc, 24 (6), 9-14

Rigolot, C., Meda, B., Espagnol, S., Trochet, T., Dourmad, J-Y. 2009. ACV de 5 systèmes porcins avec différentes hypothèses de comptabilisation des impacts. Journées Rech. Porcine, 41, 281-282.

Nemecek, T., Kägi, T., 2007. Life cycle inventories of agricultural production systems. EcoInvent report No.15. Zürich, CH. www.ecoinvent.ch

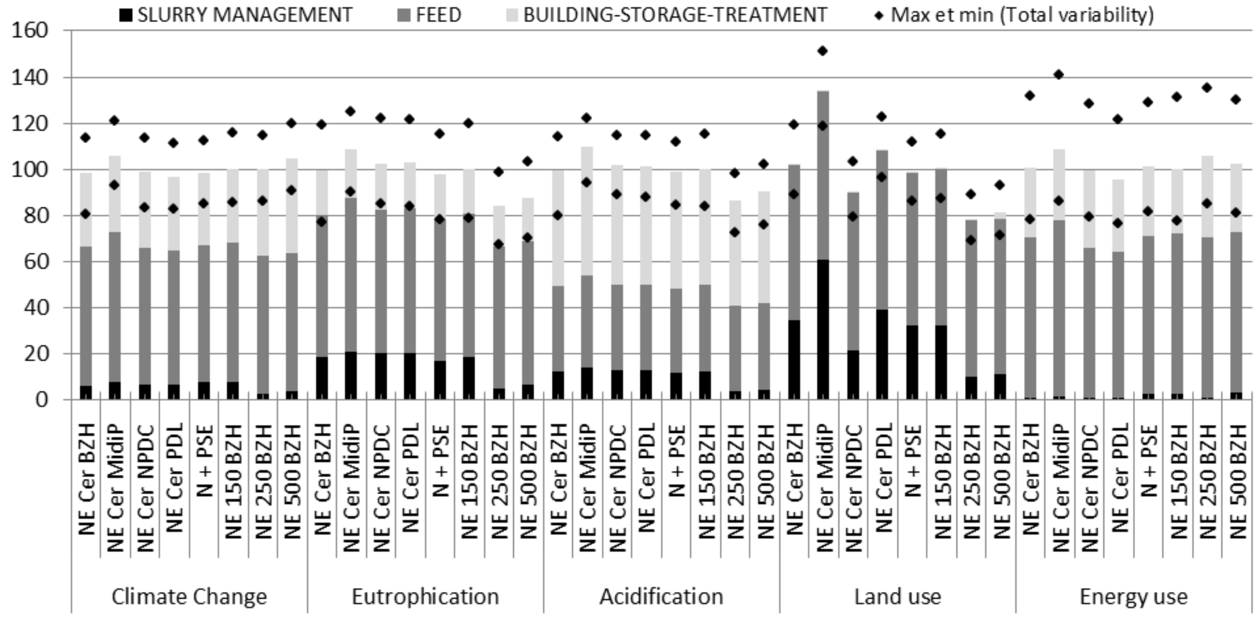


Figure 1. LCA Impacts of different types of pig units and variability (results in% of the pig unit NE 150 BZH).

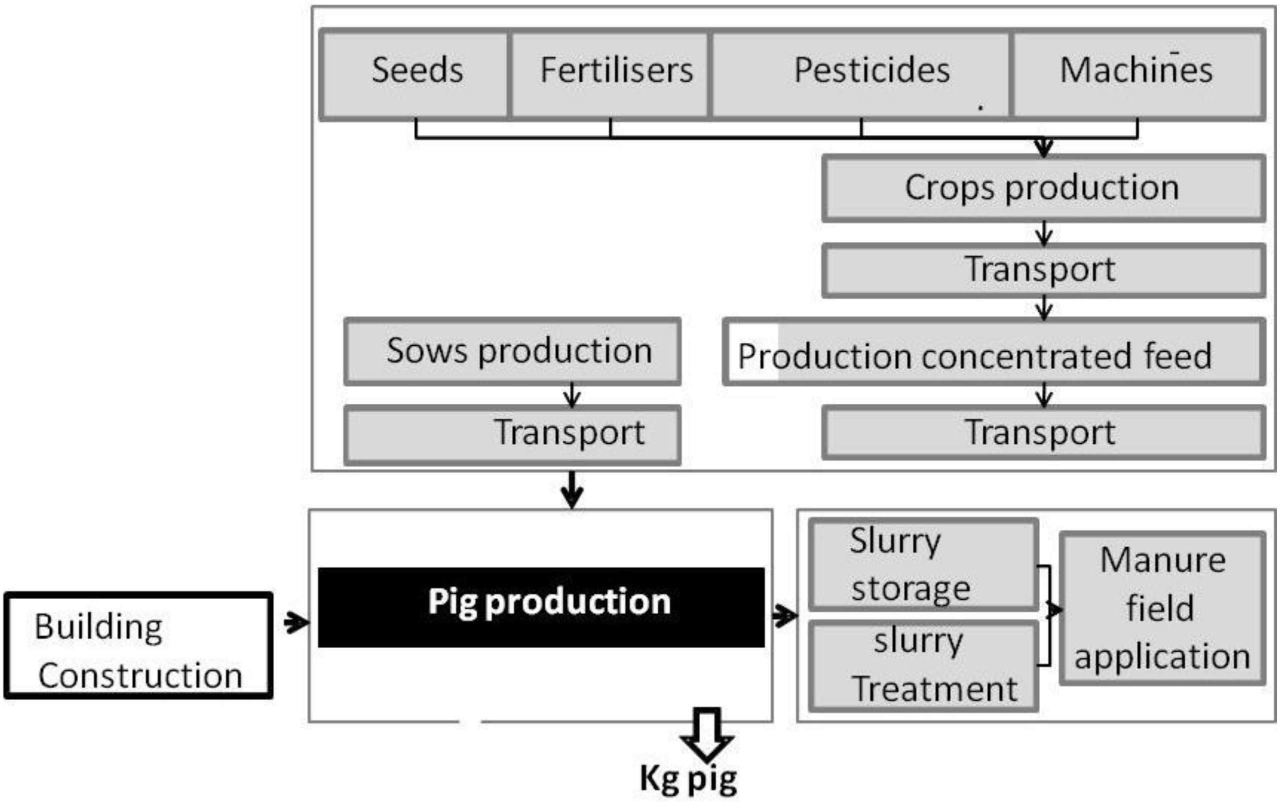


Figure 2. Flow diagram for LCA pig production.

16. Environmental impact of the pork supply chain depending on farm performances

Karoline Reckmann*, Joachim Krieter

Institute of Animal Breeding and Husbandry, Christian-Albrechts-University Kiel, Germany, * Corresponding author. E-mail: *kreckmann@tierzucht.uni-kiel.de*

The aim of the study is to figure the effect of changes in farm performances on the environmental impact of pork production. The environmental assessment was performed using data representing the typical Northern German pork production in 2010 on the one hand. On the other hand, data of German pig farms with 25% highest and lowest efficiency in terms of net profit (economic success) was used. The data for the farms was gathered from an extension service for pig farms in Northern Germany (SSB, 2010). The database of the feed and slaughtering stage was composed of own data collection from specific companies and of literature. The system boundaries of the Life Cycle Assessment cover the feed production, pig housing as well as the slaughtering stage. Infrastructure, packaging, retail and consumption were excluded. The manure produced at the farm had also a value as a fertiliser, thus substituting synthetic fertiliser. The environmental impact of the whole production chain was expressed per '1 kg pork produced'. Three impact categories were considered: Global warming potential (GWP), Eutrophication potential (EP) and Acidification potential (AP), expressed in equivalents (eq).

Table 1 summarises the environmental performance with respect to the three impact categories. The average pork production results in a GWP of 3.62 kg CO_2-eq, an EP of 42 g PO_4-eq and an AP of 89 g SO_2-eq per kg pork. A higher efficiency on farm level reduces the estimated environmental impacts of GWP as well as EP. The GWP is improved by 358 g CO_2-eq per kg pork, whereas the reduction of EP reaches 0.37 g PO_4-eq per kg pork. In contrast, the average production results in a 1% increased AP compared to the production with a lower efficiency. However, the lowest AP arises out of the pork production with an enhanced performance on farm level. Nguyen et al. (2011) estimated a GWP of 3.1 kg CO_2-eq per kg carcass weight produced in Denmark. These results are in line with ours. In case of higher efficiency, the potential of reducing the GWP (2.8 kg CO_2-eq per kg carcass weight) was shown more clearly than in the present study. This could arise from the different method used or the fact that the higher efficiency was based on biological parameters.

For estimation of the contribution of these stages to the overall impacts, the production chain was divided into the stages of feed production, pig housing and slaughtering. Feed production is the main contributor in the case of GWP with a share of 80% (Fig. 1), followed by pig housing (15%) and slaughtering. The largest part of Eutrophication is caused by pig housing (51%) and feed production (48%). In the case of AP, pig housing plays a key role with an amount of 72%. Feed production is responsible for 27% of the AP, whereas only 1% of the AP originates from the slaughtering process. Over all the shown impact categories, slaughtering has only a marginal share (1-5%) to the environmental impacts of the pork supply chain.

Further calculations will include different scenarios by varying single parameters on farm level, as e.g. number of litters per sow and year, feed intake, daily weight gain and feed conversion rate. Results from the different LCAs will be compared to identify performance parameters with a high effect on the overall impacts. In order to illustrate the variation of the impacts, Monte Carlo methods will be used for further calculations.

References

Nguyen TLT, Hermansen JE, Mogensen L., 2011. Environmental assessment of Danish pork. Aarhus, Denmark: Aarhus University.

SSB, 2010. Schweinereport 2010. Schweinespezialberatung Schleswig-Holstein e.V.

Table 1. Results for the impact categories Global Warming Potential (GWP), Eutrophication (EP) and Acidification Potential (AP) for the three scenarios of pork production, related to 1 kg pork produced.

Impact category	+ 25% farm efficiency	Average production 2010	- 25% farm efficiency
GWP (kg CO_2-eq)	3.46	3.62	3.67
EP (g PO_4-eq)	42.48	42.84	44.88
AP (g SO_2-eq)	85.41	89.56	89.10

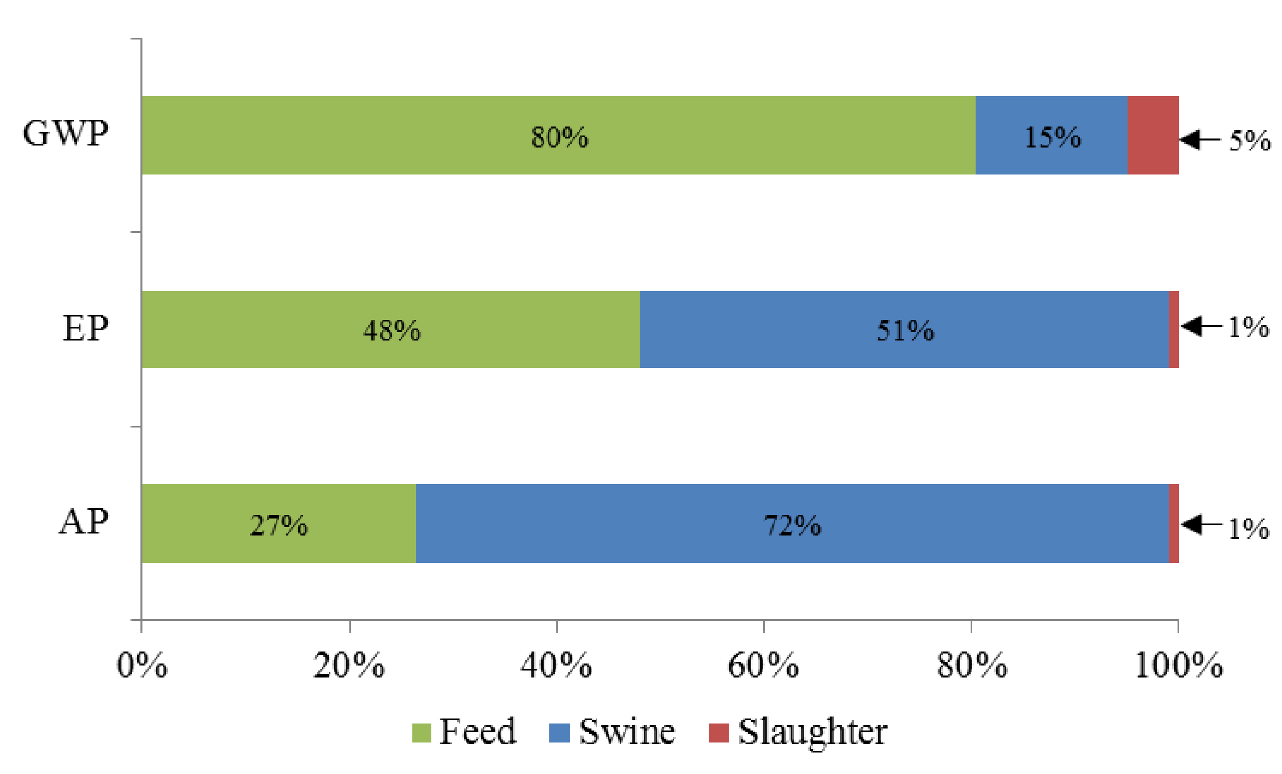


Figure 1. Percentages of the different stages of an average pork production for the impact categories Global Warming Potential (GWP), Eutrophication (EP) and Acidification Potential (AP) related to 1 kg pork produced.

17. Characterisation of the pig systems panel for the production of environmental data in the program Agri-BALYSE

Sandrine Espagnol[1,*], Thibault Salou[2,3], Hayo van der Werf[2,3]

[1]IFIP Institut du porc, La Motte au vicomte BP35104, 35651 Le Rheu, France, [2]INRA, UMR1069 Sol, Agro and hydroSystem, 35000 Rennes, France
[3]Agrocampus Ouest, 35000 Rennes, France, * Corresponding author. E-mail: *sandrine.espagnol@ifip.asso.fr*

The program Agri-BALYSE is a French initiative which aims to develop a public LCI-database of French agricultural products by the end of 2012.This database has two goals: (1) it will provide data to set up environmental labeling for consumers; (2) it will also include environmental assessments of different contrasting production systems which will be more useful for the R&D and the identification of action levers.
Pig production/breeding is one of the agricultural products which is covered by the Agri-BALYSE-database. Given the fact that there exists wide variety of pig production systems in France, it has been necessary to choose the best panel of pig production/breeding systems in order to answer to the different objectives of the project. Actual representativeness of production systems was an important selection criteria but not the only one. Emphasis was given also to future representativeness as well as social desirability and agricultural practices. Three systems were defined to provide data for the environmental labeling: a national average standard production system, an organic production system as well as a pig system "fermier label rouge". Four other systems have been selected to analyse the incidence of specialisation levels and feed strategies which are considered to be the most important action drivers.
The chosen systems are qualified concerning their adaptability to the different uses by giving their actual and future representativeness, analyzing if the main levers of action are considered, considering system which are wanted by society. The final definition of the production system has also to consider if date and information are available and to see if it is (or will be) possible to trace system characteristics to the kilogram of pig.

References

IFIP, 2005-2009. Porc performances.
IFIP, 2009. Bâtiments d'élevage porcin et environnement: Analyse de l'enquête de novembre 2008 réalisée par le SCEES. France AgriMer, 272 p.
ITP, 2005. Compositions des effluents porcins et de leurs co-produits de traitement - Quantités produites, 68 p.

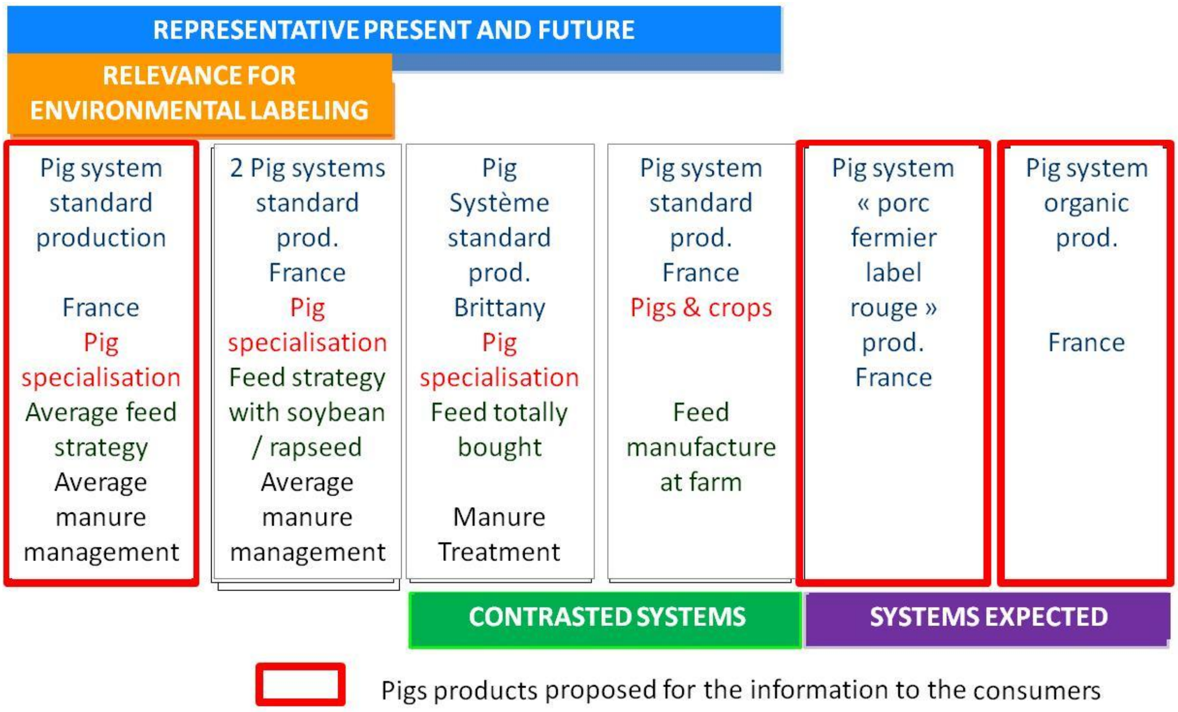


Figure 1. Pig systems for Agri-BALYSE

18. Life cycle thinking applied to an immunological product (vaccine) used for boar taint control in male pigs

Paulo Moraes[1], Jim Allison[1], Joseph A. Robinson[1], Stéphan Martin[1], Gian Luca Baldo[2], Paola Borla[2]

[1] Pfizer Animal Health, USA and Europe, [2] Life Cycle Engineering (LCE), Italy, * Corresponding author. E-mail: *baldo@studiolce.it*

In 2009, Pfizer Animal Health decided to apply the LCA methodology to some innovative products, with a first case-study on Improvac®, an immunological product (vaccine) for male pigs that provides farmers with an alternative way to avoid the problem of boar taint: its use increases the efficiency of male pig production and could consequently provide life-cycle environmental benefits.
A meaningful LCA study of the vaccine required the collection, with a global perspective, of reliable data about the life-cycle environmental burden of an average farm that uses or not the product. Information from within-farm comparative studies was also integrated into the analysis. This led to an understanding of any possible environmental benefits of the vaccine's adoption by benchmarking against existing, traditional practices (castration).
The two most relevant phases contributing to the life-cycle environmental burden of the examined system are the production of feed given to pigs and pig slurry management (Fig. 1). Starting from the feed recipe, an interesting close examination was conducted on agricultural practices by country, providing a valuable description of how feed production burden changes according to local conditions (yields, fertiliser use, etc.). The same conclusion applies to different slurry management procedures and technologies. Overall, the LCA provided meaningful information for use by farmers who are interested in reducing their carbon footprint when rearing swine for pork meat.
The study shows a reduction in the environmental impacts considered for the vaccinated pig life cycle compared to the castrated one. In particular, the calculated carbon footprint for the Improvac pig system demonstrates a reduction vs. the physically castrated pig system of 3.7% in terms of kg live-weight; given the annual production of pigs reared globally for protein consumption (about 500M males), this carbon footprint reduction is incrementally significant and supports the adoption of Improvac over the traditional approach of castrating boars.
The product is approved for use and distributed in nearly all pig producing countries worldwide: from South America to the US and from Europe to Australia, including the world leader in swine production, China.
The Improvac Environmental Product Declaration was first published in January 2011 on the International EPD register and renewed in early 2012 after the required external review by a third party (www.environdec.com).

References

Allison J., McKeith F., C Souza, Boler D., Killefer J. Hennessy D., 2009. Impact of using vaccination with IMPROVAC® rather than physical castration on the carcass characteristics of finishing male pigs in ICOMST, 55th International Congress of Meat Science and Technology, Copenhagen 16-21 August 2009, Denmark, paper n° 7.38.

Basset-Mens C., Van der Werf H.M.G., Durand P., Leterme P., 2006. Implications of uncertainty and variability in the life cycle assessment of pig production systems. Int J LCA 11, 298-304.

Dunshea F., Colantoni C., Howard K., McCauley I., Jackson P., Long K.A., Lopaticki S., Nugent E.A., Simons J.A., Walker J., Hennessy D.P., 2001. Vaccination of boars with a GnRH vaccine (Improvac®) eliminates boar taint and increases growth performance. J. Anim. Sci. 79, 2524-2535.

European Commission, 2003. Integrated Pollution Prevention and Control (IPPC) Reference Document on Best Available Techniques for Intensive Rearing of Poultry and Pigs. JRC.

Fàbrega E., Velarde A., Cros J., Gispert M., Suárez P., Tibau J., Soler J., 2010. Effect of vaccination against gonadotrophin-releasing hormone, using Improvac®, on growth performance, body composition, behaviour and acute phase proteins. Livest Prod Sci 132, 53-59.

Pfizer Animal Health, 2011. Environmental Product Declaration of Improvac (N° S-P-00261). The EPD International System.

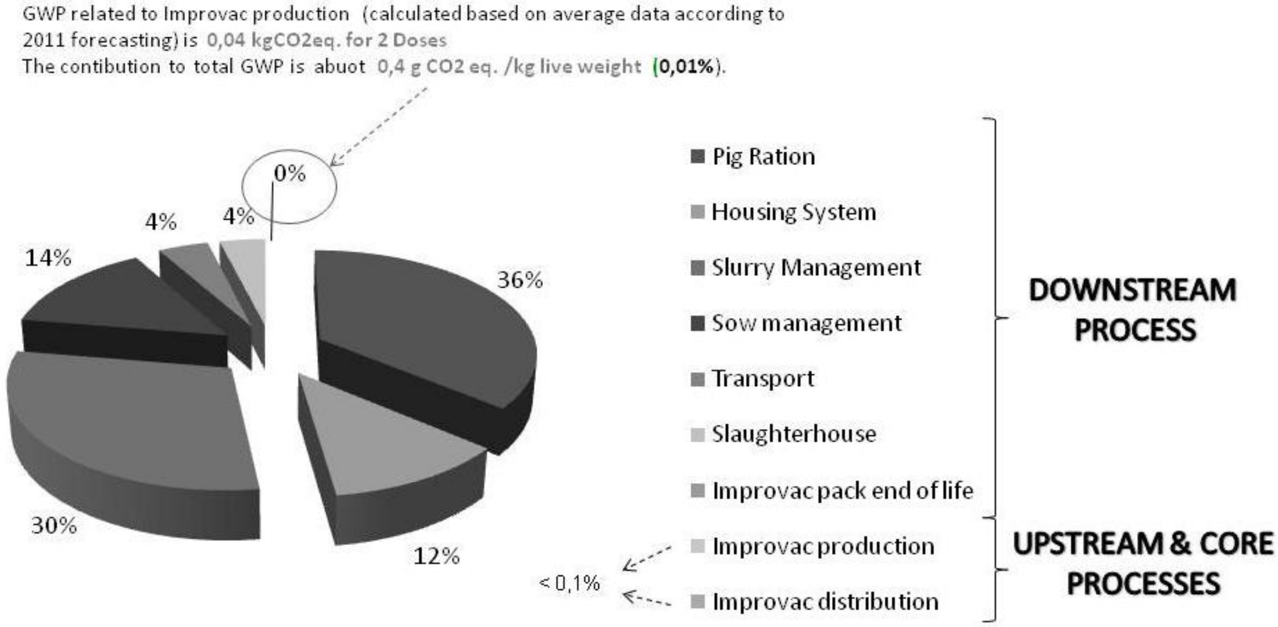


Figure 1. The most relevant contributors to the Carbon Footprint Indicator of the Improvac® system.

19. Emergy and life-cycle sustainability of pig meat products

Benedetto Rugani[1,*], Elena Neri[2], Valentina Niccolucci[3], Federico M. Pulselli[3], Simone Bastianoni[2]

[1]Environmental evaluation & management, Public Research Centre Henri Tudor (CRPHT) - Resource Centre for Environmental, Technologies (CRTE) 66, rue de Luxembourg - P.B. 144, L-4002, Esch-sur-Alzette, Luxembourg, [2]Ecodynamics Group, Dept. of Chemistry, University of Siena. via A. Moro, 2 – 53100 Siena – Italy, [3]Ecodynamics Group, Dept. of Chemistry, University of Siena, via della Diana, 2A – 53100 Siena, Italy, [*] Corresponding author. E-mail: *benedetto.rugani@tudor.lu*

The effects of globalisation have inevitably an impact on food choices in relation to mass produced at low cost and greater usability at the expense of local products, which are often more expensive but with higher quality. One of these examples is represented by the meat of Cinta Senese, which is a typical pig race of the rural area of Siena, Italy. The breeding and production of this race is very different from that of the intensive white race (Large White). Indeed, Cinta Senese is reared in an almost completely natural way, within forests and usually without using industrial fodder (Basset-Mens et al., 2006). The aim of this work is to assess the sustainability of pig meat products by explicitly focusing on the breeding phase. We have compared the two races through the application of Life Cycle Assessment (LCA) and eMergy analyses. In this connection, eMergy (Odum, 1996) is considered to be complementary to LCA allowing for a broad assessment of resource consumptions and also of social and economic issues (Rugani et al., 2011). As shown in Fig.1, the application of LCA highlights that the production of 1 kg of Cinta Senese meat has lower potential impacts than the production of Large White within a set of impact categories considered (i.e. climate change, acidification and eutrophication). Indeed the production of 1 kg of Cinta Senese pig has a potential climate change impact of 2.25 kg CO_2eq, while for Large White is 3.6 kg CO_2eq (Fig.1). A greater discrepancy is observed on the potential impact related to acidification (0.016 kg SO_2eq for Cinta Senese and 0.045 kg SO_2 eq for Large White), while similar scores are depicted for the potential impact on eutrophication (around 0.23 kg NO_3eq). Fertilisers, water and agricultural machinery operations, used for fodder production, are the main responsible of all environmental impacts in Cinta Senese rearing system. On the other hand, results from eMergy evaluation show that Cinta Senese is less efficient than the White race in terms of yield. In fact, the specific eMergy of Cinta Senese was about 3.5 times greater than that of Large White: 7.53E+09 seJ/g and 2.57E+09 seJ/g respectively, this is principally due to the rearing system. During one year of growth, Cinta Senese living pig weighs 110 kg while Large White 140 kg and the available space for each head is 12.00E+03 m^2/head vs 0.23E+03 m^2/head, respectively. Emergy evaluation highlights that the production system of Cinta Senese, due to the large use of renewable and local resources, generates less direct and indirect environmental impacts than the Large White breeding (the percent of renewability is 21.03 and 2.15 respectively). The "monetary" value of renewable (R) and non-renewable (N) emergy flows, created by giving a price to the local environmental eMergy, is 9.80E+03 seJ/€ for Cinta Senese and 8.78E+03 seJ/€ for Large White. Results highlight higher relative contributions of labour for the production of Cinta Senese, demonstrating the wider relevance of direct human resources for this extensive system. The present study points out that it is possible to discuss the three fundamental pillars at the base of the sustainability concept (environment, society and economy) by using eMergy combined with a life cycle inventory. Emergy evaluation emphasised the peculiarities of the Cinta Senese system, in comparison to a conventional pig breeding system of White race pigs, and the importance of the local ecosystem for the entire process dynamics.

References

Basset-Mens, C., van der Werf, H.M.G., Durand, P., Leterme Ph., 2006. Implications of Uncertainty and Variability in the Life Cycle Assessment of Pig Production Systems. Int. J. Life Cycle Assess. 11, 298-304.

Dalgaard, R., 2007. The Environmental Impact of Pork Production from a Life Cycle Perspective. Ph.D. thesis, Aalborg University, Denmark.

Guinèe, J. B., Gorreèe, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. A., de Bruijn, H., van Duin, R., Huijbregts, M. A., 2001. Handbook on Life Cycle Assessment. Operational Guide to the ISO standards, Kluwer Academic Publisher: Dordrecht, The Netherlands.

Odum, H.T., 1996. Environmental Accounting: Emergy and Environmental Decision Making. Wiley & Sons, New York, USA.

Rugani, B., Pulselli, F.M., Niccolucci, V., 2011. Emergy Assessment of Cinta Senese Breeding and Comparison with Conventional Pig Production, in: Emergy Synthesis 6. Theory and Applications of the Emergy Methodology. The Center for Environmental Policy, Gainesville, USA, pp. 347-358.

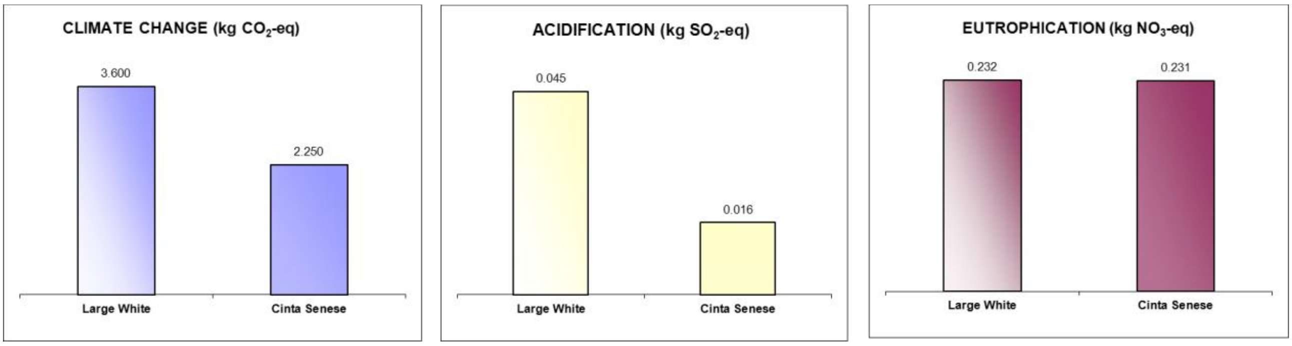


Figure 1. Results comparison of impact characterisation for the production of Cinta Senese and Large White (LW data source: Dalgaard et al. (2007); reference = 1 kg live pigs). Life Cycle Inventory (LCI) & Life Cycle Impact Assessment (LCIA) elaborated using SimaPro 7. Impact characterisation performed using CML2001 method, as proposed by Guinèe et al. (2001).

20. Life cycle assessment of an intensive Iberian pig farm

Joel Gonzàlez*, Aitana Santamaria, Emma Fàbrega, Joan Tibau, Assumpció Antón

IRTA, Research & Technology, Food & Agriculture, Spain, * Corresponding author. E-mail: *joel.gonzalez@irta.cat*

Iberian pig production has been a major livestock practice in Spain for a long time, offering meat products destined to a niche market which demands a high sensory quality. Nowadays the most extended system is the intensive production of the crossbred Iberian x Duroc, reared during at least 10 months of age and slaughtered at approximately 150 kg live weight.. The recent unfavourable economic situation and increase in feed price has resulted in the closing at some farms, pushing them to improve their efficiency using the resources. Within this benchmark, an evaluation of the efficiency of the system is needed to assess farming viability by means of standardised tools. One of the main pillars of this efficiency is environmental impact, which was estimated by a Life Cycle Assessment in the present study, to be considered in an integrated sustainability evaluation. The results should determine weaknesses and strengths of the system, as well as numerical scores, which could be used as consumer information.

A representative closed cycle farm of Iberian pig production (Iberian x Duroc) located in Catalonia (Alt Empordà) was selected for the study. It has capacity for 450 sows, 1,120 piglets and 3,000 fatteners. One 150 kg pig at the farm gate was chosen as a functional unit. The system boundary is defined up to the farm gate, considering waste disposal, but not considering post stages such as slaughtering or commercialisation. Impact categories selected were midpoint impact categories defined by the CML (Guinée, et al., 2002).

Primary data were obtained from the representative farm object of the study. ECOGAN software from the Agriculture Department (MARM, 2011) was used to calculate the NH_3, NO_2 and CH_4 emissions and the resources used within each production stage. Secondary data were obtained from Ecoinvent database. Origin of feed ingredients was based on data from cereal producers in Spain. It was assumed that soybean came from Brazil. The software used for the assessment was the SimaPro version 7.2 (PRé Consultants, 2010), performing the compulsory phases of classification and characterisation. For the whole closed cycle, data were collected from the different production stages including gestation, lactation, rearing and fattening, they were considered as a part of the foreground (Fig. 1).

Table 1 shows the absolute values for each environmental impact category for a 150 kg pig and related to 1 kg of meat. It is important to bear in mind that an Iberian pig is less efficient than conventional pigs with regard to its conversion rate, and this elevates the values of the impacts because they consume more feed. Therefore it would be interesting to relate the impacts not only to kg of meat, but also to other quality indexes such as the percentage of intramuscular fat.

Crop and feed production, including grain and soybean production needed for their manufacture, were shown to be the main environmental constrains for impact categories such as eutrophication, air acidification and climate change, while the use of energy within maternity contributed to a larger extent to impacts related to energy consumption, such as abiotic depletion.

This study also provides information for the main drawbacks regarding the application of the methodology and will therefore need further research, especially if it is to be used in environmental communication or labels. Main drawbacks can be summarised as: lack of local datasets to be used in the background system; agreement and homogenisation among models used in the emission factors estimation, as well as emissions related to land use for imported feeds; consideration of nutritional qualities as functional units instead of meat quantity.

References

Ecoinvent, 2010. ecoinvent Data v2.2. Swiss Centre for Life Cycle Inventories.

Guinée, J. B., Gorrée, M., Heijungs, R., Huppes, G., R., K., de Koning, A., Wegener Sleeswijk A., Suh, S., Udo de Haes, H., Bruijn, H., Duin, R. v. and Huijbregts, M. A. J., (2002). Handbook on life cycle assessment. Operational guide to the ISO standards. Dordrecht, The Netherlands. Kluwer.

MARM, 2011. ECOGAN 2009.0.0 Software. Spain.

PRé Consultants, 2010. SimaPro 7.2. Amersfoort. The Netherlands.

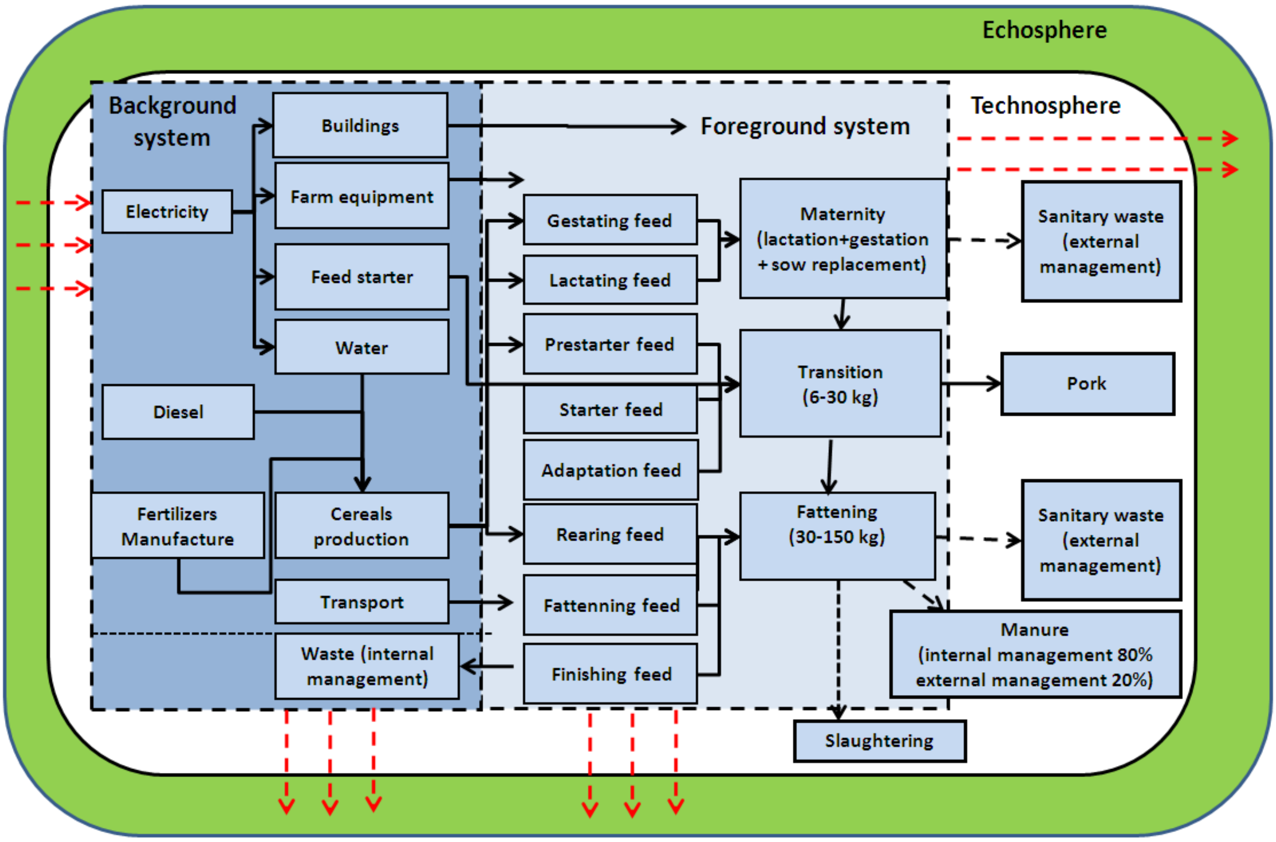


Figure 1. Flowchart of a representative closed cycle farm of intensive Iberian x Duroc pig production in Spain.

Table 1. The environmental impacts of pig production expressed per pig unit and kg pork.

Impact category	**Units**	**per pig unit (150 kg)**	**per 1 kg pork**
Air Acidification	kg SO_2 eq	5.05	0.03
Global warming	kg CO_2 eq	910.6	6.07
Abiotic depletion	kg Sb eq	2.56	0.02
Eutrophication	kg PO_4^{3-} eq	6.91	0.05
Photochemical oxidant formation	kg C_2H_4	0.79	0.005
Freshwater Toxicity	kg 1,4-DB eq	82.21	0.55
Human Toxicity	kg 1,4-DB eq	349.8	2.33
Terrestrial Toxicity	kg 1,4-DB eq	2.60	0.02

21. Comparison of environmental impacts of corn or sorghum as geese feed in *foie gras* production

Julien Arroyo[1,3], Laurence Fortun-Lamothe[1,*], Alain Auvergne[1], Jean-Pierre Dubois[3], Joël Aubin[2]

[1] INRA, INPT ENSAT, INPT ENVT, UMR 1289 Tissus Animaux, Nutrition, Digestion, Ecosystème et Métabolisme, F-31326 Castanet-Tolosan Cedex, France, [2] INRA, Agrocampus Rennes, UMR1069, Sol Agronomie Spatialisation, 65 rue de Saint Brieuc, CS 84215, F-35042 Rennes Cedex, France , [3] ASSEL-DOR, Station d'expérimentation appliquée et de démonstration sur l'oie, La Tour de Glane, 24420 Coulaures. , [*] Corresponding author. E-mail: *laurence.lamothe@toulouse.inra.fr*

Innovations are needed to improve the sustainability of livestock production systems, i.e. to reduce their environmental impacts while maintaining or increasing their economic viability. The feed represents the greatest part of the economic and environmental costs of poultry rearing (Boggia et al., 2010). Therefore, changing feeding practices, i.e. the choice of raw materials, seems one of the promising ways. Raw materials should be chosen according to their environmental impacts and availability. To increase the number of raw materials usable improves the flexibility of systems, limiting the reliance on price fluctuating products. The feeding of waterfowl in the production of "*foie gras"* is based in large part on corn as an energy source during both the rearing and the overfeeding periods. Sorghum (*Sorghum bicolor*) has been chosen to substitute corn in "*foie gras*" production. Indeed, this cereal has similar nutritional characteristics to corn's ones (Sauvant et al., 2004), but is more drought-resistant. Thus, it is an interesting candidate to reduce the vulnerability of French agriculture to the water shortage risk (Amigues et al., 2006), by reducing the need of irrigation. Arroyo et al. (2012) showed that sorghum could be used as goose feed during growing-finishing period (GF period) and during overfeeding period (O period) of "*foie gras*" production. The aim of this work was to evaluate with LCA method, the environmental impacts of the effects of substitution of corn by sorghum during GF and O periods on "*foie gras*" production. Attributional LCA was conducted on different scenarios of partial and total substitution of corn by sorghum, based on experimental data (Arroyo et al., 2012) and the running on average goose farms. Ecoinvent was used as the source of secondary data, but specific data were generated for corn, sorghum and goose productions. The impact categories were calculated using mainly CML2 method: eutrophication (EP, kg PO_4- eq.), climate change (CC, kg CO_2 eq.), acidification potential (AP, kg SO2 eq.), terrestrial ecotoxicity (TE, kg 1.4- DCB eq.), cumulative energy demand (CED, MJ), water use (WU, m^3) and land occupation (LO, m^2 per year). The functional unit was 1kg of "*foie gras*". The impact calculation was conducted using SimaPro_ 7.2 software and mass allocation approach. 1kg of *"foie gras"* from geese fed with sorghum as the only cereal during both GF and O periods induced lower environmental impacts than the "*foie gras"* from the corn fed geese (i.e.: CC: 1,323 vs. 1,471 kg CO_2-eq respectively). For all the impacts, the highest values were observed for 1kg of "*foie gras"* from geese fed with sorghum during the G period, due to higher bird mortality during O period (1,623 kg CO_2-eq). Using sorghum during the O period only did not affect the environmental impacts compared to the use of corn (1,427 kg CO_2-eq). Present results suggested that total substitution of corn by sorghum in goose diet offers interesting perspectives for more sustainable feeding strategy in the production of "*foie gras".*

References

Amigues, J.P., Debaeke, P., Itier, B., Lemaire, G., Seguin, B., Tardieu, F., Thomas, A. 2006. Sécheresse et agriculture. Réduire la vulnérabilité de l'agriculture à un risque accru de manque d'eau. Expertise scientifique collective, synthèse du rapport, INRA (France), 72pp.

Arroyo, J., Auvergne, A., Dubois, J.P., Bijja, M., Lavigne, F., Fortun-Lamothe, L., 2012. Influence of feeding sorghum on the growth, gizzard development and carcass traits of growing geese. Animal (in press)

Boggia, A., Paolotti, L., Castellini, C., 2010. Environmental impact evaluation of conventional, organic and organic-plus poultry production systems using life cycle assessment. World's Poultr. Sci. J. 66, 95-114.

Sauvant, D., Perez, J.M., Tran, G., 2004. Tables of composition and nutritive value of feed materials: pigs, poultry, cattle, sheep, goats, rabbits, horses, fish. eds INRA Editions, Paris and Wageningen Academic Publishers, The Netherlands. 304pp.

22. Nitrogen content allocation to handle co-products in livestock systems – case study on a poultry supply chain

Alexandre Thevenot[1,*], Jonathan Vayssières[2], Joel Aubin[3], Emmanuel Tillard[1]

[1] CIRAD Saint-Pierre, Reunion Island, France, [2] CIRAD Dakar, Senegal, [3] INRA UMR SAS, Rennes, France, [*] Corresponding author. E-mail: *alexandre.thevenot@cirad.fr*

In food sectors processes along the life cycle of a product can be multifunctional. ISO standards for Life Cycle Assessment specify rules in order to allocate the environmental burden between co-products. First recommendation is to avoid allocation with subdivision or system expansion. However when it is not possible, emissions and raw materials consumption allocation must reflect the physical relationship between products. Usually economic, mass or gross energy content allocation rules are used. But several problems remain for agricultural productions: economic allocation is highly sensitive to market fluctuations and mass and gross energy content allocations could lead to counter-intuitive results. Co-products may indeed weight or contain more energy than the product under study itself. For these points, allocation has always been considered as one of the most controversial issues in LCA and particularly for agricultural systems (Audsley et al., 1997).

Livestock productions are highly multifunctional (e.g. dairy farming produces milk, meat, and manure). In industrialised countries, its main function is the provision of proteins for human diet and its major environmental problems are linked to high nitrogen (N) losses occurring during manure management. For these reasons, we proposed in this study to compare results obtained with allocation rule based on product's nitrogen content with other classical allocation rules (Mass and economic allocation and economic allocation with system expansion to manure use). Effects of these different allocation rules were applied on a poultry supply chain in La Réunion (French Tropical Island). Allocation is applied at different production stages: i) breeders rearing where co-products are breeders and litter, ii) layer production with hatching eggs, cull animals and unfertilised eggs, iv) broiler production with broiler and litter, v) slaughterhouse vi) Incineration plant with production of feathers and blood meal as fertiliser and wastes management. For economic allocation we use the product price at process level. Manure price was estimated by on farm surveys. For system expansion, poultry litter was in this case replaced by mineral fertiliser which is imported from mainland France over ten thousand kilometres. The functional unit was defined as one tonne of chicken carcass at slaughterhouse gate. System boundaries are shown in Fig. 1. LCA was performed using CML 2 Baseline 2000 for Global Warming (GW), Energy Use (EU), Acidification Potential (AP) and Eutrophication Potential (EP) impact categories, and Cumulative Energy Demand method v1.08, all implemented in Simapro Software.

Impacts categories were significantly sensitive to the allocation rule (Fig. 2). Economic allocation leads to higher impact over all categories. System expansion reduced by 10% GW and EU and 5% EP and AP. Nitrogen content and mass allocation show results around 25% and 30% lower than economic allocation respectively. Most of differences were observed at farming stage with manure management.

Manure management patterns could differ a lot within a same territory that it is often difficult to establish a reasonable cost for economic allocation. Mass allocation has to be avoided because litter weight highly depends on moisture content. System expansion is not recommended in this case because of additionally maritime transport burden. Nitrogen content allocation seems to be an interesting option for livestock production environmental assessment and is in the range of other allocation rules. Finally, the choice of allocation rule for agricultural systems always depends on the manure value in the given system. Using this allocation rule, poultry litter takes however a high part of environmental burden of meat production, which seems consistent regarding its high value all over the world.

References

Audsley, A., et al., 1997. Harmonisation of environmental life cycle assessment for agriculture. Final report, Concerted Action AIR3-CT94-2028. European Commission DG VI, Brussels, Belgium.

Guinée, J.B., et al., 2002. Handbook on Life Cycle Assessment. An Operational Guide to the ISO Standards. Kluwer Academic Publishers, Dordrecht, The Netherlands.

ISO, 2006. ISO 14040—Environmental management—life cycle assessment—principles and framework. International Organisation for Standardisation, Geneva.

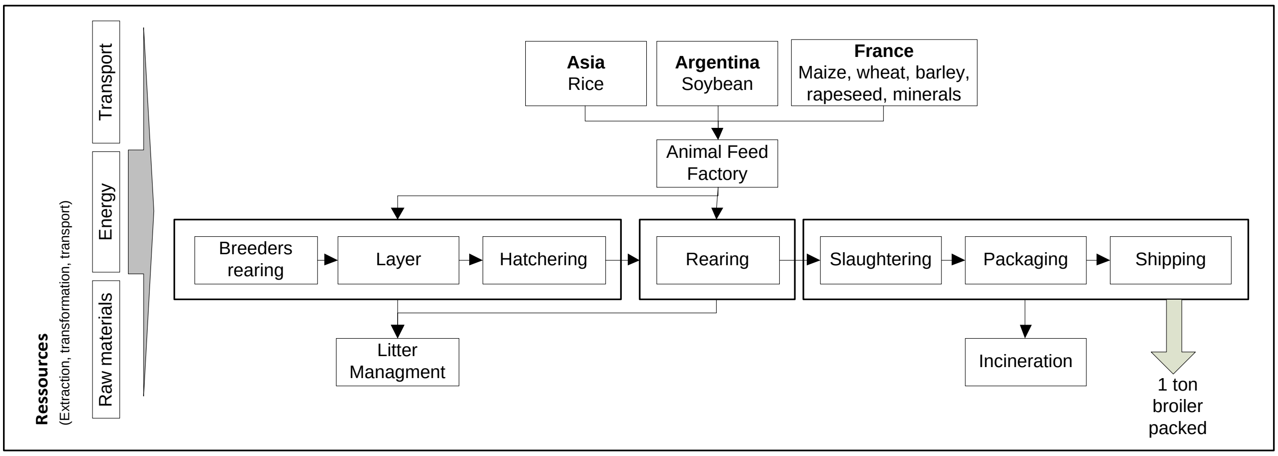


Figure 1. System boundaries for a cradle to slaughterhouse gate for 1 ton of broiler packed ready for transport

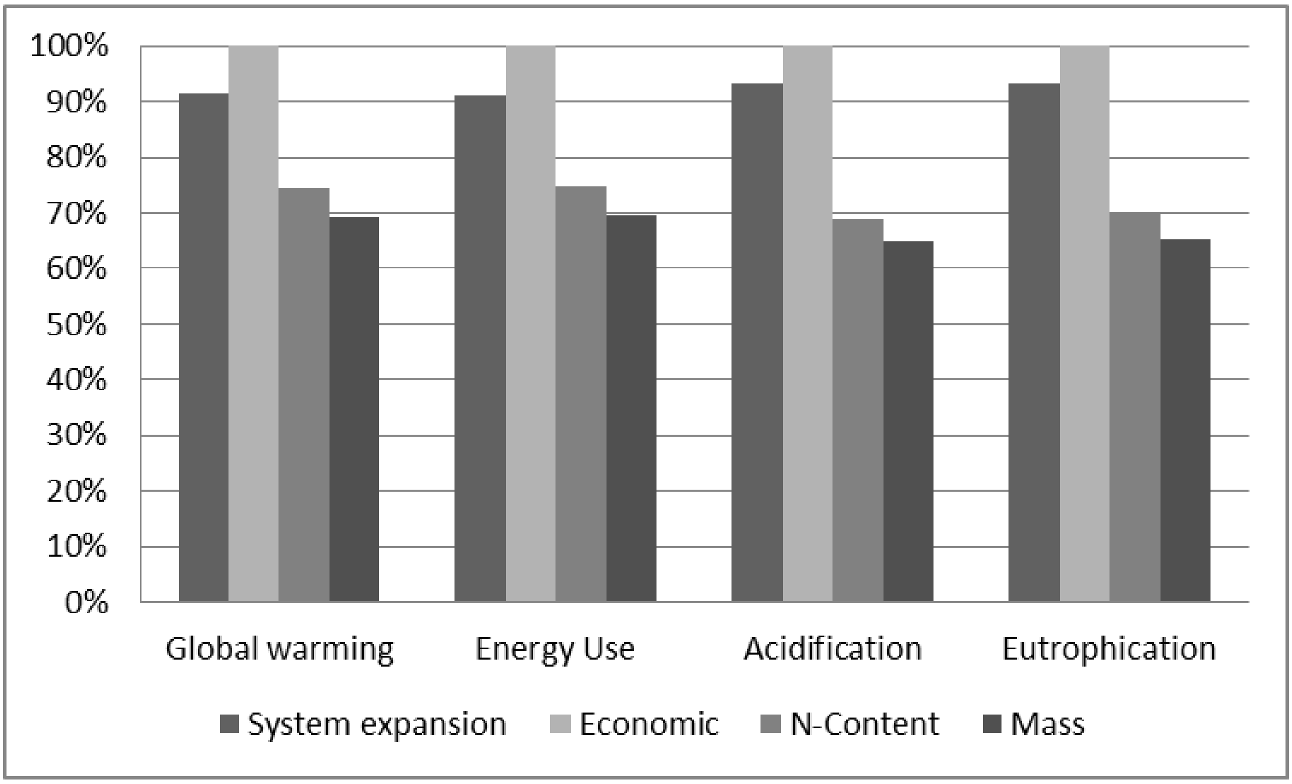


Figure 2. Results of impact assessment for 1 ton of broiler packed depending on the chosen allocation method

23. Influence of allocation methods and system boundaries in LCA of broiler production

Edivan Cherubini[1,*], Vamilson Prudêncio da Silva Junior[2], Sebastião Roberto Soares[1], Marco Antônio Santos[3]

[1] Universidade Federal de Santa Catarina, Florianópolis, Brazil, [2] Empresa de Pesquisa Agropecuária e Extensão Rural de Santa Catarina, Florianópolis, Brazil, [3] Eng. Agrônomo - Especialista em Gestão Ambiental Agroindustrial, Brazil, [*] Corresponding author. E-mail: *cherubini@ens.ufsc.br*

Although life cycle assessment is regulated by the ISO 14040 series, there are still several issues, such as described by Reap et al. (2008), in an LCA study that might generate questions and discussions. Definitions as system boudaries or allocation for example, which may induce to a misleading comparison between LCA studies. To develop this study we evaluated the broilers production in southern Brazil, ranging the composition of the chicken diet in two scenarios: (i) feeds with use of by-products derived in the abattoir process (animal origin), and (ii) substitution of by-products by an increase of soybeans (vegetable broiler). First were evaluated the broiler with the system boundary comprising the extraction of raw materials used for growing grain in the diet of chickens to the farm gate, and then were expanded the boundaries to the port for export, including the slaughtering process. This change in the boundaries makes necessary the use of allocations methods in the abattoir stage which depending of the adopted procedure adds high sensitivity in the final LCA results. Luo et al. (2009) studied the allocation’s influence in LCA of ethanol from corn and concluded that the results are highly sensitive to the allocation method and a challenge from a scientific point of view. Therefore, the aim of this paper is to demonstrate the difference in the final results in a product’s LCA using two different methods of allocation, mass and economic in the vegetable broiler, also intending to demonstrate the importance of defining the system boundaries. For the broiler with animal protein were used the mass allocation. The functional units were a ton of broilers live weight and a ton of broiler slaughtered, eviscerated and frozen at the port. The impact categories used for life cycle impact assessment were global warming potential from CML 2 baseline 2000 method plus the total cumulative energy demand. The results showed that the broiler feed with chicken by-products (animal protein) has a better environmental performance than the chicken with vegetable diet (without chicken by-products) for the evaluated impact categories. When the system boundary is increased for the broiler slaughtered at the port for export, it shows the high sensitivity of the results depending on the allocation procedure used in the slaughter process the outputs can either improve as getting worse the environmental performance of the vegetable broiler, as shown in Table 01 and Figure 01.

References

Associação Brasileira de Normas Técnicas – ABNT, 2009. NBR ISO 14044 Gestão Ambiental – Avaliação do ciclo de vida – Requisitos e Orientações. Brasil.

Luo, L., van der Voet, E., Huppes, G., Udo de Haes, H. A., 2009. Allocation issues in LCA methodology: a case study of corn-stover-based fuel ethanol. International Journal of Life Cycle Assessment, 14, 529-539.

Reap, J., Roman, F., Duncan, S., Bras, B., 2008. A survey of unresolved problems in life cycle assessment: Part 1: goal and scope and inventory analysis. International Journal of Life Cycle Assessment, 13, 290-300.

Table 1. Results for 1 ton of frozen broiler.

Life Cycle	Broiler with animal protein		Vegetable Broiler (MA)[a]		Vegetable Broiler (EA)[b]	
	GWP[c]	*CED*[d]	*GWP*[c]	*CED*[d]	*GWP*[c]	*CED*[d]
Total maize	*39.8*	*32.6*	*32.1*	*26.*	*39.5*	*32.6*
Total soybeans	*34.6*	*26.8*	*32.2*	*25.1*	*40.2*	*31.1*
Other feed ingredients	*7.3*	*14.9*	*6.3*	*9.6*	*8.2*	*12.1*
Feed subtotal	81.6	74.4	70.5	61.0	87.9	75.9
Eggs to hatch	6.3	8.5	5.4	7.2	6.4	8.6
Day-old chicks	0.4	0.8	0.4	0.1	0.5	0.8
Live poultry	17.4	22.5	15.1	19.6	17.5	22.6
Livestock transportation	3.3	4.6	2.5	3.7	3.3	4.7
Slaughter	3.5	7.7	3.2	6.5	3.5	7.7
Packaging	1.4	4.9	1.4	4.9	1.4	4.9
Total	113.9	123.3	98.6	103.2	120.5	125.0
Avoided fertiliser	13.9	23.3	11.5	19.3	14.1	23.7
Total	100.0	100.0	87.1	83.9	106.4	101.3

[a] Mass allocation., [b] Economic allocation., [c] Global Warming Potential, in%., [d] Cumulative Energy Demand, in%.

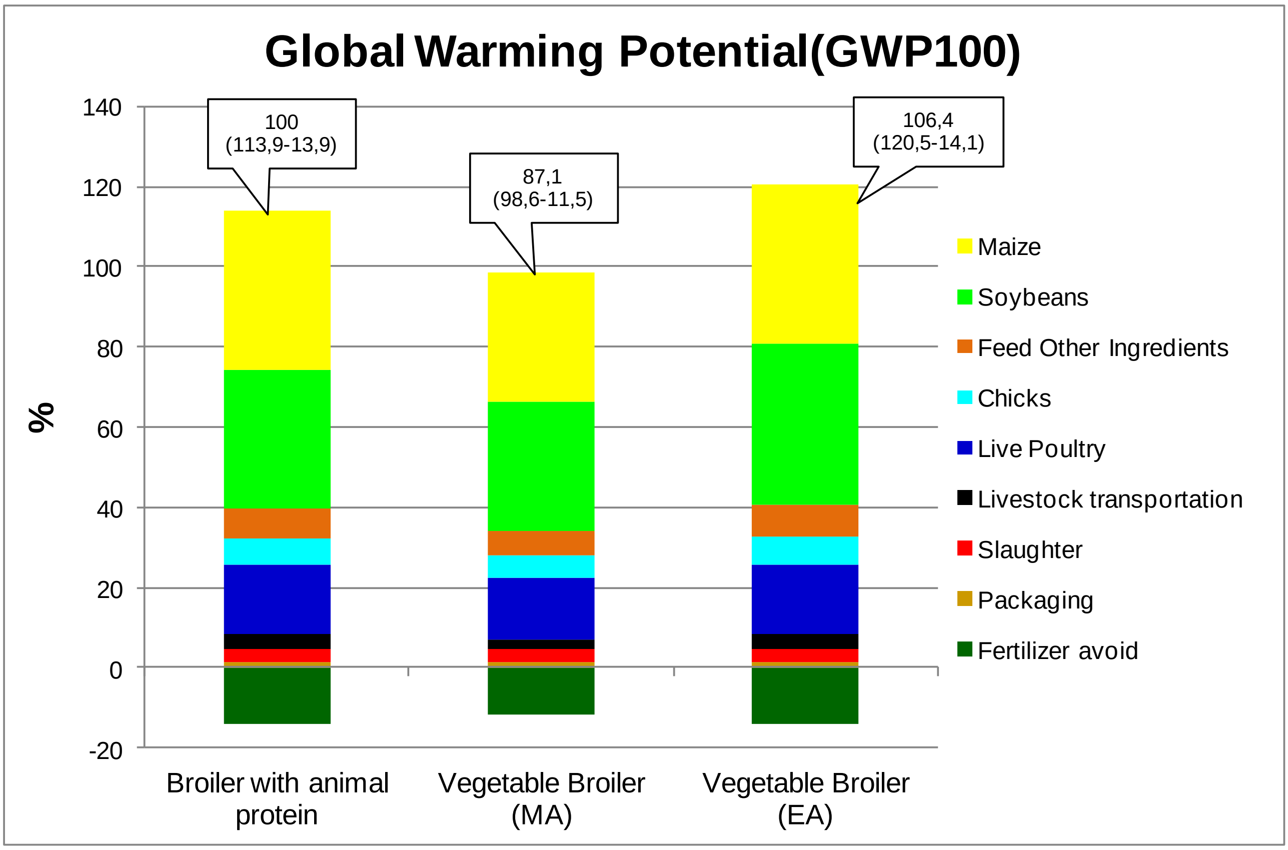


Figure 2. Comparison of CO_2 eq emissions of 1 ton of frozen broiler, in%. Considering the scenario of Broiler with animal protein as base of comparison, showing the difference in the results of the vegetable broiler with the allocations methods applied.

24. Reduction of GHG emissions from broilers fed a phytogenic additive

Stefan Hörtenhuber[1,2,*], Martin Veit[3]

[1] Research Institute of Organic Agriculture (FiBL) Austria, [2] University of Natural Resources and Life Sciences Vienna, Department of Sustainable Agricultural Systems, [3] Delacon Biotechnik GmbH, Steyregg, Austria, [*] Corresponding author. E-mail: *stefan.hoertenhuber@fibl.org*

Mitigation of greenhouse gas (GHG) emissions from broiler production may be achieved by various options, including increased performance, changes in diets and adaptations for housing and litter management systems (LMS). The overall aim of this study was to analyse GHG emissions from broiler production in Austria including a quantification of the impact of feeding the phytogenic feed additive “Biostrong® 510” (BSG; Delacon Biotechnik GmbH, Steyregg, Austria). Methods and emission factors for calculation of emitted NH_3 and GHG are based on IPCC (2006), Anderl et al. (2011), and Hörtenhuber et al. (2011). The reducing effect concerning NH_3 emissions is derived from experimental data (Jelinek et al. 2004). Performance data and data on nitrogen excretion were taken from van Krimpen (2011).

The calculated CO_2-equivalents per kg BSG are about 2.0 kg. The inclusion of BSG in broiler feed (150 ppm) does not increase the feed production-related GHG emissions (less than 0.1%) per ton of feed. Emissions from the basal diet (corn, soybeans, wheat) were found to be the most important drivers for GHG emissions for broilers (see Fig. 1). It was concluded that GHG emissions per kg live weight could be reduced by 5% (4 to 6%, depending on the origin and production methods for feedstuffs and raw materials), if the corn-soy-wheat diet was supplemented with BSG. According to van Krimpen (2011), who analysed 18 comparable trials with BSG, this reduction is achieved by: (1) Improved digestibility and thus a better feed conversion ratio (contributing 60% of the reduction), which also results in (2) less excreted nitrogen (responsible for 20% of the reduction); (3) saponins in BSG directly inhibit NH_3 formation (Weber et al., 2012; 20% of reduction). Considering that the use of this feed additive does not demand for major changes in supply chains or cost-intensive investments, the reduction of about 5% of GHG emissions is remarkable. Furthermore, it is possible to combine the feed additive’s effect with other mitigation options (e.g. adaptations in LMS). The improved feed conversion and lower mortality (see van Krimpen, 2011) also result in a higher profit.

References

Anderl, M., Freudenschuß, A. , et al., 2011. Austria's National Inventory Report 2011. Submission under UNFCC and under the Kyoto Protocol. Vienna, Austria.

Hörtenhuber, S., Lindenthal, T., Zollitsch, W., 2011. Reduction of greenhouse gas emissions from feed supply chains by utilizing regionally produced protein sources - the case of Austrian dairy production. Journal of the Science of Food and Agriculture 91, 1118-1127.

IPCC (International Panel on Climate Change), 2006. Guidelines for National Greenhouse Gas Inventories. [Available at www.ipccnggip.iges.or.jp/public/2006gl/index.html].

Jelinek, A., Dědina, M., Plíva, P., Souček, J., 2004. Research of biological agents effects on reduction of ammonia concentration in stables of intensive farm animals breeding. Res. Agr. Eng., 50(2), 43-53.

van Krimpen, M., 2011. Assessing the effect of the phytogenic feed additive Biostrong® 510 on performance and ammonia emissions in broilers. Confidential report 281. ISSN 1570 - 8616.

Weber, T., Veit, M., Wendler, K.R., Weghuber, J., Höglinger, O., 2012. Evaluation of ammonia reduction potential of saponin containing plant materials by measurement of urease inhibition. In: Mair, C., Kraft, M., Wetscherek, W., Schedle, K. (Eds.), Proc. of the 11th BOKU-Symposium Tierernährung. Vienna, Austria, 19 April 2012. University of Natural Resources and Applied Life Sciences Vienna, Austria. pp. 213-215.

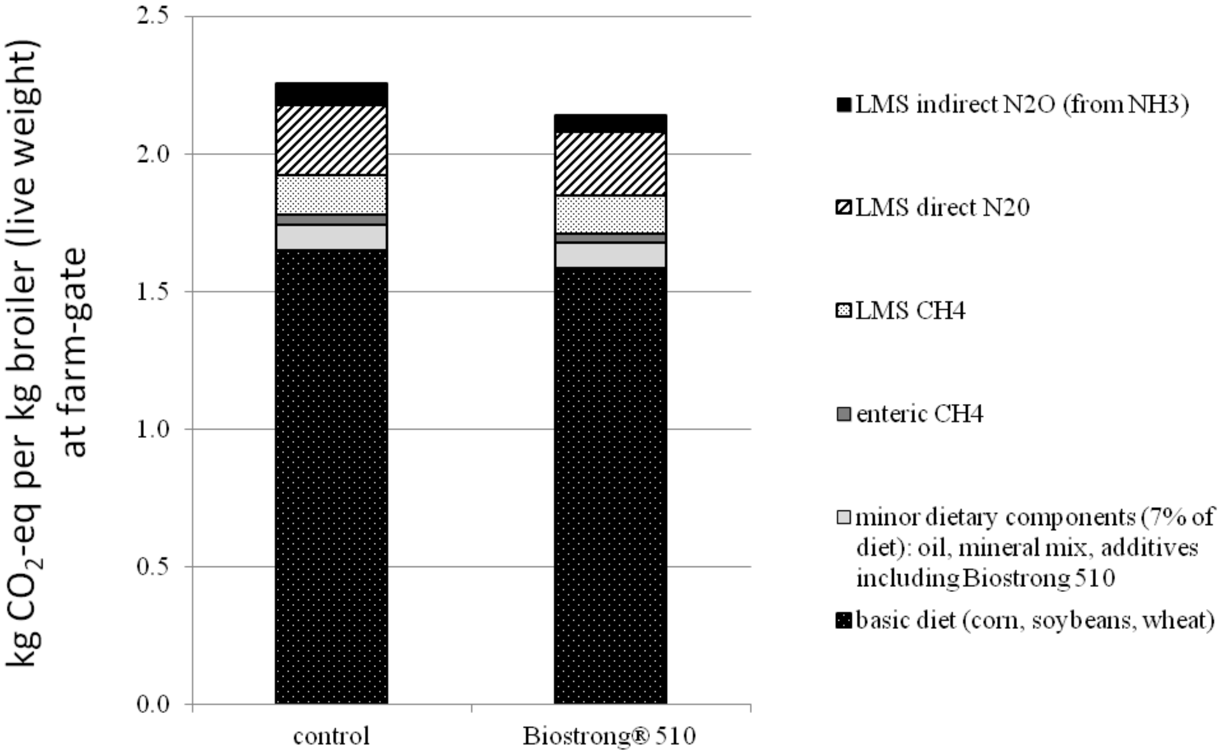


Figure 1. GHG emissions per kg of broiler with and without Biostrong® 510 supplementation (kg CO_2-eq per kg of live weight at farm gate before slaughtering).

25. Allocation between high value co-products from livestock: a case study from Australian sheep production

Stephen Wiedemann

FSA Consulting, Toowoomba, Queensland, Australia, E-mail: *stephen.wiedemann@fsaconsulting.net*

Much of the Australian sheep industry is based Merino sheep, which produce high value wool and also sheep meat. In contrast to many other sheep producing regions of the world, wool is the primary product from Merino sheep, though sheep meat is also an important and high value co-product. Within the meat production component of the system, two grades of meat are produced; high grade (HG) meat from lambs and low grade (LG) meat from older cull-for-age (CFA) breeding animals. These co-production issues were investigated in a 'farm-gate' study of Australian sheep production. Handling of co-products for sheep systems has been investigated previously by Eady et al. (2011) for an Australian Merino sheep farm. Eady et al. (2011) investigated a biophysical allocation process based on partitioning the feed consumed by the breeding flock to either wool or lamb production and an economic allocation process. Impacts were found to be similar using either approach. Eady et al. (2011) determined that system expansion was difficult to apply to a sheep system because of the sensitivity around choices with substitution products for the meat. They also noted the difficulty in selecting substitution processes for other outputs from the flock such as rams, which are sold for breeding purposes. All other studies reviewed by the author for sheep production have used economic allocation to handle wool / sheep meat, though most of these studies have been for production systems where the wool produced is of low quality, and sheep meat is the primary product. Hence, no study has yet applied system expansion to Merino sheep production. A similar co-production issue exists in the dairy industry, where the cow herd produces both milk, calves (that enter the beef production system) and CFA cows (beef). Two studies (Flysjö et al. 2011; Cederberg and Stadig 2003) have specifically investigated the sensitivity of methodology choices around this co-production issue, applying a variety of methods including economic allocation, biophysical allocation and system expansion. Cederberg & Stadig (2003) argue that system expansion is a logical approach for accounting for beef produced in dairy systems, because this product directly enters the beef market affecting supply and demand. Both Cederberg & Stadig (2003) and Flysjö et al. (2011) note that using system expansion to handle meat (by substituting with beef from purpose grown beef herds) resulted in lower environmental burdens for the milk product. Co-production in dairy farming has some similarities to Merino sheep production, though sheep meat from Merino production is more important than meat from dairy systems in terms of total mass of product and economic value.

The aim of the study was to investigate the sensitivity of co-production decisions by applying three methods; economic allocation, a simplistic biophysical allocation based on the total mass of product, and system expansion. The farm selected for the study was a small Merino producer from a high rainfall region in New South Wales. The farm had 1500 ewes producing 4.1 kg wool / head.yr and 1200 lambs (80% weaning rate).

Allocation between HG and LG meat is not unique to sheep production. From a review of the literature, few beef and pork studies were found that differentiated between meat from young animals and older CFA animals. Allocation between these meat products depends on the definition of product function. The main differences between HG and LG meat relate to eating quality factors such as tenderness, meat colour and flavour, not the nutritional properties (mass of energy, protein etc). Hence, if the focus of a study is the provision of nutrition for human consumption, it is reasonable to group HG and LG meat together as they are functionally comparable. Further to this, the quality factors associated with HG and LG meat are market specific. For example, some Australian sheep meat markets (such as the Middle East) prefer LG sheep meat because the flavour is considered superior to HG meat. Applying an allocation process (such as economic allocation) therefore introduces market preferences and a wide range of quality factors, which need be reflected in the definition of the functional unit. This study chose to consider meat from HG and LG meat functionally equivalent, thereby avoiding the need for allocation at this point.

Co-production of wool (greasy weight) and sheep meat (live weight) was handled using three approaches; economic allocation, mass allocation and system expansion. Economic and mass allocation factors are provided in Table 1. The system expansion approach followed a similar approach used in the dairy industry by Cederberg and Stadig (2003). Meat from Merino systems enters the lamb and mutton supply chain in Australia where it is considered functionally equivalent (on a nutritional basis) with meat from 'purpose grown meat sheep' flocks. 'Purpose grown meat sheep' is used here to describe production systems that focus on meat production, which tend to use different sheep breeds that produce lower quality wool that has negligible value. Some purpose grown meat sheep systems produce no saleable wool because breeds have been selected that naturally shed their wool each year. Considering meat from Merino flocks is not differentiated in the meat supply chain (post slaughter), meat from purpose grown meat sheep was considered an appropriate substitution product.

Comparison of the three methods showed a four-fold difference in greenhouse gas (GHG) emissions for wool. The system expansion method resulted in total GHG of 7.6-9.2 kg CO_2-e / kg greasy wool. Economic allocation resulted in total GHG of 31.7-33.8 kg CO_2-e / kg wool. The simple mass allocation approach resulted in total GHG emissions of 8.1-8.3 kg CO_2-e / kg greasy wool, which was similar to the results using system expansion. The difference between the system expansion and economic allocation results were similar, though much more pronounced, than the findings of Flysjö et al. (2011) for dairy production, which showed system expansion to generate the lowest impacts for the primary product.

Economic allocation was sensitive to annual and cyclical changes in the value of wool and sheep meat. This changed the GHG emissions allocated to wool by ± 30% between different years (over a five year period). While mass allocation is generally not favoured, it has some merit for Merino systems. Because wool and meat are closer to a joint production system than a typical primary product/by-product system, it follows that the burdens should be allocated in a more even manner. Following a biological causality approach, wool and meat are both protein based products that require broadly similar processes within the animal for production. This offers a simple alternative to system expansion while generating similar results.

This study concluded that allocation was not required to differentiate between HG and LG meat, and highlighted the sensitivity of allocation processes between wool and sheep meat. System expansion offers a useful approach that reflects the dynamics of the Australian sheep meat market well, and is considered the most suitable approach for further research in this industry.

References

Cederberg C & Stadig M. 2003. System expansion and allocation in Life Cycle Assessment of milk and beef production. Int J LCA 8 (6) 350-356

Eady, S., Carre, A., Grant, T., 2011. Life cycle assessment modelling of complex agricultural systems with multiple food and fibre co-products, J. Cleaner Prod. Volume 28, June 2012, Pages 143–149

Flysjö, A, Cederberg, C, Henriksson, M & Ledgard, S 2011, How does co-product handling affect the carbon footprint of milk? Case study of milk production in New Zealand and Sweden, Int. J or LCA, vol 16, 420-430.

Table 1. Co-products and allocation factors for Merino wool production

Products	Mass Allocation Factors	Economic Allocation Factors	System expansion substitution products
Wool (greasy wool) kg	14-15%	55-65%	
Sheep sales (lamb + mutton – Live weight basis) kg	85-86%	35-45%	Purpose grown lamb and sheep meat from sheep meat enterprises. Substitution applied using a factor of 95% to account for higher dressing percentage of purpose grown sheep compared to Merinos

26. How to decrease farmed fish environmental burdens through feed formulation and feeding management

Assunta Filareto[1,*], Massimo Marino[1], Umberto Luzzana[2], Sonia Pignatelli[1]

[1] Life Cycle Engineering, [2] Skretting, [*]Corresponding author. E-mail: *filareto@studiolce.it*

The aim of this work is to quantify improvements in farmed fish (European seabass *Dicentrarchus labrax* and gilthead seabream *Sparus aurata*) environmental burdens resulting from the application of latest know-how in feed formulation and feeding management.

Environmental impacts from the whole aquaculture production chain were assessed according to the life cycle assessment (LCA) methodology in order to identify critical areas. The present paper reports results in terms of greenhouse gases emissions (GWP) and considerations about the reduction of pressure on wild fish stock.

The LCA was performed per 1 kg of farmed fresh fish considering the whole production chain from hatchery to fresh fish distribution platform (including the raw materials used for feed production). Results from this study show that the most significant phases in terms of GHG emissions are feed production and farming (Fig. 1).

On the basis of the results from the LCA, critical areas for improvement were identified, and the effects of possible actions quantified on the basis of practical data from the industry, with specific reference to the feed as the main variable affecting environmental burdens. Our study confirms that FCR improvement (through the use of nutritionally balanced formulations and careful feeding management) is the most efficient strategy to reduce environmental burdens (Fig. 2), together with flexible use of raw materials. Moreover, improved FCR, together with advanced nutritional know-how and freedom in raw materials choice, results in reduced fish meal and fish oil consumption in farmed fish production, hence alleviating pressure on wild fish stocks. As consumers recognise the importance of sustainability and the need to reduce environmental impacts from their food, the adoption of strategies such as the ones mentioned above should be promoted in specifications for high-value farmed fish.

References

Jackson, A., 2009. Fish in–Fish out ratios explained, Aquaculture Europe Vol. 34 (3).

van den Burg S.W.K. et al., Environmental performance of wild-caught North Sea whitefish, a comparison with aquaculture and animal husbandry using LCA, LEI report 2011-090, January 2012, Project code 2272000225, LEI, part of Wageningen UR, The Hague.

Winther U., Ziegler F., Skontorp Hognes E., Emanuelsson A., Sund V., Ellingsen H., 2009, Carbon Footprint and energy use of Norwegian seafood products, SINTEF Fisheries and Aquaculture.

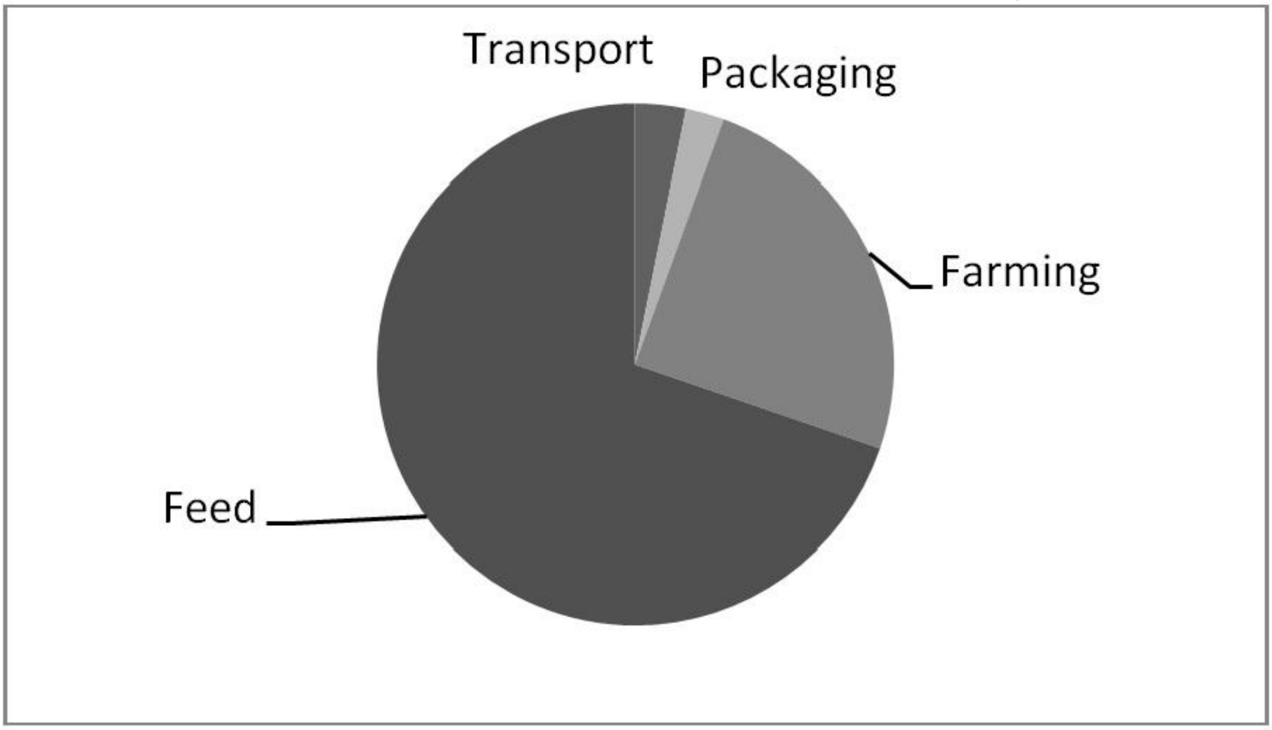


Figure 1. GHG Emissions per kg of fresh fish.

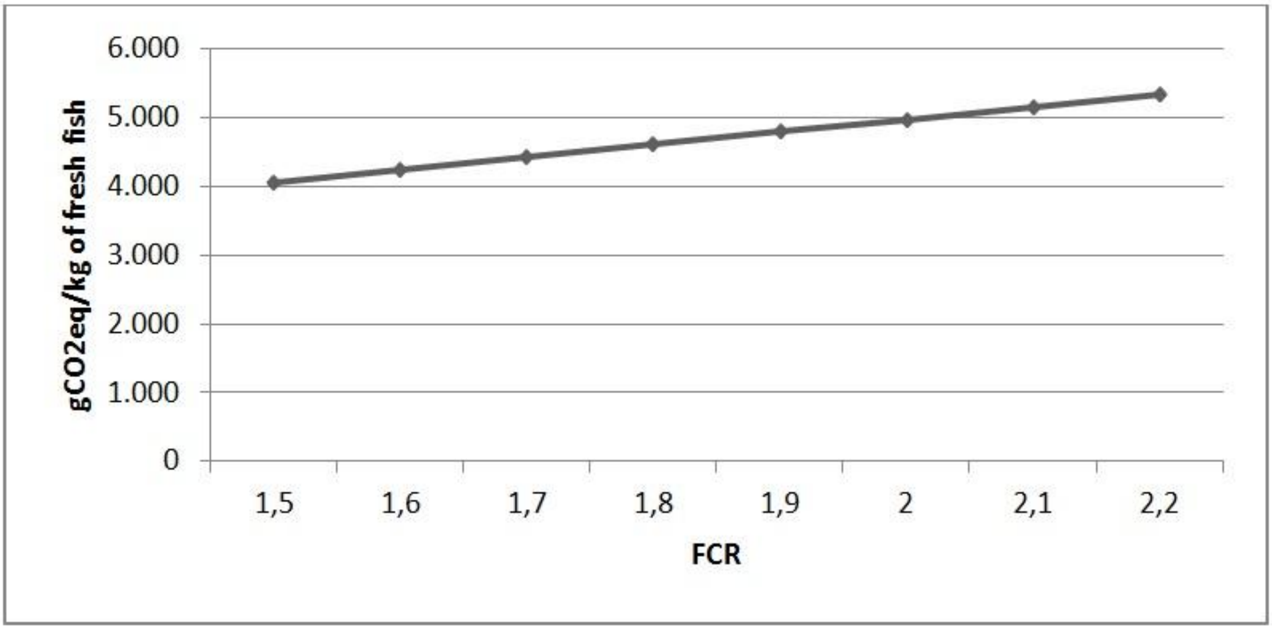


Figure 2. Correlation between the feed-consumption ratio (FCR) and GHG emissions.

27. A multi-scale method for assessing ecological intensification in aquaculture systems

Aurélie Wilfart[1,2,*], Jéhane Prudhomme[1,2,3], Jean-Paul Blancheton[3], Joël Aubin[1,2]

[1] INRA, UMR1069 Sol Agro et hydrosystème Spatialisation, F-35000 Rennes, France, [2] AGROCAMPUS OUEST, F-35000 Rennes, France, [3] IFREMER, Laboratoire de Recherche Piscicole de Méditerranée, Chemin de Maguelone, 34250 Palavas-les-Flots, France, [*] Corresponding author. E-mail: *aurelie.wilfart@rennes.inra.fr*

To meet the challenges of producing more while lowering impacts on ecosystems, new farming systems have to be designed. To define development strategies, a multi-scale assessment method that estimates the tradeoff between human demand and natural services, as well generates consistent performance indicators on utilisation of natural resources and environmental emission levels based on the same set of input data is needed. LCA estimates resource use and potential environmental impacts throughout a product's life cycle at global and regional scales (ISO, 2006) but does not consider the provision of ecosystem services or products (Ulgiati et al., 2006). Emergy accounting (EA) is an ecology-based tool developed to integrate all system inputs (environmental and economic values) using a common unit, solar emergy joule (Odum, 1996). EA inserts the productive cycle into a local environmental context and quantifies the energy flows between the environment and the production system. Through three contrasting fish-farming systems, we attempted to demonstrate the interest of a combination of LCA and EA to define the major components of environmental sustainability and ecological intensification of fish farming and more globally of agricultural systems.

The first system is a recirculating system (RSF) of Atlantic salmon depending highly on external inputs (feed and energy). The second one is extensive fish polyculture in a pond (PF1) with few external inputs. The last one is a small pond farm with use of external feeds. These systems were assessed according the ISO standards for attributional LCA during one production year. The assessment covered farm operations and transportation at all stages. Local emissions of nutrients were estimated using nutrient balance modeling and pond emissions were refined to include nitrogen-fate factors. LCA results are presented as traditional midpoint indicators according CML 2 baseline 2001and are expressed by tonne of fish produced. Emergy accounting [3] is based on LCA system definition but includes also the contributions of natural systems (sun, rain, groundwater, etc.) and provide indicators to evaluate the efficiency of energy use and its quality during the lifecycle. The chosen Emergy indicators are: Percentage of renewability (%R); the Emergy Yield Ratio (EYR, ability to rely on local resources; Environmental Loading Ratio (ELR, level of exploitation of nonrenewable resources compared to renewable ones).

For 1 tonne of living fish, RSF had higher potential impacts for NPPU and all the Emergy indicators (Fig. 1). PF2 had higher potential impacts in comparison with PF1 except for water dependence. However, RSF had lower potential impacts for climate change, eutrophication, land competition and water dependence than ponds, which reflects the level of intensification of the systems. The consumption of energy (calculated by LCA, Figure 2) was similar for RSF and PF1 and higher for PF2. But, the contributors to this impact differed among the systems (direct energy use for ponds and feeds and direct energy used for RSF). The difference in%R between systems was due to water origin: for RSF water was pumped whereas for ponds it came essentially from rain and water run-off. PF1 has a higher EYR, which means that it depends less on market resources than RSF. The RSF higher value of ELR (7.98) indicates a moderate environmental impact.

The combination of LCA and Emergy accounting on contrasting systems provides a perspective of what ecological intensification could mean in aquaculture: a decrease in potential impacts per unit mass of final products, especially for global warming, eutrophication and acidification; a decrease in dependence on market-based and external resources; and an increase in the use of renewable natural resources and input efficiency. This is particularly true for choices regarding feed ingredients and the origin of energy sources.

References

ISO, 2006. Environmental Management -Life Cycle Assessment- Principles and Framework. ISO 14040.

Odum HT. Environmental Accounting: Emergy and Environmental Decision Making. New York: Wiley; 1996.

Ulgiati S, Raugei M, Bargigli S., 2006. Overcoming the inadequacy of single-criterion approaches to Life Cycle Assessment. Ecol. Modell., 190:432-42.

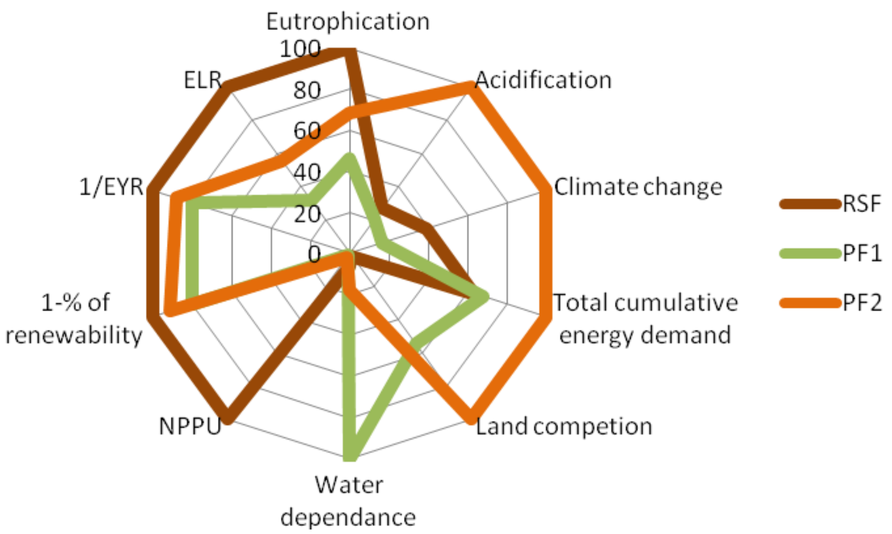


Figure 1. Comparison of the LCA and emergy impacts of the recirculating-system farm (RSF), large pond farm (PF1) and small pond farm (PF2). Impacts are represented as a percentage of the largest impact in each category. Certain emergy indicators were inversed accordingly. NPPU - net primary production use; EYR - Emergy Yield Ratio; ELR - Environmental Loading Ratio

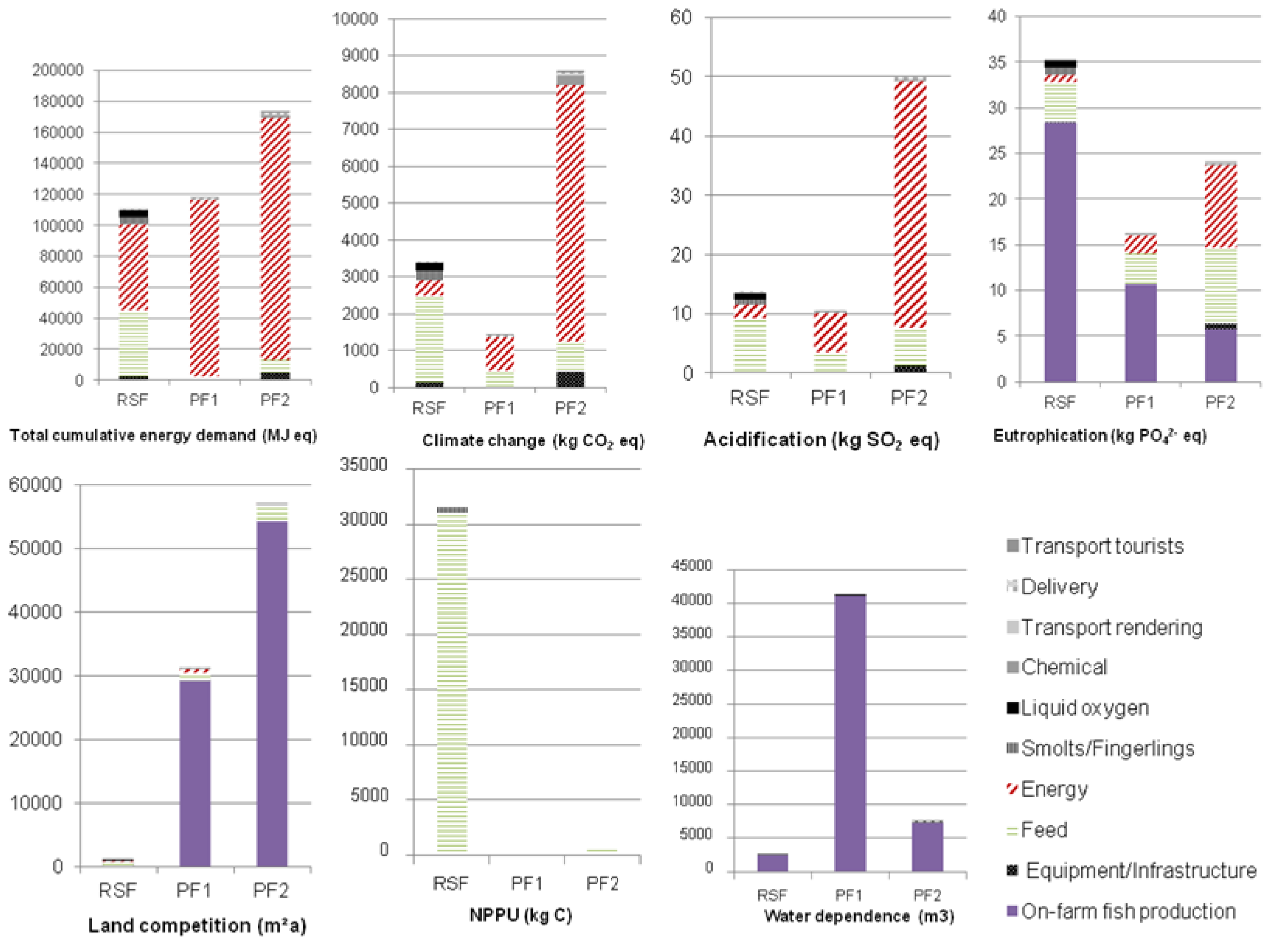


Figure 2. Environmental impacts (climate change, total cumulative energy demand, water dependence, acidification, eutrophication, and land competition) per tonne of fish of the recirculating-system farm (RSF), large pond farm (PF1) and small pond farm (PF2) calculated by LCA. For each environmental impact, the contribution of each input or production stage is indicated.

28. A participatory approach framework to integrate social aspects in LCA: the case of aquaculture systems

Syndhia Mathé[1,*], Joël Aubin[2], Aurélie Wilfart[2], Hélène Rey-Valette[3], Myriam Callier[4], Jean-Paul Blancheton[4]

[1] CNRS, UMR5474 LAMETA, F-34000 Montpellier, France, [2] INRA, UMR1069 Sol Agro et hydrosystème Spatialisation, F-35000 Rennes, France, [3] Université Montpellier 1, UMR5474 LAMETA, F-34000 Montpellier, France, [4] IFREMER Palavas, F-34250 Palavas les flots, France, [*] Corresponding author. E-mail: *syndhia.mathe@univ-montp1.fr*

The identification of relevant social impact indicators for a Social Life Cycle Assessment (SLCA) is still difficult and poorly documented (Jorgensen, et al., 2009). It requires the identification of the main social concerns for each considered case study and the adaptation of selected social impacts corresponding to the actual social situation and which can be easily appropriated by the actors of the value chain.

In the case of aquaculture system the studies based on social aspects are essentially focused on manpower or on conflicts with other activities. In the PISCEnLit project (Ecologically Intensive PIScículture, funded by the French National Research Agency), we aim at emphasising a larger vision of the social impacts of fish farming systems using a new approach of SLCA. We studied fish farming pond systems in France and Brazil. In this study, we focussed on the choice of the impact categories using the participation of stakeholders (James et al., 2002). We emphasised the role of a participatory approach to identify and select the relevant social impacts to be assessed. From a practical viewpoint, the proposed approach consists in implementing surveys and focus groups about the social representations at different stages of the assessment process. Through this process, the opinions of the stakeholders about potential or real social impacts of aquaculture may be taken into consideration. However, the technical construction of the relevant impact indicators allowing evaluation of the impacts must be done by the researchers in the project.

Our presentation focuses on the advantage of using a participatory approach based on the Principle, Criteria and Indicator (PCI) method (Rey-Valette et al., 2008) to identify relevant social indicators for a SLCA in fish farming pond systems cases. This method provides a basis for discussion, allowing the stakeholders to rank and validate a list of impacts (Fig. 1). This list was constructed using international conventions (e.g. Human Rights Declaration), the well-being components of the Millennium Ecosystem Assessment, social aspects which appear in referential of aquaculture sustainable development (Rey-Valette et al., 2008) and results of the survey based on social representation. The adaptation of this method to the social LCA allows the comparison of different systems at the level of the principles without standardisation of social impacts.

References

James, K. L., Grant, T., Sonneveld, K., 2002. Stakeholder involvement in Australian paper and packing waste management LCA study. Int J Life Cycle Assess. 7, 151-157

Jorgensen, A., Hauschild, M. Z., Jorgensen, M. S., Wangel, A., 2009. Relevance and feasibility of social life cycle assessment from a company perspective. Int J Life Cycle Assess. 14, 204-214

Rey-Valette, H., Clément, O., Aubin, J., Mathé, S., Chia, E., Legendre, M., Caruso, D., Mikolasek, O., Blancheton, J-P., Slembrouck, J., Baruthio, A., René, F., Levang, P., Morrissens, P., Lazard, J., 2008. Guide to the co-construction of sustainable development indicators in aquaculture. © Cirad, Ifremer, INRA, IRD, Université Montpellier 1. Diffusion Cirad-Montpellier, Montpellier.

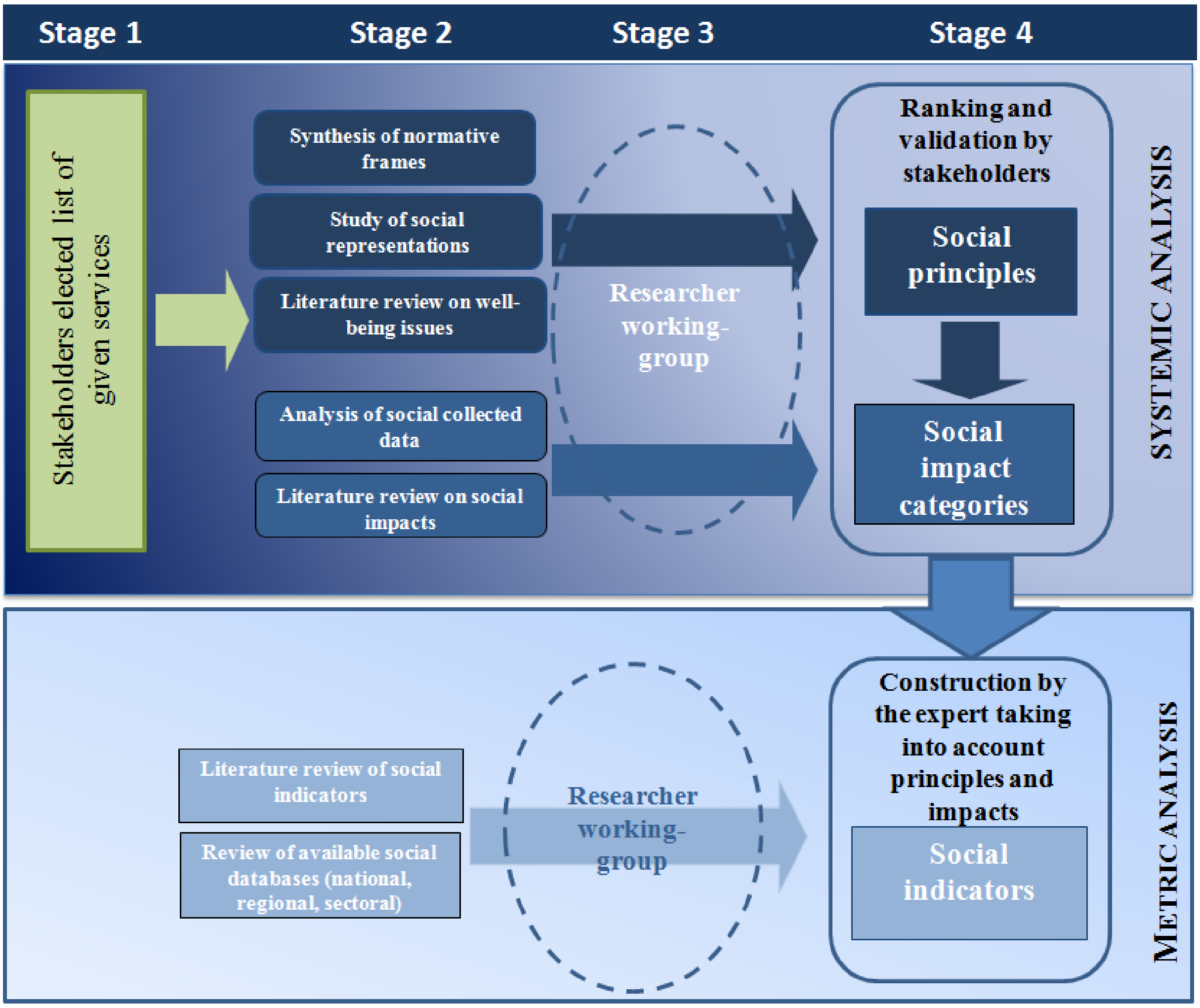


Figure 1. Methodological process integrating a participatory approach.

29. Comparing environmental impacts of wild-caught and farmed fish products - a review of life cycle assessments

E.A. Groen*, T.C. Viets, E.A.M. Bokkers, I.J.M. De Boer

Animal Production Systems Group, Wageningen University, * Corresponding author. E-mail: *Evelyne.Groen@wur.nl*

Many discuss the differences between fish products originating from aquaculture and fisheries. However, no systematic literature overview has been made yet. Most seafood LCA studies so far are limited to impact categories like energy use or global warming, or assess products on a national scale. We came up with a more extensive literature overview of the differences in environmental impact between aquaculture and fisheries, based on the same methodology.

The objective of this study was to compare environmental impact assessments of wild-caught and farmed fish products. We reviewed life cycle assessments (LCAs) of wild-caught plaice and cod, and farmed salmon, tilapia and pangasius, as these species are studied most often. Seven peer-reviewed studies were found that performed an LCA of these species, addressing diverse production systems. The following environmental impacts were discussed: energy use (EU), global warming potential (GWP), acidification potential (AP), eutrophication potential (EP) and land use.

To enable a comparison of EU and GWP among studies, we recalculated outcomes using the same functional unit (i.e. 1 kg of fresh fillet), allocation method (i.e. mass allocation) and similar characterisation factors. Most articles, however, did not address AP, EP, or land use. We estimated the AP, EP, and land use of the seven studies, using published technical parameters, complemented with data from *ecoinvent v2.2* and FAO. Next, the two systems were compared for GWP, EU, AP, EP, and land use using Wilcoxon rank-sum test.

Energy use (for wild-caught and farmed fish) varied between 11 to 273 MJ/kg of fresh fillet, whereas GWP varied between 0.7 and 22.9 kg CO_2-eq/kg of fresh fillet. Results of EU and GWP showed a similar pattern across studies, as especially in fisheries GWP is dominated by CO_2 emissions from fossil fuel combustion. The GWP from processing varied between 0.03 and 0.93 kg CO_2-eq/kg of fresh fillet (the fish with the highest value included freezing), the GWP due to transport between 0.05 and 3.36 kg CO_2-eq/kg of fresh fillet (the highest value comprised air transport).

Global warming potential, EU, and AP of farmed fish products were not different compared to wild-caught fish products ($P>0.05$; Fig. 1 shows GWP). However, EP and landuse were higher for farmed fish products ($P<0.05$; Fig. 2 shows EP). The EP was higher mainly due to ammonia emissions and leaching of nitrate during cultivation of feed ingredients. Land use was higher mainly due to the share of land that was required for aquaculture to cultivate feed ingredients. Differences in environmental impacts within each production system offer potential for improvement options. We do realise that the fishing industry affects the ocean, ocean floor and its biodiversity. These impacts have been roughly incorporated in a few studies, but are not yet established as to be applied in common seafood LCAs.

Based on this literature review, we concluded that GWP, EU and AP did not differ between farmed fish and wild-caught fish fillet, whereas EP and land use was higher for farmed fish compared to wild-caught fish fillet.

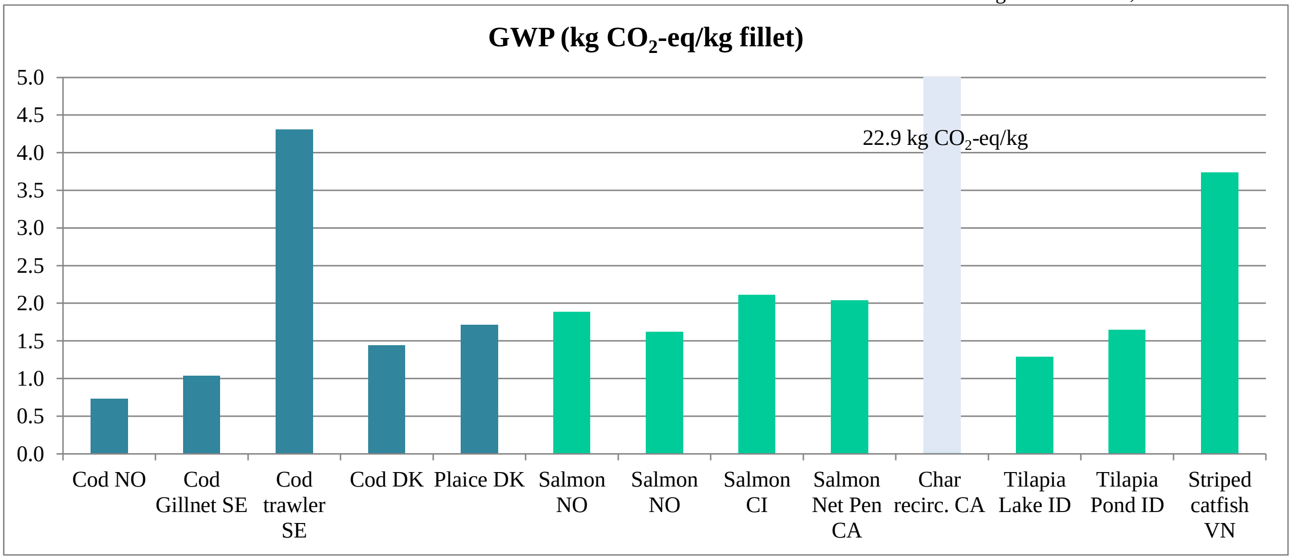


Figure 1. GWP of analysed systems (kg CO_2-eq/kg fillet). Abbreviations of country names: CA: Canada; CI: Chile; DK: Denmark; ID: Indonesia; NO: Norway; SE: Sweden; VN: Vietnam

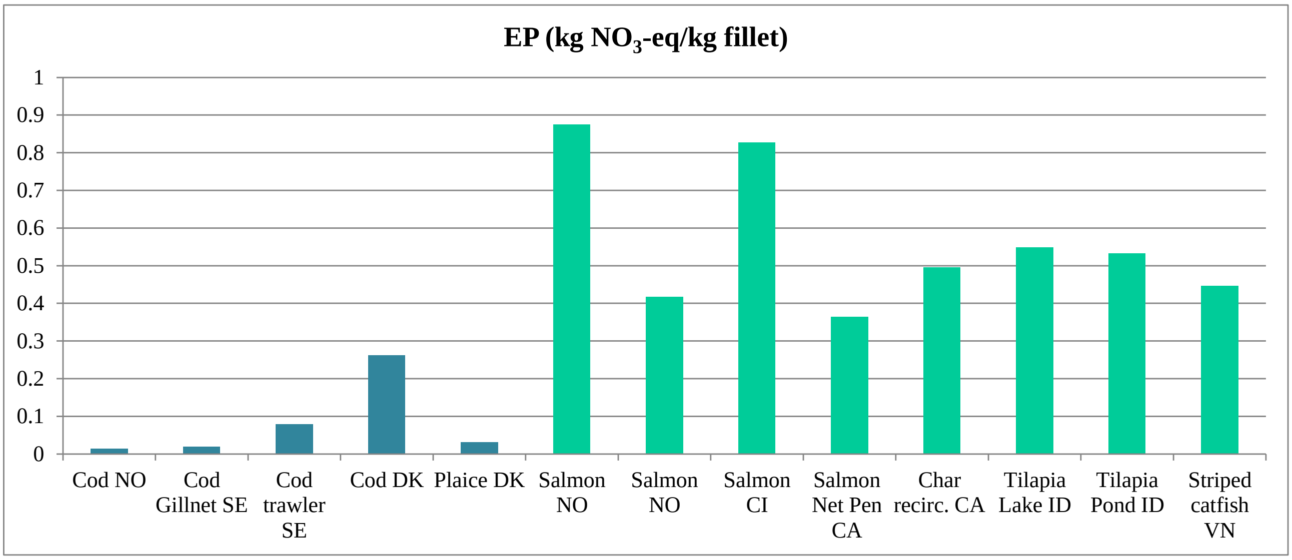


Figure 2. Eutrophication potential of analysed systems (kg NO_3-eq/kg fillet).

30. Environmental consequences of genetic improvement by selective breeding: a gilthead sea bream aquaculture case study

Joël Aubin[1,2,*], Ivone Acosta Alba[1,2], Pierrick Haffray[3], Jérôme Bugeon[4], Marc Vandeputte[5,6]

[1] INRA, UMR1069 Sol Agro et hydrosystème Spatialisation, F-35000 Rennes, France, [2] Agrocampus Ouest, F-35000 Rennes, France, [3] SYSAAF, Campus de Beaulieu, F-35000 Rennes, France, [4] INRA, LPGP, UR1037, Campus de Beaulieu, F-35000 Rennes, France, [5] INRA, UMR1313 GABI, F-78352, Jouy-en-Josas, France, [6] Ifremer, UMR110 Intrepid, F-34250 Palavas-les-Flots, France, [*] Corresponding author. E-mail: *joel.aubin@rennes.inra.fr*

Livestock genetic improvement is based on selective breeding of quantitative traits like growth or production yields. In fish farming, selective breeding has a high potential due to the recent domestication of the species. Genetic improvement (i.e. shortened duration of the production cycle or increased marketable size due to selection on body weight) could indirectly impact the use of production means and the valorisation of inputs. Consequently, the environmental performances of the production system can be modified. In order to explore the environmental consequences of genetic improvement in animal production and aquaculture, Life Cycle Assessment (LCA) was used to assess a sea bream (*Sparus aurata*) production system with expected genetic gain on growth or fillet yield.

Five scenarios were built using the results of expected selection response: initial unselected control, and 5 or 10% selection pressure on growth or fillet yield during 5 generations (15 to 20 years). The system boundary included the production of inputs such as feeds, fingerlings, production infrastructures, farm running, the commercial stage, purchase, the household cooking and consumption stage and the waste management. An attributional LCA was applied on the different scenarios, based on CML 2 (2000) and Cumulative Energy Demand methods, implemented in SimaPro 7.2 software, and the use of original data, and ecoinvent database. The calculated impact categories were climate change, eutrophication, acidification, cumulative energy demand, and net primary production use (Aubin et al., 2009). The functional unit was 1 kg of edible flesh.

For all the impact categories, the step of fish production (including the upstream processes) contributed for more than 80% of the impacts.

Due to the high influence of artificial feed production on the LCA results and to the hypothesis of lack of genetic correlation between the traits selected for and feed efficiency, selection on growth only had a limited effect on the different impact categories (less than 2%). This limited improvement was essentially due to the marginal improvement in the use of the infrastructure and the related energy use.

Selective breeding on fillet yield decreased environmental impacts from 10% to 22%. This decrease, calculated per kg of edible flesh can be explained by the positive genetic correlations between growth and fillet yield and by the decrease of waste at the household cooking and consumption stage.

This study is the first application of LCA as an assessment method of the environmental relevance of selective breeding in animal productions. It supports the lack of adverse effect of selection for the targeted traits (growth, or fillet yield) on the environment. This work is also a first step to introduce environmental goals into genetic selection schemes.

References

Aubin, J., Papatryphon, E., van der Werf, H. M. G., Chatzifotis, S., 2009. Assessment of the environmental impact of carnivorous finfish production systems using life cycle assessment. Journal of Cleaner Production 17, 354-361.

31. Environmental assessment of trout production in France: influence of practices and production goals

Joël Aubin[1,2,*], Elisabeth Samson[1,2,3], Aurélien Tocqueville[4]

[1] INRA, UMR1069 Sol Agro et hydrosystème Spatialisation, F-35000 Rennes, France, [2] AGROCAMPUS OUEST, F-35000 Rennes, France, [3] INRA, UMR1032 Structures et Marchés Agricoles, Ressources et Territoires, F-35000 Rennes, France, [4] ITAVI, F-75008 Paris, France, [*] Corresponding author. E-mail: *joel.aubin@rennes.inra.fr*

Trout production is the main fish farming activity in France, producing 37 000 tonnes of fish in 456 companies (MAAP, 2011). It is mainly based on rainbow trout (*Oncorhynchus mykiss*), reared in flow through systems directly connected to rivers. The production level of trout declined of 20% for the 10 past years, which clearly raises the question of the sustainable development of this activity. The variable market price of fish, the increasing costs of feed ingredients and the higher pressure of environmental regulations threaten the durability of the activity. Life Cycle Assessment was chosen as a method to check the environmental sustainability of fish farming, combining efficiency measurement and multicriteria characteristics, in order to draw new perspectives for trout farming. In 2007, a survey was conducted on 20 farms throughout France, and covering all types of production. We classified the trout farms in two groups depending on the commercial size of fish produced: the pan-size-trout farms (11 farms) and the large trout farms (9 farms). The attributional LCA was conducted from cradle to farm gate, using one tonne of fish as the functional unit. Environmental impacts (eutrophication, acidification, climate change, net primary production use, land occupation, water dependence) were calculated using CML 2 (2000) method (Aubin et al.2009), and cumulative energy demand was added. On farm working time was recorded. Primary data were mainly used for feed ingredients and processing, and fish farm running. The secondary data stem mainly from ecoinvent database. The comparison of the impact categories between the two groups of farms didn't show any significant differences, unlike previous studies results (Papatryphon et al., 2004). Net primary production use, water dependence and working time showed a tendency of a higher level in pan-size-trout farms. These impact categories values indicate a lower level of management and a lower efficiency of the production system. This remark is consistent with the high level of variability of the impacts in this group of farm. On the opposite, the large trout farms group had low levels of impact variability, which can be explained by a more standardised production and a higher level of technical competence. Despite the high contribution of feed production to the impacts (climate change, acidification, and cumulative energy demand) the correlation between food conversion ratio and these impacts was not significant in the group of pan-size-trout farms (unlike in large trout farms). This is probably due to the multiple factors determining the farms efficiency. These results show how the different production objectives (direct consumption and restocking for pan-size-trout farms, and filet production for large trout farms) influence the environmental impact variability of a production activity. This impacts variability indicates a higher level of system improvement in pan size trout farms than in large trout farms.

References

Aubin, J., Papatryphon, E., van der Werf, H. M. G., Chatzifotis, S., 2009. Assessment of the environmental impact of carnivorous finfish production systems using life cycle assessment. Journal of Cleaner Production. 17 (3), 354-361.

MAAP (Ministère de l'Agriculture de l'Alimentation et de la Pêche), 2011. Recensements 2008 de la salmonicultureet de la pisciculture marine et des élevages d'esturgeons, Agreste Les Dossiers, no. 11, April 2011, Paris,148p.

Papatryphon, E., Petit, J., van der Werf, H. M. G., 2004. The development of Life Cycle Assessment for the evaluation of rainbow trout farming in France. In: Halberg N., editor. Life Cycle Assessment in the Agri-food sector. Proceedings from the 4th International Conference. Danish Institute of Agricultural Sciences, Horsens, Denmark, 73-80.

32. LCA of locally produced feeds for Peruvian aquaculture

Angel Avadí[1,*], Pierre Fréon[2], Jesús Núñez[3], Camilo Cuba[4]

[1] UMR 212 EME, Institut de Recherche pour le Développement (IRD), Université Montpellier II, CRH, BP 171, 34203, Sète, France, [2] UMR 212 EME, IRD, Centre de Recherche Halieutique Méditerranéenne et Tropicale (CRH), BP 171, 34203, Sète, France, [3] UMR 5554 ISEM, IRD, Institut des Sciences de l'Evolution Montpellier , CS 19519, 34960 Montpellier cedex 2, France, [4] Universidad N. Federico Villarreal, Facultad de Oceanografía, Pesquería y Ciencias Alimentarias, 350 calle Roma, Lima, Peru, [*] Corresponding author. E-mail: *angel.avadi@ird.fr*

The research presents and discusses a LCA performed on an aquafeed plants in the Iquitos area, Peru (inventories were collected for two plants but results are presented for the larger one only). The main goal of the analysis is to explore local utilisation of Peruvian anchoveta (*Engraulis ringens*) fishmeal in aquafeeds for both omnivorous and carnivorous cultured species. These results can be useful for the comparative study of seafood supply chains (e.g. Avadí & Fréon, this Conference).

Results from other Anchoveta-SC work on two fishmeal plants (Fréon et al., in prep.) and the Peruvian industrial anchoveta fleet contributed preliminary downstream data for the fishmeal (fishing and reduction stages). One fishmeal plant was fully modelled and one "average" 395 m3 holding capacity fishing vessel (the most representative category of anchoveta-targeting vessels, regarding historical landings) was modelled including rough estimations of the construction and end-of-life phases. Data for other feed ingredients were taken from ecoinvent, and adapted when necessary to fit the sourcing of Amazonian aquafeed ingredients and energy sources (i.e. the grid-disconnected Iquitos electricity supply is based on thermal oil-powered generation). Weighted average of different feed formulations produced was utilised for determining the "typical" feed composition for *Colossoma macropomum* (mostly herbivore, the third more cultured freshwater species in Peru) and Brycon melanopterus (omnivore), two important Amazonian species provided by the Iquitos aquafeed industry. This scenario was compared with a theoretical feed plant catering to carnivorous fish (rainbow trout, *Oncorhynchus mykiss*) cultured in the Puno region, by adjusting fishmeal and fish oil use and regional energy mix (Fig. 1 a,b). Moreover, Peruvian formulations were compared with northern hemisphere formulations. Life cycle impact assessments were performed with the ReCiPe method, but additional impact categories were calculated: Cumulative Energy Demand and Biotic Resource Use as appropriation of primary production. Sensitivity analysis was carried out by exploring and contrasting various sources/proportions of key feed ingredients.

As expected, most of the environmental impacts during the life span of the plants are due to the provision of feed ingredients (>65%), especially fishmeal (>35% for trout), corn, wheat and soy meals. An allocation strategy for fishmeal and fish oil is in preparation, but a preliminary gross energy content criterion (71:29) was used for preliminary results. The oil-based Iquitos grid energy determines a high contribution of electricity used (~268 kWh/t feed) to several impact categories. Soy meal used in Peru is mainly imported from Bolivia, and thus a Bolivian soy meal was adapted from an existing ecoinvent Brazilian soy meal by adjusting the extent of the natural land transformation impact (provision of stubbed land, based on the characteristics of expansion zones for soy production in both countries (Dros, 2004)). Seasonal flood system rice grown in Iquitos was adapted from US rice, by reducing chemical input and eliminating irrigation. Proxies for wheat and corn where used (US produce), but their important contribution to certain impact categories suggests a full adaptation to local conditions is needed.

It was observed, in line with previous LCA studies of seafood systems, that construction and maintenance of feed plants contributes negligibly to environmental impacts of aquafeed products. It was also demonstrated that increasing use of fishmeal and fish oil, as well as the source of agricultural inputs, contribute to important variations in certain environmental impacts. Comparison between similar feeds for carnivore fish from Peru and Northern countries showed the specificity of South American ingredients: roughly comparable performance of fishmeal and soy meal (except Brazilian). The sourcing of feed ingredients was found to be critical for the contribution of feeds to impacts.

References

Avadí, A. and Fréon, P. 2012. A framework for sustainability comparison of seafood supply chains. This Conference.

Fréon, P. Durand, H., Huaranca, S., Avadi, A. 2012. LCA of two Peruvian fishmeal plants. In preparation.

Dros, J. M. 2004. Managing the Soy Boom: Two scenarios of soy production expansion in South America. WWF Forest Conversion Initiative. AIDEnvironment. Amsterdam.

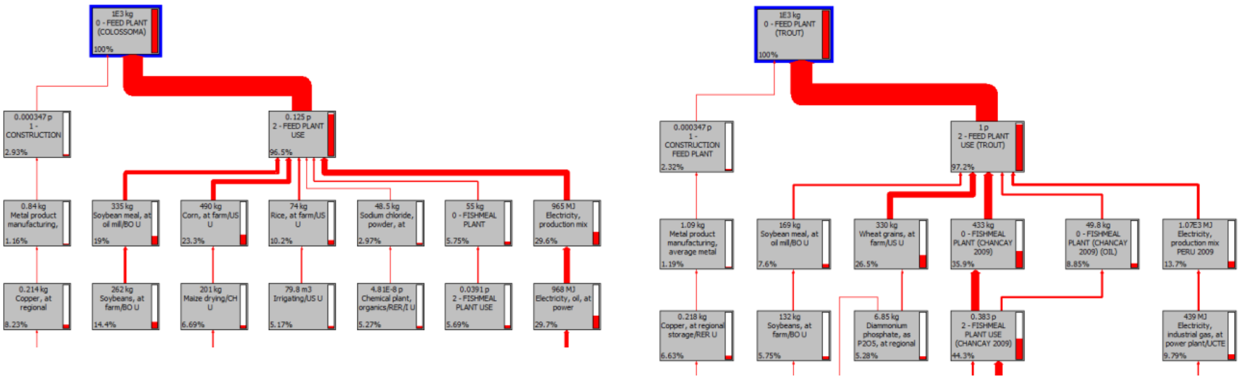


Figure 1. SimaPro-generated network view of (a) the studied Iquitos aquafeed plant (for Colossoma) and (b) the hypothetical Puno feed plant (for trout).

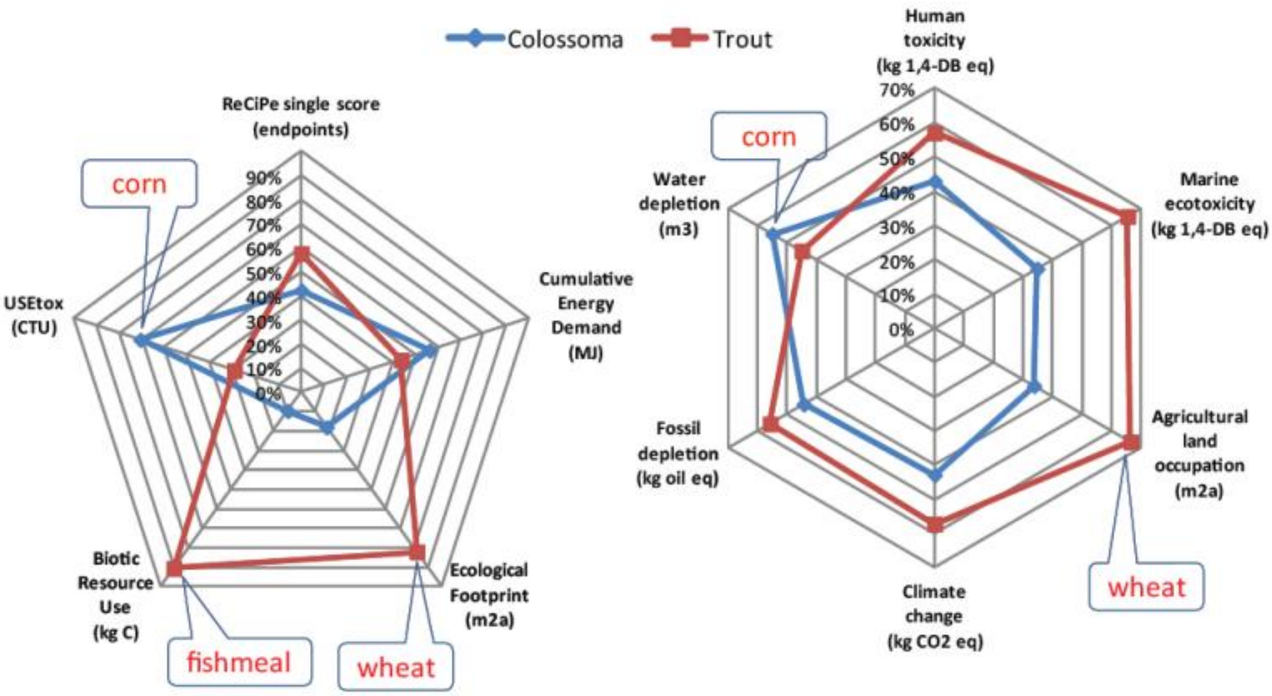


Figure 2. Comparison of two feed compositions (selected ReCiPe categories, additional categories). ReCiPe endpoints single score favours the Colossoma feed plant.

33. A framework for sustainability comparison of seafood supply chains

Angel Avadí[1,*], Pierre Fréon[2]

[1] UMR 212 EME, Institut de Recherche pour le Développement (IRD), Université Montpellier II, CRH, BP 171, 34203, Sète, France, [2] UMR 212 EME, IRD, Centre de Recherche Halieutique Méditerranéenne et Tropicale (CRH), BP 171, 34203, Sète, France, [*] Corresponding author. E-mail: *angel.avadi@ird.fr*

The research, carried out in the context of the ANCHOVETA-SC project (http://anchoveta-sc.wikispaces.com), proposes a modelling and sustainability assessment framework for comparing competing seafood supply chains based on common inputs, in terms of a selected indicators focusing on environmental and socio-economic performance, with emphasis on environmental and energy performance. The approach allows for comparison of alternative stock exploitation scenarios and transformation strategies.

Competing seafood supply chains are modelled in terms of their material and energy flows and, from a life cycle perspective, several tools and approaches are combined and applied to assess their sustainability. LCA is used for environmental impact assessment (including seafood-specific impact categories), but further analyses on energy efficiency and nutritional value of feed ingredients and seafood co-products are carried out. Basic socio-economic data is also compiled and used for estimating socio-economic performance by means of key indicators. A trophic model of the exploited marine ecosystem sourcing the studied systems is integrated within the supply chain model in order to capture the ecosystem-fishery interactions. The resulting supply chain model is implemented in a material flow modelling tool. This framework thus extends the ecosystem/supply chain coupled model proposed by Christensen et al. (2011) by accounting for biophysical flows. To test the framework, scenario comparisons of supply chains based on Peruvian anchoveta (*Engraulis ringens*) fishmeal are carried out. Scenarios are generated by varying the fishing stage, in terms of catches volumes of anchoveta (and its predators) and the intended "fate" of landings (direct or indirect human consumption, served by different fleets and delivering different final products/species to consumers).

The material flow modelling tool used proves useful for representation of interdependent industrial processes (i.e. supply chains) as Petri nets (Fig. 1) and Sankey diagrams, as well as providing the programming environment required to code the required material, energy and monetary flows logic. LCA and other life cycle tools provided the data and methods for extending the material and energy flows analysis towards sustainability assessment, in such a way that basic mass and energy flows are complemented with specific biophysical flows (e.g. energy/nutritional value of substances). Indicators used (Fig. 2; Table 1), scaled respect to a functional unit, are based on a) energy use: Cumulative Energy Demand; b) energy efficiency: edible protein EROI (Tyedmers 2000); c) seafood-specific impact categories: Biotic Resource Use approximated from the primary production appropriation of feed ingredients, especially fish-derived (Pauly & Christensen, 1995) and a related ecological footprint; and d) nutritional value for the consumer: protein/lipid content of raw materials and co-products. Socio-economic factors considered are limited to the most accessible ones (employment, added value). Alternative exploitation scenarios along the whole ecosystem-supply chain are obtained by coupling models: outputs from a published trophic model (Ecopath with Ecosim) of the Humboldt Current ecosystem are used as inputs of a material flow analysis model (Umberto). The coupled model, indicators and communication devices are currently in progress.

Conclusions derived from the model will be suitable for policy recommendations regarding exploitation volumes and seafood processing strategies, aiming at different interests: ecosystem health, human nutrition, food security, energy efficiency, reduction of environmental impacts and rent redistribution.

References

Christensen, V., Steenbeek, J., & Failler, P., 2011. A combined ecosystem and value chain modeling approach for evaluating societal cost and benefit of fishing. Ecological Modelling, 222(3), 857-864.

Pauly, D., & Christensen, V., 1995. Primary production required to sustain global fisheries. Nature, 374(6519), 255-257.

Tyedmers, P., 2000. Salmon and Sustainability: The biophysical cost of producing salmon through the commercial salmon fishery and the intensive salmon culture industry. Ph.D. Dissertation. The University of British Columbia.

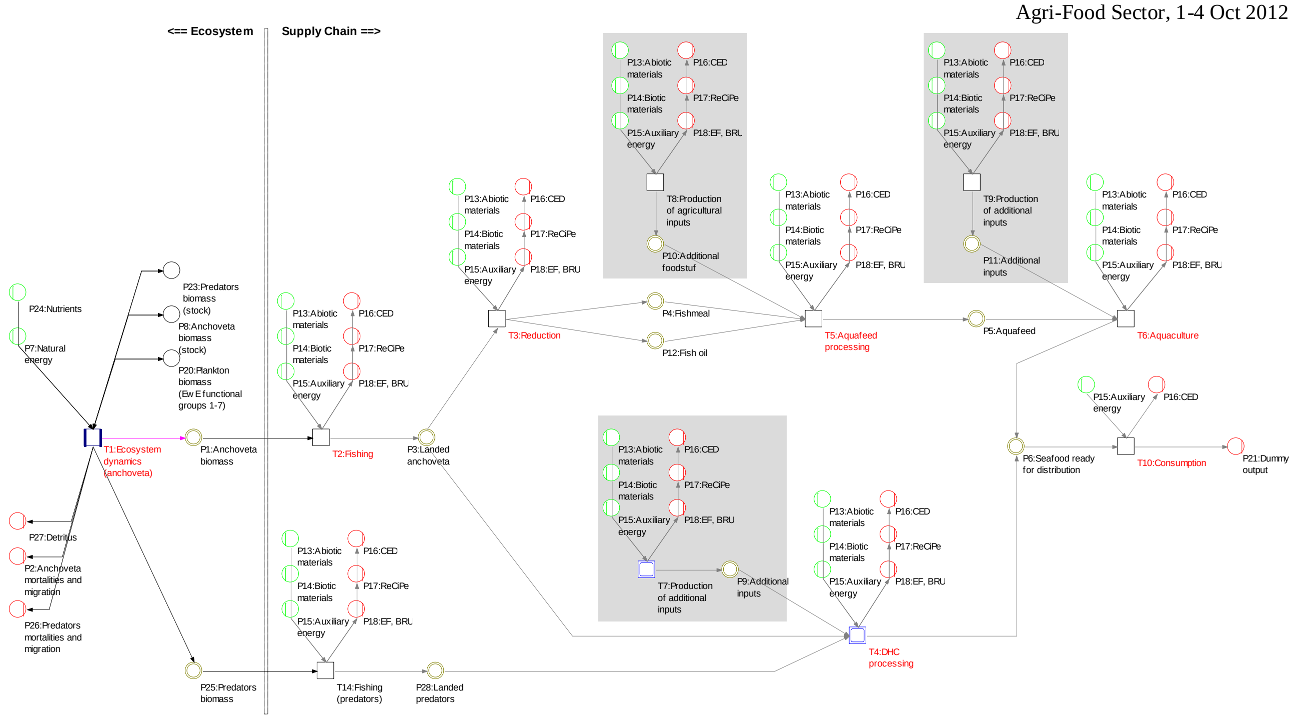


Figure 1. Proposed Petri net of a Peruvian anchoveta-based supply chains model.

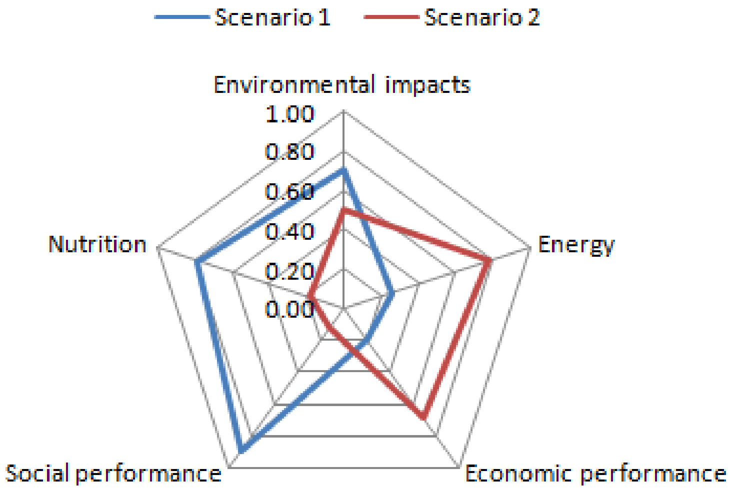


Figure 2. Proposed representation for sustainability scores of selected seafood supply chains based on Peruvian anchoveta (dummy values and scenarios).

Table 1. Key indicators calculated.

Indicators➔ Comparison objects🡻	Gross Energy Content (MJ/kg) Based on fat & protein	Edible protein EROI (%)	Biotic Resource Use (g C/kg) Ecological Footprint (ha/t)	**LCA** Cumulative Energy Demand (MJ)	Socio-economic indicators
Feed ingredients	X		X	X	
Aquafeeds	X		X	X	
Seafood products		X	X	X	X
Supply chains / scenarios				X	X

34. LCA of Finnish rainbow trout, results and significance on different allocation methods

Frans Silvenius[1,*], Juha Grönroos[2], Hanna Hartikainen[1], Sirpa Kurppa[1], Markus Kankainen[3], Timo Mäkinen[3], Raija Tahvonen[1], Jouni Vielma[3]

[1] MTT Agrifood Research Finland, Responsible Food Chain – Better Consumer Well-being, FI 00790 Helsinki, Finland [2] Finnish Environment Institute, P. O. Box 140, FI-00251 Helsinki, Finland [3] Finnish Game and Fisheries Research Insitute, P.O. Box 2, FI-00791 Helsinki, Finland, [*] Corresponding author. E-mail: *frans.silvenius@mtt.fi*

An LCA of Finnish rainbow trout was conducted in 2010-2012 by MTT Agrifood Research Finland, Finnish Game and Research Insitute and Finnish Environment Institute. The functional unit of the study was one ton rainbow trout fillet at the gate of the fish processing plant. The system boundaries included product chain of feed raw materials, feed production, hatchery, fish farming plants, fish processing, packages and transports. The impact classes studied were climate change, acidification, eutrophication, tropospheric ozone formation and primary energy use.
One important part of the study was to make comparison to the previous LCA study of rainbow trout (Seppälä et al. 2001, Silvenius and Grönroos 2003, Grönroos et al. 2006). Because of the development of feed and more effective feeding of the fish, the environmental impacts of the rainbow trout product chain have reduced over 20%. The carbon footprint of the rainbow trout in this study was about 4300 kg CO_2/t of fillet (Fig. 1)and eutrophication impact 38 kgPO_4-eq/t of fillet.
Allocation is one of the most biggest challenges related to LCA-investigations. In our study, the effects of different allocation methods to the results was studied. There have been some earlier studies concerning effects of the allocation on the results of fish products LCAs (e.g. Winther et al. 2009, Svanes et al. 2011). In our study, allocations based on different existing LCA standards were compared (Table 1). The allocation situations studied were allocation between captured fish species used as raw materials for the fish meal and oil, allocation between fish meal and oil, allocation between soy meal and oil, rapeseed meal and oil and allocation between fillet and by-products of slaughtering and filleting. Additionally, scenarios for the possible utilisation of the by-products were made.
The chosen allocation method had high effects on the final results. The mass allocation between fish fillet and by-products of the slaughtering and filleting resulted in lowest figures for environmental impacts. However, when using mass allocation for fish meal and oil raw material fishing, and mass allocation between soy meal and soy oil the estimated environmental impacts were highest.

References

Grönroos, J., Seppälä, J., Silvenius, F. & Mäkinen T., 2006. Life-cycle assessment of Finnish cultivated rainbow trout. Boreal Environment Research 11:401-404.

Hartikainen, H. 2011. Allokointimenettelyt elintarvikkeiden elinkaariarvioinneissa. Helsingin yliopisto, Taloustieteen laitos, Ympäristöekonomia, Pro gradu–tutkielma, Marraskuu 2011.

Seppälä, J., Silvenius, F., Grönroos, J., Mäkinen, T., Silvo, K. & Storhammar, E. 2001. Kirjolohen tuotanto ja ympäristö. Suomen ympäristö 529. 164 pp.

Silvenius, F. & Grönroos, J. 2003 Fish farming and the environment. Results of inventory analysis. Suomen ympäristökeskuksen moniste 276, 71 pp.

Svanes, E., Vold, M & Hanssen, O., J. 2011. Effect of different allocation methods on LCA results of products from wild-caught fish and on the use of such results. Int J Life Cycle Assess 16:512-521.

Winther, U., Ziegler, F., Hognes, E., Emanuelsson, A., Sund, V. & Ellingsen, H. 2009. Carbon footprint and energy use of Norwegian seafood products. SINTEF Fisheries and Aquaculture, Norway.

Table 1. The carbon footprint of rainbow trout fillet calculated by different standards (Hartikainen 2011).

Standard	**Carbon footprint**
ILCD Handbook	4.2
ISO14040	4.2-4.4
GHG Protocol	4.1-4.3
PAS 2050	4.2
PCR Basic Module (2010a)	2.4-4.4

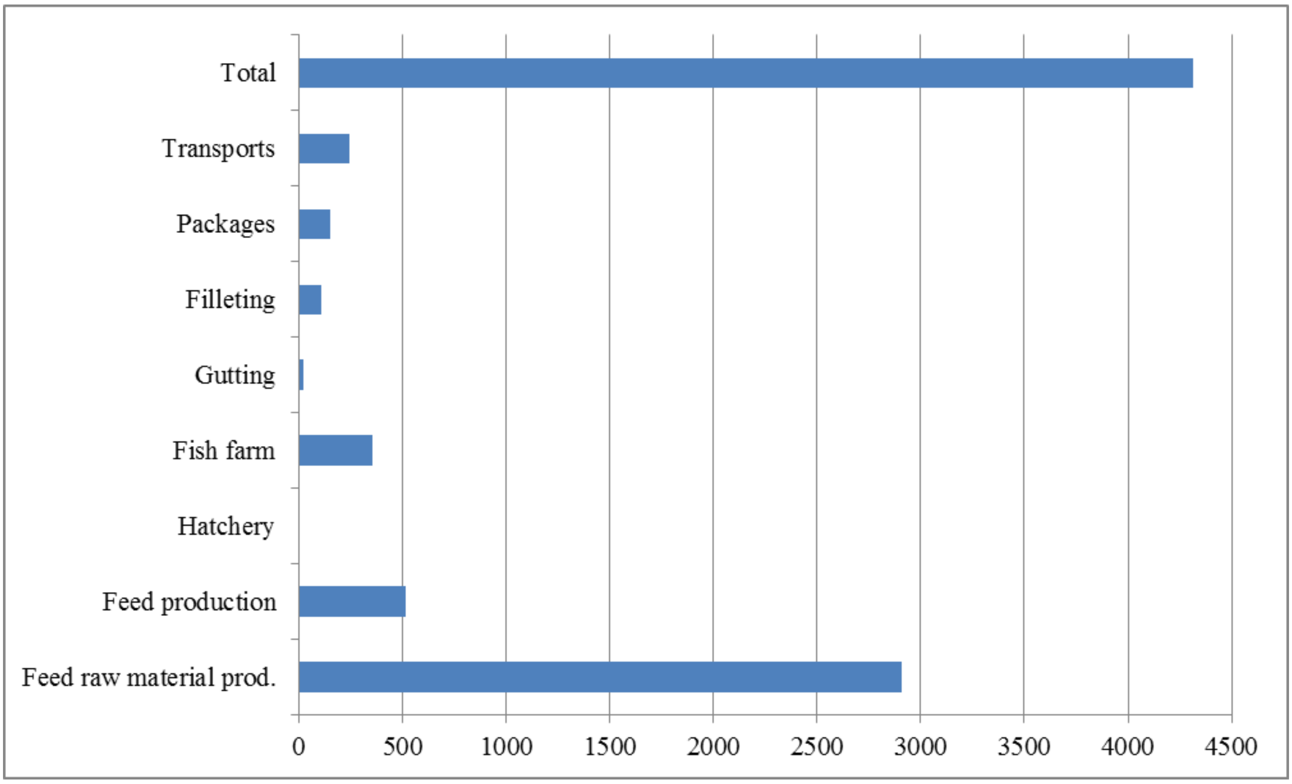


Figure 1. Carbon footprint of rainbow trout fillet divided into process stages.

35. Mainstreaming Life Cycle Management in the seafood sector: using a sector based and regional approach in Northern France

Catherine Beutin[1,*], Jodie Bricout[2], Naeem Adibi[2], Christelle Demaretz[3]

[1] Aquimer competitivity cluster 16 rue du Commandant Charcot - 62200 Boulogne-sur-Mer, France, [2] cd2e – Plateforme [avniR], Rue de Bourgogne, Base du 11/19 62750 Loos en Gohelle, France, [3] Direction de l'environnement du Conseil Régional Nord-Pas de Calais, 141 rue Berthe Morisot, Lille, France, [*]Corresponding author. E-mail: *catherine.beutin@poleaquimer.com*

Whilst Life Cycle Management (LCM) is becoming commonplace in larger corporations, or forward thinking governments, it is far from mainstream. To achieve sustainable production and consumption patterns, LCM needs to be taken up by whole supply chains that include many small and medium enterprises. These businesses typically lack the financial capacity or human resources to implement LC tools on their own, and are wary of working with support organisations outside of their sector or local area.

The French competitively cluster Aquimer is piloting an innovative study to develop a strategic action plan to integrate Life Cycle Approaches (including eco-design and product environmental labelling) into businesses, education and research organisations in the seafood sector. Supported by the Nord Pas de Calais Regional Council and the ADEME (French Environment and Energy Management Agency), the study for the seafood sector is being undertaken in parallel with the textile, packaging and mechanical sectors.

The general process consists of four steps: benchmark, sector maturity assessment, needs identification, action plan development and Implementation.

The benchmark identifies life cycle based initiatives and tools relevant to seafood sector, focussing on, but not limited to North West Europe. The maturity of businesses, education bodies and research centres in the region with in relation to LCM practices is undertaken via interviews with key stakeholders. Stakeholder engagement is a key aspect of the needs identification and action plan development phases, not only to ensure that the proposed LCM action plan "fits" the needs of the sector, but to create ownership for the implementation phase.

The paper will also explain how we develop an inventory and a multi criteria matrix to measure the environmental maturity level of the seafood industry considering a life cycle vision in Nord Pas de Calais based on two main criteria: Maturity and willingness of key actors for life cycle assessment and eco-design.

This innovative approach to mainstreaming LCM leverages sectorial and regional networks to help overcome barriers to implementation. From a business perspective, integration with existing professional organisations means that SMEs access advice and tools through organisations that they already know and trust.

References

UNEP/SETAC. 2009. Life Cycle Management: How business uses it to decrease footprint, create opportunities and make value chains more sustainable.

European Commission. 2003. COM Integrated Product Policy Building on Environmental Life-Cycle Thinking

Boeglin. 2011. L'expérimentation nationale de l'affichage des caractéristiques environnementales in Proceedings LCA Conference 2011, 3-4 November Lille, France

36. Life cycle assessment of integrated fish pond aquaculture of household farms in sub-Saharan Africa

Thomas Efole Ewoukem[5,*], Joël Aubin[1,2], Olivier Mikolasek[4], Michael S. Corson[1,2], Hayo van der Werf[1,2], Dominique Ombredane[2,3]

[1]INRA, UMR1069, Soil Agro and hydroSystem, F-35000 Rennes, France, [2] Agrocampus Ouest, F-35000, Rennes, France, [3] INRA, UMR0985, Ecology and Ecosystem Health, F-35000 Rennes, France, [4] CIRAD F-34398 Montpellier, France, [5] University of Dschang P.O.Box: 222 Dschang, Cameroon, * Corresponding author. E-mail: *efole_thomas@yahoo.fr*

Life Cycle Assessment (LCA) of integrated pond aquaculture of household farms in sub-Saharan Africa was carried out from 2008-2010 to identify ways to improve the systems and better understand the contribution of fish-pond systems to regional sustainable development.

A total of 20 polyculture fish ponds corresponding to three types of integrated fishpond system (semi-intensive medium or small; extensive) were chosen after screening household farms with aquacultural activities. Thus, inland fish-farm production in two contrasting areas of Cameroun was analysed. The fish-production system were integrated in the farms with either pig- or crop-production systems. In all farms, fish production was a polyculture based on tilapia (*Oreochromis niloticus)* or a tilapia-catfish (*Clarias gariepinus*) association. Secondary species often were included depending on the type of fish farm. For example, Hemichromis fasciatus or the snakehead fish (Parachana obscura) could be included to control tilapia population size, while common carp (Cyprinus carpio) or Heterotis niloticus could be included (according to availability) due to their bottom-feeding activity.

LCA was conducted according to the CML2001 method adapted to aquaculture production using economic allocation (taking manure into account), one tonne of fish as the functional unit, and background data from Ecoivent. The system boundary included on-farm processes, production of feed and fertiliser, fingerlings, and transportation at all stages. Dynamics of nitrogen and phosphorus were evaluated using a nutrient mass-balance modelling approach.

With the exception of extensive systems, there was large variability in the magnitude of impacts within each type of system. The main processes contributing to impacts were fish production, manure fertilisation, and fry production. Environmental impacts of the semi-intensive systems analysed were high compared to similar aquatic systems in Brazil. Their low efficiency and the source and origin of their inputs determined the magnitude of impacts. A water-purification role of extensive systems was also observed. On the other hand, the amount of human labour, which plays a strong social role sub-Saharan Africa, was higher in all semi-intensive systems.

37. Distribution of Norwegian fresh fish fillets from an environmental and economic point of view

Mie Vold*, Ole Jørgen Hanssen, Erik Svanes

Ostfold Resarch, Fredrikstad, Norway, * Corresponding author. E-mail: *mie@ostfoldforskning.no*

Optimisation towards more sustainable distribution solutions for fish fillets are important topics in Norway. Several research projects have shown that it is important to consider the packaging system from a holistic point of view, where the whole value chain is included. More material intensive packaging systems might be more sustainable if this leads to less loss of product through the value chain. It is also important to consider how the effectiveness of packaging systems can be improved, through maximising the volume of product in relation to total pallet volume.

Sea food is one of the most important export products from Norway. During the past years relatively more fish have been distributed as filets, and it is a long term strategy to distribute as much as possible as filets. Packaging research in Norway has thus been focused on development of new solutions for sea food distribution. Through common projects financed by the Norwegian Research Council, new packaging solutions have been developed, tested and optimised. Important drivers for innovation have been to develop solutions that are less voluminous, less material intensive, more effective in the packing process, utilise transport capacity better and are easier to recycle after use and still preserve high quality and low product loss. In parallel, the traditional packaging solutions have also been improved, to meet competition from new solutions. For fresh products, time to market is a critical factor in distribution, and new solutions have also been developed to increase the shelf life of fresh seafood products.

This presentation will focus on experiences from industrial case studies of packaging systems and distribution of fish filets, with main focus on changes from conventional packing with ice to super freezing without ice and with new fibre based packaging. The presentation includes life cycle data for processes and handling, production of packaging materials, impacts on transport, converting to final packaging solutions and treatment of packaging waste resources (material, economic and energy data)

References

Rubach, S. & Hanssen, O.J. 2002. Nordic methods for documentation of packaging optimisation. Proposal for Nordic method – Part B. Benchmarking system, Østfoldforskning.

Hanssen, O.J. 1997. Sustainable product management. Dr.techn. thesis, Norwegian University of Science and Technology, Trondheim

Nyland, C.A., Rubach, S., Hanssen, O.J., 2006. Value Chain Analysis of Different Types of Distribution Packaging For Fresh Fish Fillets, Proceedings IAPRE Conference, Tokyo Oct. 2006.

Svanes E., Vold M. and Hanssen O.J., 2011. Environmental assessment of cod (Gadus morhua) from auto-line fisheries. International Journal of Life Cycle Assessment, 16: 611-624.

Svanes E., Vold M., Møller H., Kvalvåg Pettersen M, Larsen H. and Hanssen OJ, 2010. Sustainable Packaging Design: a holistic methodology for packaging design. Packaging Technology and Science.

Vold M, Hanssen O.J., Svanes E., 2006. Value Chain Analysis of Different Types of Distribution Packaging of Fresh Fish Fillets, IAPRI conference,

38. System delimitation in life cycle assessment of aquaculture

Biniam Samuel-Fitwi[1,2,*], Jan P. Schroeder[2], Carsten Schulz[1,2]

[1]Institute of Animal Breeding and Animal Husbandry, Christian-Albrechts-Universität zu Kiel, 24098 Kiel, Germany, [2]Gesellschaft fuer Marine Aquakultur (GMA) mbH, Hafentoern 3, 25761 Buesum, Germany, [*] Corresponding author. E-mail: *biniam@gma-buesum.de*

In the past two decades, life cycle assessment (LCA) is recognised as a standardised and structured method of evaluating the environmental impacts arising throughout the entire life cycle of a product, process or activity. However, methodological issues still exist, when allocating the environmental burden of a specific production system between products and co-products. According to the ISO standards, the first option is to avoid allocation by making use of a subdivision or to expand the systems investigated. Aquaculture system hardly applied system expansion whenever a multifunctional process has more than one functional flow. The objective of this study is to model the system expansion for aquaculture production. The different affected processes with co-products are selected for system expansion. Thus, in this study we have considered the system expansion in two different stages in the life cycle of the fish production: aquacultural stage, with case study of trout aquaculture; and feed manufacturing stage. Rainbow trout (*Onchorhynchus mykiss*) production was used as a case study to illustrate the method using different scenarios of system expansion. This article showed that system expansion best describes the environmental impact of systems affected by increase in demand of rainbow trout.

Table 1. Relative emission load of rainbow trout farming estimated following different scenarios of system expansion. The results are characterised results and are related to the functional unit: 1 kg of rainbow trout demanded in Germany.

Impact category	**Unit**	**Scenario I**	**Scenario II**	**Scenario III**	**Scenario IV**	**Scenario V**	**Scenario VI**
Acidification	kg SO_2 equiv.	0.0077	0.0083	0.0113	0.0113	0.0090	0.0086
Eutrophication	kg PO_4 equiv.	0.0600	0.0601	0.0604	0.0601	0.0599	0.0598
Global warming	kg CO_2 equiv.	0.7670	0.9000	2.3219	3.6333	2.5516	2.4500
Land competition	m²a	0.7422	0.7440	1.3030	1.0268	0.6013	0.5996

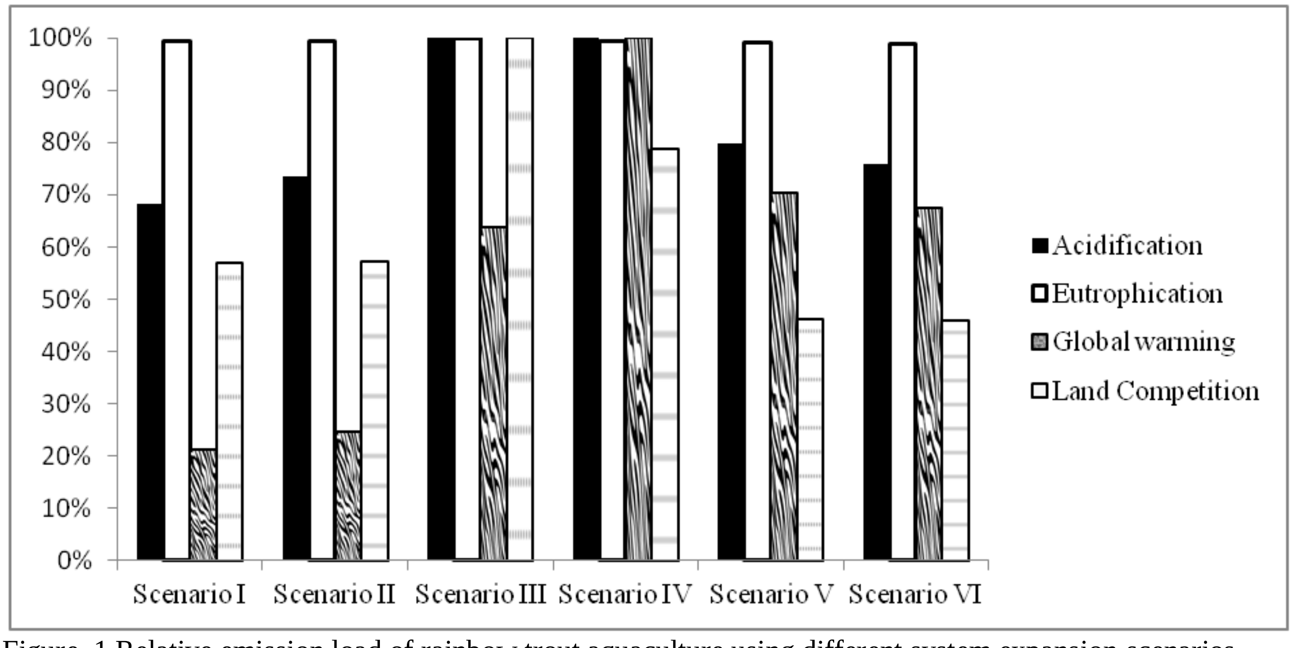


Figure. 1 Relative emission load of rainbow trout aquaculture using different system expansion scenarios.

39. The assessment of biodiversity within UK farming systems using an extended LCA ecosystem approach

Catherine Keeling, Rob Lillywhite*

School of Life Sciences, The University of Warwick, UK, * Corresponding author. E-mail: *robert.lillywhite@warwick.ac.uk*

In 2009, the UK Government commissioned a three-year project to develop a methodology to assess the economic, environmental and social characteristics of UK farming systems. Thirty-two farming systems were identified to represent the diversity of UK agriculture and forty indicators were described to cover a wide range of economic, environmental and social parameters. The methodology uses a development of the existing Cranfield LCA model (Williams et al., 2007) combined with newly developed economic and social matrices. A matrix was constructed to accommodate the data generated by the 1280 farming system/characteristic combinations. The matrix uses both actual and normalised values to investigate the advantages and disadvantages that accrue from different farming systems. This overall approach is described as an extended LCA ecosystem approach model. This paper reports on the inclusion of an aggregated biodiversity indicator into the methodology.

One major difficulty in representing biodiversity is the definition and subsequent calculation of single or multiple indicators. Single indicator species groups such as birds or carabids may be used, but effects of management practices have been shown to be specific to the different indicator groups (Jeanneret et al., 2008). Meta-analyses have investigated the dose-effect relationships between pressure factors and biodiversity in agricultural landscapes (Reidsma et al., 2006; Alkemade et al., 2009). Following on from this work, ecosystem quality values (defined as the mean species abundance relative to the undisturbed situation) were assigned to each combination of land-use type, intensity level, and type of management (organic and non-organic). Using data from the UK's Farm Business Survey, an estimation of the remaining ecosystem quality of each farming system could then be made.

Highest ecosystem quality values were associated with the grass-based, low intensity systems, whilst the lowest scores were obtained by the most intensively housed livestock systems. Within production systems, the indicator was capable of differentiating between different types of management. For example, within pig production, outdoor bred/indoor finished systems scored higher than permanently housed systems. This methodology can be used to examine how changes in the farming landscape would affect levels of biodiversity; whether through land-use change, such as the conversion of grassland to arable land, through intensification, or through conversion to organic production.

References

Alkemade, R., van Oorschot, M., Miles., L. Nellemann, C., Bakkenes, M., ten Brink, B., 2009. GLOBIO3: A framework to investigate options for reducing global terrestrial biodiversity loss. Ecosystems 12, 374-390.

Jeanneret, P., Baumgartner, D. U., Freiermuth Knuchel, R., Gaillard, G., 2008. A new LCIA method for assessing impacts of agricultural activities on biodiversity (SALCA-Biodiversity). 6th International Conference on Life Cycle Assessment in the Agri-Food Sector 2008, Proceedings, 34-39.

Reidsma, P., Tekelenburg, T., van den Berg., M., Alkemade, R., 2006. Impacts of land-use change on biodiversity: an assessment of agricultural biodiversity in the European Union. Agriculture, Ecosystems and Environment 114, 86-102.

Willams, A.G., Audsley, E., Sandars, D.L. 2006. Determining the environmental burdens ands resource use in the production of agricultural and horticultural commodities. Final report for Defra project IS0205. Cranfield University, Bedford and Defra, London.

40. Proposal of a unified biodiversity impact assessment method

Jan P Lindner[1,*], Tabea Beck[2], Sebastian Schwarz[3], Ulrike Eberle[4], Martin Blumberg[5]

[1] Fraunhofer IBP, dept. GaBi, Germany, [2] University of Stuttgart, LBP, dept. GaBi, Germany, [3] Five Winds International, [4] Corsus – Corporate Sustainability, Germany, [5] brands & values GmbH, [*] Corresponding author. E-mail: *jan-paul.lindner@ibp.fraunhofer.de*

Driven by the insight of society that biodiversity is worthy of protection, the decrease in habitats and species recently became a central topic in environmental policy. The European Union aims to study and to stop anthropogenic loss of biodiversity. In Germany this EU target led to the National Strategy on Biological Diversity (BMU 2007). Among others, one consensus point in environmental policy is that private and public actors need instruments to identify the "biodiversity performance" of product systems and services.

A team under the lead of the GaBi department at Fraunhofer Institute for Building Physics (IBP) launched a project in 2012 to develop a method for the assessment of the biodiversity impact of product systems. Regarding future application the crucial point is that this tool has to be broadly accepted by scientific and economic actors. Thus, knowledge and experience of the LCA community must be involved in the development of the method.

It is suggested that the assessment method includes a preliminary quick check to decide on the extent to which the full method should be applied (Fig. 1). Criteria may be land requirement or activities in biodiversity hot spots. The biodiversity impact assessment method fits in the general land use impact assessment framework described in Milà i Canals et al. (2007). There, the impact of a land using process is defined as:

$$\text{Impact} = \text{affected area} \times \text{duration of impact} \times \text{quality change of the area}$$

The biodiversity impact assessment method reflects the quality axis for the impact category "biodiversity". It is loosely based on the method proposed by Michelsen (2008) in that it employs region-specific characterisation models and allows aggregation across regions through weighting factors. Biodiversity is described by means of a multidimensional potential function (Fig. 2). Setting the parameters of the function sets the biodiversity value for a certain process at a certain place. Comparison to the biodiversity value under given reference conditions allows the calculation of the quality change for the equation above. The form and parameters of region-specific biodiversity potential functions are derived from literature research about the state of the regional ecosystem(s), national/regional conservation goals, as well as expert judgement. Weighting factors for the aggregation across regions e.g. in global supply chains (Fig. 1) are derived from globally agreeable descriptors for biodiversity and/or ecosystem quality.

The proposed method allows relatively precise and accurate biodiversity impact assessment at the price of relatively high data acquisition. In the unified method, the detailed procedure described above is used for the most relevant elements of a product system, as determined by the quick check. A broad-brush impact assessment method is applied to fill in the blanks for less relevant elements of the product system.

References

BMU, 2007. National Strategy on Biological Diversity. Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU), Public Relations Division, Berlin.

Michelsen, O., 2008. Assessment of Land Use Impacts on Biodiversity. Proposal of a new methodology exemplified with forestry operations in Norway. Int. J. LCA 13 (1), 22-31.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007. Key elements in a framework for land use impact assessment within LCA. Int. J. LCA 12 (1), 5-15.

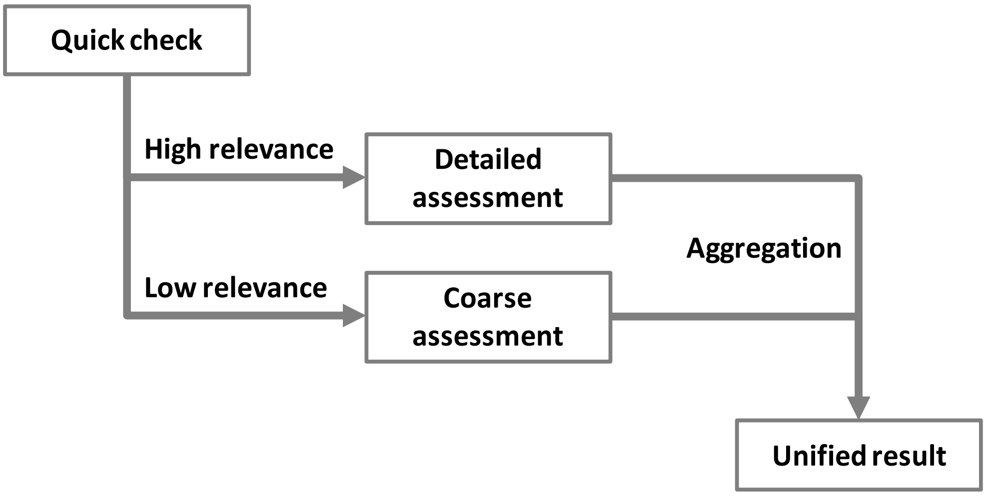


Figure 1. Methodology overview.

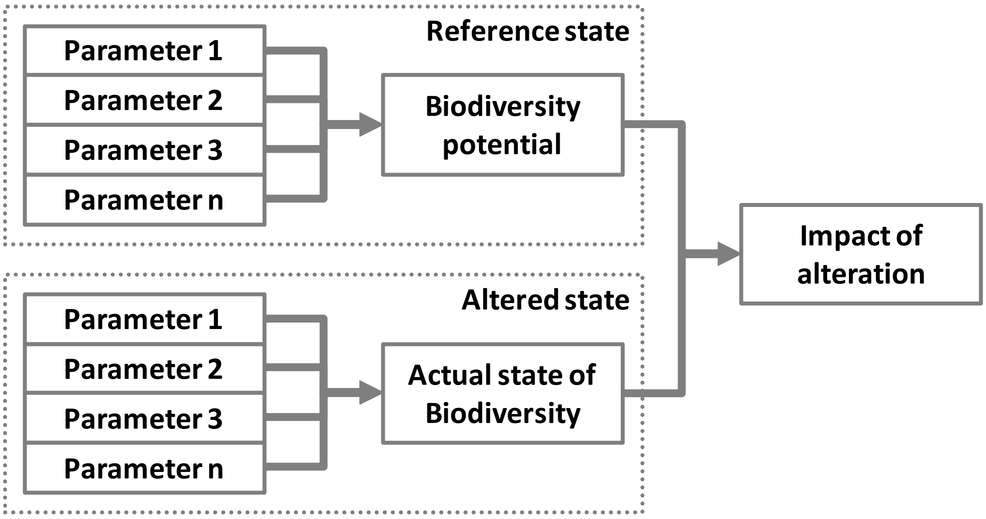


Figure 2. Region-specific impact model.

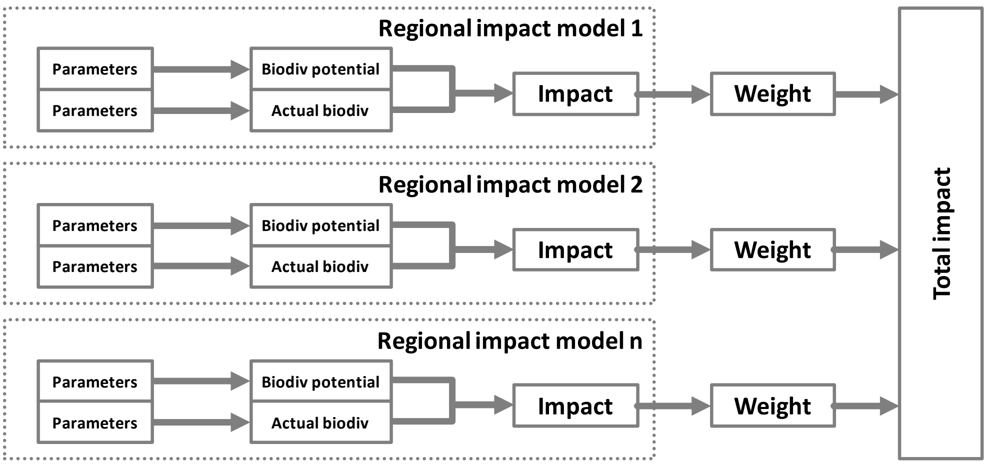


Figure 3. Aggregation of several regional impacts.

41. Displaying environmental footprints of agricultural products: a "biodiversity" indicator based on landscape features

Antonin Vergez*, Martin Bortzmeyer

Ministry of Ecology, Sustainable Development and Energy, CGDD, * Corresponding author. E-mail: *antonin.vergez@developpement-durable.gouv.fr*

Sustainable household consumption can be an important engine for a greener economic growth. Introducing information about the environmental impacts of products throughout their life cycle is one way to make the global consumption system more sustainable. In France, the "Grenelle" environmental laws introduce the right for consumers to have information related to the multi-criteria environmental footprint of (mass market) products at the point of sale. A year-long experiment began in July 2011 involving around 160 volunteer companies and other relevant economic stakeholders. Because the agricultural and food sectors are concerned, it is expected that providing consumers with this environmental information could have a positive effect and lead the whole food chain, from farmers to retailers, to produce and market more sustainable goods.

This legislative pillar of the French system relies on the technical ADEME AFNOR platform (expertise and standardisation) building methodological documents with all concerned stakeholders. In the food, beverage and pet-food sector, the transversal methodological document finalised in April 2012 by the platform recommends using 100 mg or 100 ml as a functional unit and addressing the following environmental impact indicators, units (and degrees of precision) and calculation methods: greenhouse gas emissions (in kg eqCO_2, using IPCC 2007), water consumption (litres, using net consumption), water eutrophication (in g EqN, using ReCiPe), water ecotoxicity (CTUe, using USEtox), and impacts on biodiversity (but do not recommend any unit or method for this issue: it has to be defined).

The French Ministry of Ecology, Sustainable Development and Energy (MEDDE CGDD) decided to finance a study to propose such a product-level biodiversity indicator because i) a multi-criteria environmental footprint is more representative, ii) stakeholders of the ADEME AFNOR platform have identified biodiversity as an environmental challenge to address, iii) agriculture's contribution to both loss and preservation or improvement of biodiversity is important in the world and in France, and iv) existing LCA methods to address biodiversity impacts of food and agricultural products are unsatisfactory: based on land "occupation" or "consumption", they associate higher impacts with more extensive farming practices. The poster presents the main characteristics of this study, which will end in autumn 2012.

A call for tenders specified the type of biodiversity indicator to build. It i) has to express not only loss or impacts but also a positive contribution to biodiversity; ii) could be specific to the farm level, since major biodiversity impacts of food products are concentrated in the agricultural production phase; iii) has to be calculated as a ratio dividing a quantity of biodiversity by a quantity of product; and iv) has to be consistent with biodiversity indicators already used in existing agricultural policies (CAP good agricultural and environmental conditions, CAP second pillar subsidies for grasslands and French farm environmental certification scheme). Two companies, Solagro (an agro-environmental consultancy firm) and ACTA (French agricultural technical institutes), were selected to perform.

Consequently, "landscape features", which are "semi-natural, unfarmed or extensively farmed environments", were the proxy chosen to characterise biodiversity. Examples in the database cover extensive permanent grasslands and temporary grasslands with low nitrogen inputs, hedges, ponds, ditches, trees in a line or in a group, isolated trees, field margins and terraces, land left fallow, buffer strips, etc.

The year-long study is divided into 3 phases: i) stabilise methodology and definitions and start data collection (3 months); ii) calculate values of the biodiversity indicator for the selected agricultural products (6 months); and iii) discuss the results and properties of the indicator (3 months). Selected animal products are cow milk, cow meat (dairy or beef cattle), sheep meat, sheep milk, lamb meat, pig meat, chicken meat, chicken eggs. Selected plant products are wheat, barley, maize, sunflower, rapeseed, potatoes, protein crops, orchards, vineyard, sugarbeets, grapes, apples, tomatoes, vegetables, and fodder (silage maize, permanent and temporary grasslands, fallows).

Three different approaches are used to calculate this indicator: i) national statistical databases (National Forestry Inventory 2nd cycle, TERUTI-Lucas, National Agricultural Survey 2000), ii) representative farms for typical production systems (database of agricultural institutes) and iii) the farm database DIALECTE (containing around 2000 extensive agricultural units). "Landscape features" are measured at different levels depending on the database, for example, at the commune level (in the national database) or at the farm level. Since “landscape features” are of very different natures (grasslands, hedges, trees, etc.), four types of weighting coefficients are used to convert them into “biodiversity-equivalent area”, an estimated real surface area for which all coefficients of different “landscape features” are equal to 1 (CAP and direct-aid eco-conditionality coefficients, developed-area coefficients and coefficients based on developed areas but that also consider agricultural practices).

Initial results show that it is possible to link an agricultural product with a quantity of biodiversity. Yet, much work is still required, for example, considering yield effects, allocation between co-products or biodiversity impacts linked to animal-feed production. Results will be verified and compared in phase 3, which will also consist of an analysis of the indicator's properties. Its main advantages are its existence (given the scarcity or absence of similar indicators), its practicality and feasibility (to calculate an indicator based on “landscape features” for several different agricultural products using several databases and entry levels), its consistency with other agricultural policy indicators, its ability to measure a positive contribution to biodiversity and its acceptability by farmers. Its main limits and drawbacks include that it is specific to the farm- and farming-production phase (it is thus not a life cycle indicator) and it is very recent (pioneering and opening a new international research agenda). We conclude by insisting on the urgent need to develop academic and operational research to consolidate this agricultural-product-level biodiversity indicator or to build a better one that contains the properties desired.

References

IPCC, 2007. Intergovernmental Panel on Climate Change.

MEDDE (French Ministry of Ecology, Sustainable Development and Energy), 2012a. Display of the environmental footprints of products: French developments. Online: http://www.developpement-durable.gouv.fr/Product-Environmental-Footprint.html

MEDDE (French Ministry of Ecology, Sustainable Development and Energy), 2012b. Expérimentation nationale de l'affichage environnemental. Online: http://www.developpement-durable.gouv.fr/-Experimentation-de-l-affichage,4303-.html

Solagro and ACTA, 2012a. Indicateur biodiversité pour l'affichage environnemental des produits agricoles. Rapport final de la phase 1 au MEDDE CGDD, February 2012, 82 p.

Solagro and ACTA, 2012b. Indicateur biodiversité pour l'affichage environnemental des produits agricoles. Rapport final de la phase 2 au MEDDE CGDD, June 2012, 107 p.

Solagro and ACTA, 2012c. Indicateur biodiversité pour l'affichage environnemental des produits agricoles. Rapport final de la phase 2 au MEDDE CGDD, October 2012.

42. Carbon footprint of the Chilean apple orchard system: study in the main southern hemisphere producer

Alfredo Iriarte[1,2,*], Pablo Villalobos[3], Pablo Yañez[3], Carlos Huenchuleo[3]

[1] Department of Industrial Management and Modelling. Faculty of Engineering. Universidad de Talca. Casilla 747, Talca, Chile, [2] Chilean Food Processing Research Centre (Centro de Estudios en Alimentos Procesados CEAP). R09I2001. Av. San Miguel km 2. Talca, Chile, [3]Department of Agricultural Economics. Faculty of Agricultural Sciences. Universidad de Talca. Casilla 747, Talca, Chile, [*] Corresponding author. E-mail: *airiarte@utalca.cl*

Chile is a major off-season fruit supplier and covers a significant portion of fresh fruit imports made by the United States, the European Union and Japan. Chile is the largest southern hemisphere producer and exporter of apple (CCM, 2010). Estimating carbon footprint of agricultural systems is becoming an important issue for country's horticulture sector. This study evaluated, using a life-cycle approach, the carbon footprint of the intensive apple orchard system in Chile. The methodology used is according to the PAS 2050 specifications (BSI, 2008). The system boundaries included all the life cycle stages from the cradle to the farm gate (harvested apples). The apple production analysed in this study corresponds to nationwide representative practices. The results indicated that mineral fertilisers caused the major contribution to the carbon footprint. In contrast, packaging waste had a minor influence. The application of the life-cycle approach helped to identify improvement measures to reduce greenhouse gas emissions of the orchard production system.

References

BSI, 2008. PAS 2050:2008 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution (BSI). London, UK.

CCM (Centro de Competividad del Maule), 2000. Manzanos. Universidad de Talca, Talca.

43. Assessment of carbon footprint methodologies used to evaluate Mediterranean horticultural production in standard multi-tunnel greenhouses

Rubén Beltrán-Palafox[1], Julia Martínez-Blanco[2,*], Joan Rieradevall[2,3], Assumpció Antón[4,5], Pere Muñoz[5]

[1]Joint European Master in Environmental Sciences (JEMES), Universitat Autònoma de Barcelona (UAB). 08193 Bellaterra Barcelona, Spain, [2]Institute of Environmental Science and Technology (ICTA), Universitat Autònoma de Barcelona (UAB). 08193 Bellaterra, Barcelona, Spain, [3]Chemical Engineering Department, Universitat Autònoma de Barcelona (UAB). 08193 Bellaterra, Barcelona, Spain, [4] Departament d'Enginyeria Química, Universitat Rovira i Virgili (URV). 43007 Tarragona, Spain, [5]Institute of Research and Technology in Agrifood Sector (IRTA), Environmental Horticulture. 08348 Cabrils, Barcelona, Spain, [*]Corresponding author. E-mail: *julia.martinez@uab.cat*

Evaluating the environmental behaviour of the agri-food products has become an essential issue, not only to avoid the pollution, but also to reduce potential threats to human health and to assure the provision of better quality food. Furthermore, novel demanding consumers are concerned about this information and they are also aware of the possible carbon footprint inflicted by their consumption habits. Carbon footprint has become a popular tool to measure the impact of products in the environment because it is easy to use and understand and the study times are shorter and cheaper than other environmental methodologies. One drawback of this approach is the different methodologies that simultaneously exist and that provide different, and sometimes contradictory, results regarding the environmental impact of products (Brenton et al., 2010).

This project aims to compare three well-known carbon footprint methodologies: The PAS2050:2011, the French Bilan Carbone and the WBCSD's GHG Protocol (see Table 1). The comparison highlights the different results due to variations in the procedure of each carbon footprint methodology when it is applied to agricultural systems (i.e. including capital goods, allocation criteria, inventory origin, system boundaries and expansion, carbon sinks, land use and/or transportation approaches among the most important variations). The study evaluates the environmental impact of five horticultural products: tomato, lettuce, beans, green peas and cucumber. The scope of the experiment also includes the evaluation of two fertilising treatments, including the application of mineral fertilisers and compost (Martínez-Blanco, 2011). The stages assessed include the fertilisers production and transport, the cultivation stage and the greenhouse phase.

The exclusion of capital goods represents a 4-16% decrease in the total CF when compared to the footprint measured by the GHG protocol and the Bilan Carbone. The inclusion of indirect processes (i.e. commuting employees) and the capital goods generate an increase of 12-50% in the CF. Additionally, when including the capital goods and adding a commercialisation stage (that includes packaging and transportation towards the retail markets) the CF suffers an increase of 6-40% from the one obtained under the PAS 2050 parameters. The packaging sub-stage provides 99% of the total impact within the commercialisation stage. These variations could be avoided if the system boundaries are extended to the end user stage, as suggested by the Bilan Carbone methodology; and including indirect processes, as considered by the GHG protocol. Paradoxically, still very few changes among the methodologies provide relevant variations (of more than 5%) in the carbon footprint calculations.

References

Brenton, P., Edwards-Jones, G., & Friis Jensen, M. Carbon Footprints and Food Systems. The World Bank Publications, 2010.

Martínez-Blanco, J. Muñoz, P., Antón A., Rieradevall, J. Assessment of tomato Meditarrean production in open-filed and standard multi-tunnel greenhouse, with compost or mineral fertilisers, from an agricultural and environmental standpoint, 2011. J Clean. Prod. 19, 985-997.

Martínez-Blanco, J. Rieradevall, J., Muñoz, P., Pascua, P., Oliver-Solà, J., Gabarrell, X., Gasol, C., 2012. Comparison of life cycle methodologies for the GHG assessment: CF and LCA in the horticulture sector . Int. J LCA.

Table 1. Main information of the Carbon footprint methodologies included in the study

Methodology	PAS 2050	Bilan Carbone	Greenhouse Gas Protocol
Developer	Bristish Standards Institute (BSI)	Agence de l'Environnment et de la Maîtrise de l'Energie (ADEME)	World Resources Institute (WRI) / World Business Council for Sustainable Development (WBCSD)
Country of origin	United Kingdom	France	United States
Last update	2011	2007	2007
Government sponsorship	Yes	Yes	No
Reach	Extensively used within the UK, but also in Japan, Western Europe and Northern Africa	Used only in France and Belgium	Used and promoted in the US, Brazil, Europe, India, China, Mexico and Australia
Annexes on agricultural products	Yes	Unidentified	Yes
Link	www.bsigroup.com	www.r-co2.com/Bilan-Carbone-ADEME--FRANCAIS	www.ghgprotocol.org

44. Carbon footprint and energy use of different options for greenhouse tomato production

Joana Almeida[1,*], Wouter Achten[1,2], Bruno Verbist[1], Bart Muys[1,*]

[1] Group Forest Ecology and Management, Department of Earth and Environmental Sciences, University of Leuven, Belgium, [2]Department of Metallurgy and Materials Engineering, University of Leuven, Belgium, [*] Corresponding authors. E-mail: *joana.almeida@ees.kuleuven.be; bart.muys@ees.kuleuven.be*

Rising sustainability awareness makes consumers increasingly interested in the environmental performance of the products on the market, concerning e.g. the emission of greenhouse gases (GHG). For this reason, there are market-driven environmental impact assessment tools, which can both inform consumers and be used by producers to highlight their sustainability efforts. The carbon footprint has been suggested as straightforward and resonant indicator for product environmental performance (Weidema et al., 2008).

In this case-study we report the carbon footprint and energy use of the production of *Cuore di bue* tomatoes in a greenhouse in Northern Italy with different energy-provision options. Despite the existence of other studies of this nature performed in Europe (e.g. Roy et al., 2009; Boulard et al., 2011), none has been performed in this region nor compares different energy supply configurations.

The conventional system consists of a polyethylene greenhouse supplied with grid electricity and heated through a natural gas boiler. The CO_2 released in the combustion is used to fertilise the plants. There are future plans to obtain heat through municipal solid waste (MSW) incineration and recover CO_2 from industry exhausts (scenario 1). Alternatively, both can be obtained from a co-generation facility (scenario 2), the latter providing also electricity. We assessed the life cycle GHG emissions and energy use of the present production chain and those alternatives. This exercise was performed according to appropriate official standards for life cycle assessment (ISO 14040-14044) and for carbon footprint (ISO 14067). Allocation was avoided by system boundary expansion. Assessment was made with software SimaPro® (PRé, the Netherlands), through the single-issue methods IPCC GWP 2007 100a and Cumulative Energy Demand (expressed in kg CO_2 eq and MJ eq, respectively). The functional unit is 1 kg of fresh tomatoes packed and delivered at the local market.

The carbon footprint of each kg of tomato produced in the current system is 2.33 kg CO_2 eq (Fig. 1-A). Coupled heating and CO_2 enrichment are responsible for 54% of GHG emissions. Construction of the greenhouse and fertilisation contribute 21% and 13%. The footprint can be decreased by 16% if MSW incineration supplies heat and CO_2 fertilisation (scenario 1). If the co-generation facility were to be installed (scenario 2), the carbon footprint would be lowered by 7% (2.15 kg CO_2 eq kg^{-1} tomato), owing to the excess electricity credited to the system. From the energetic point of view, the scenarios imply higher reductions (Fig. 1-B). While the energy use of the conventional system is 77 MJ eq kg^{-1} tomato, the co-generation and the waste valorisation options reduce it by 52 to 55, respectively. Heating and CO_2 fertiliser reach 75% of the energy input to the current system and 50% and 54% in scenarios 1 and 2 respectively. In all cases, construction and transport, packaging and waste disposal are the second and third energy consumers.

Literature presents wide ranges of GHG emissions and energy use of tomato production, influenced mainly by location and sophistication level of the system as described by Roy et al. (2009). These results are in line with averages from literature, even though *Cuore di bue* variety typically shows lower yields than conventional varieties.

References

Boulard T., Raeppel C., Brun R., Lecompte F., Hayer F., Carmassi G., et al., 2011. Environmental impact of greenhouse tomato production in France. Agron. Sust. Devel. 31(4), 757-77.

Roy, P., Nei D., Orikasa T., Xu Q., Okadome H., Nakamura N., Shiina T. 2009. A review of life cycle assessment (LCA) on some food products. J. Food Eng. 90(1): 1-10.

Weidema B., Thrane M., Christensen P., Schmidt J., Løkke S., 2008. Carbon footprint: a catalyst for Life Cycle Assessment? J. Ind. Ecol.12(1), 3-6.

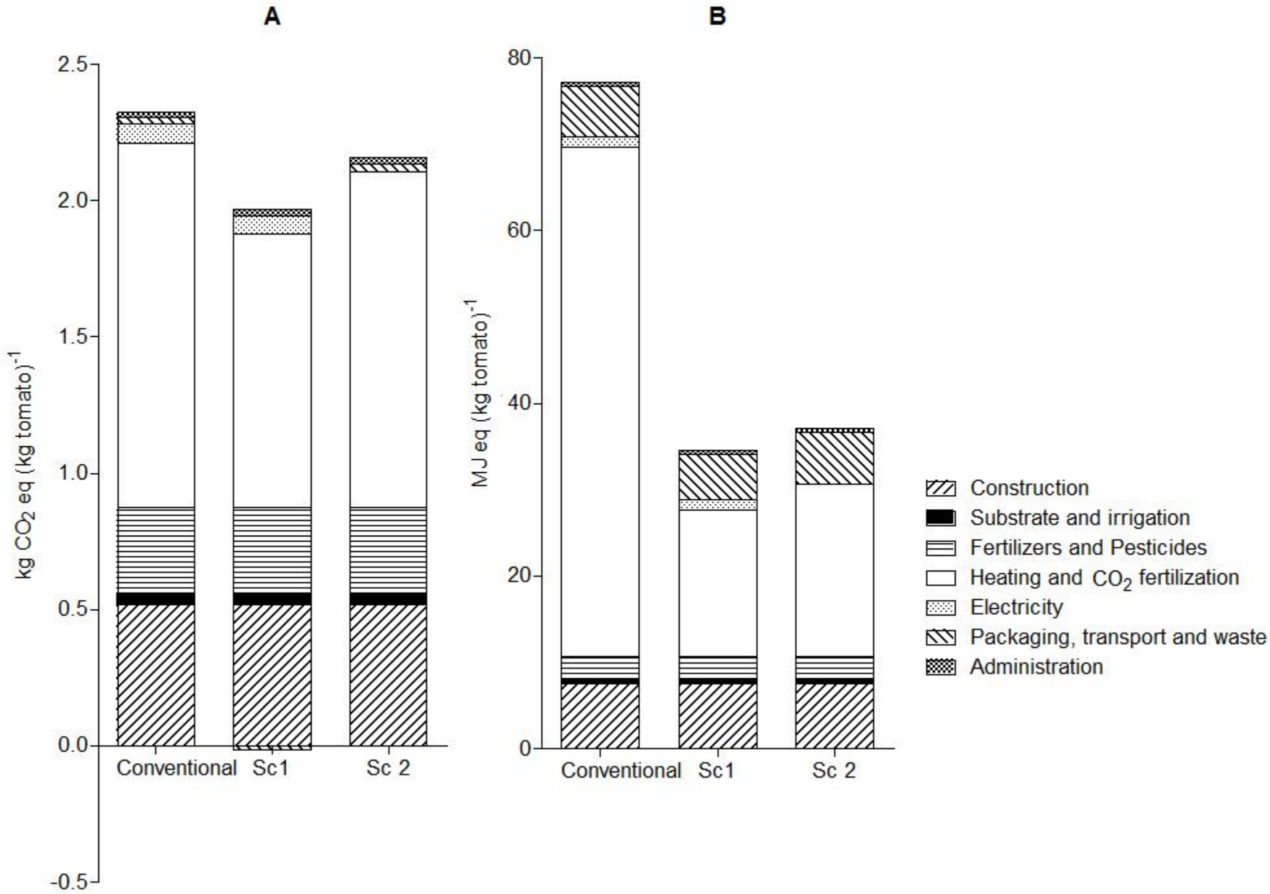


Figure 1. The carbon footprint (A) and cumulative energy demand (B) of the baseline tomato production system and scenarios 1 (municipal waste incineration) and 2 (cogeneration facility).

45. Assessment of Finnish cultivation practices with carbon footprint and energy index

Aki-Heikki Finér*, Tero Hirvi, Jaakko Laurinen, Pasi Lähdetie

Raisio Group, * Corresponding author. E-mail: *aki.finer@raisio.com*

Finnish food chain effect on climate change is estimated to be 14% of total climate change effect of Finnish consumption. Most of the food chain emissions are originated from agriculture. These emissions represent 69% of the total climate change impacts from the food chain. (Virtanen et al., 2011)
A method for determining the energy efficiency and environmental impact of a production of cultivation plants and for increasing positive environmental impact has been developed for Raisio Group's contractual farmers. Environmental impact and energy efficiency are characterised from contractual farmers' agricultural input-output data. Production parameters for cultivation are selected and cultivation procedures are performed by farmers, and further, the performed procedures are documented to give required information for environmental impact assessment and energy efficiency calculation.
Farm specific input-output data is collected after a produced crop yield is harvested. A representative sample of the crop yield is delivered to laboratory analysis with the required information. The grain sample is analysed and energy content of the yield is measured. Energy index and carbon footprint of the crop yield are calculated using the representative sample information. The system boundary used for energy index and carbon footprint calculation is cradle-to-farm gate. Energy index represents a ratio of energy content of the crop yield per hectare and energy used in cultivation practices per hectare. Functional unit for carbon footprint is a grain tonne.
Agricultural input-output information has been collected from contractual farmers during harvesting seasons 2008-2011. The input-output database includes 2000 to 2500 grain variety specific datasets from every season. Energy index together with CO_2 emissions of contractual farmers farming practices are estimated since 2008. Other greenhouse gas emissions are taken into account and a carbon footprint of contractual farmers' grain yield is calculated since 2009. Development of the carbon footprint and the energy index trends are followed and the most significant factors for development are identified. Information is used for instructional purposes and also for product carbon footprint calculation.

References

Virtanen, Y., Kurppa, S., Saarinen, M., Katajajuuri, J-M., Usva, K., Mäenpää, I., Mäkelä, J., Grönroos, J., Nissinen, A., 2011 Carbon footprint of food - approaches from national input-output statistics and a LCA of a food portion. Journal of Cleaner Production 19: 1849-1856.

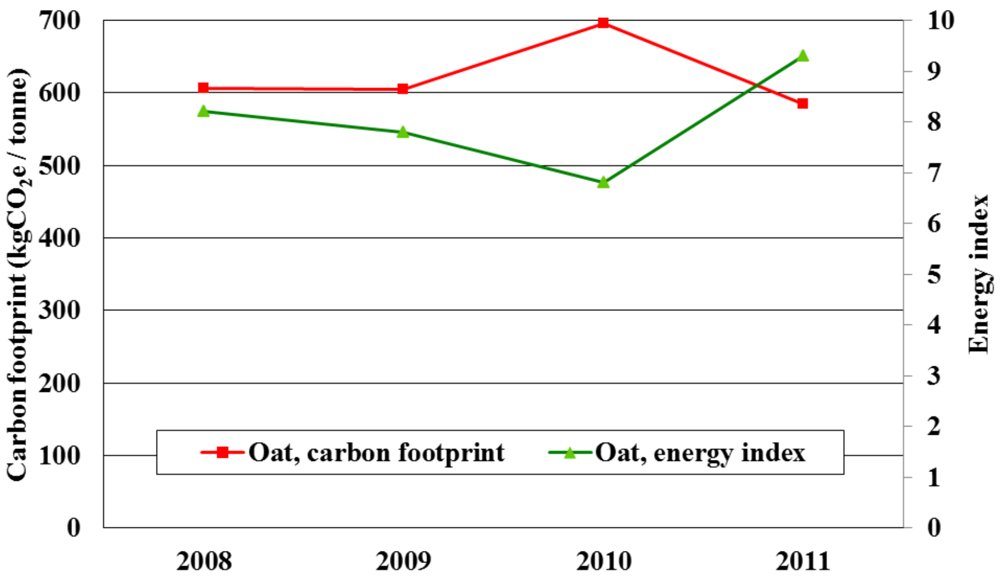


Figure 1. Average carbon footprint and energy index of oat produced by Raisio Group's contractual farmers during cultivation seasons 2008 to 2011.

46. Carbon footprint of Irish milk production

Ming-Jia Yan[1], James Humphreys[2], Nicholas M. Holden[1,*]

[1] UCD School of Biosystem Engineering, University College Dublin. Belfield, Dublin 4, Dublin, Rep. of Ireland, [2] Animal and Grassland Research and Innovation Centre, Teagasc, Moorepark, Fermoy, County Cork, Rep. of Ireland, [*] Corresponding author. E-mail: *nick.holden@ucd.ie*

The objective of this paper was to calculate the carbon footprint (CF) of Irish milk production on commercial dairy farms using life cycle assessment (LCA) to estimate the greenhouse gas (GHG) emissions to calculate CF. LCA of milk production has been carried out on commercial farms in Europe using both national statistics and ad hoc survey of farm-gate turnover. Most research has focused on comparing production types (e.g. organic vs. conventional), however relatively little work has examined how detail of farm management may influence CF.

In this paper, a four stage LCA was implemented following ISO 14040. The foreground data were based on a survey of specialist dairy farms, and the background data were taken from the Ecoinvent database. The functional unit was 1 kg energy corrected milk (ECM). Economic allocation was used for concentrate feed ingredients at the pre-farm stage. Emission factors (EFs) were taken from relevant literature while the EF for enteric CH_4 was determined by estimating net energy for maintenance, lactation, and pregnancy. Global warming potential of CH_4 and N_2O were taken as 25 and 298 times CO_2 equivalent. The system boundary was set at the farm gate and included production and transportation of fertiliser and concentrate feed and the on-farm activity. Infrastructure and machinery, soil carbon sequestration, pesticides, medicine and minor consumables were not included. In order to exclude on-farm activities not relevant to dairy production a proportioning rule was devised. This was done by (1) converting all animals into livestock unit (LU) equivalents according to the ratio of nitrogen excretion compared to a dairy cow as defined in the Irish "Statutory Instrument (SI) No. 610 (2010)"; (2) assuming the dairy herd consisted of dairy cows + replacement animals + bulls or suckler cows (if any), deriving the proportion factor of dairy herd as "dairy LU/total LU"; and (3) excluding from the farm GHG inventory electricity production, which was predominantly used by dairy herd, and multiply the rest by the proportion factor, and then adding back the electricity contribution to derive the dairy unit GHG. After proportioning, economic allocation between milk and meat (from surplus calves and culled cows) was performed based on farm sales records.

Much variation in the tactical management of the farms was found. For example, as much as 1.5-fold difference in fertiliser N input to support the same stocking density, and up to 2 fold difference in concentrate feed for a similar milk output per cow. The CF of milk production of the farms averaged 1.23 ± 0.16 kg CO_2 eq/kg ECM. CF was found to be correlated with various tactics using step-wise regression: milk per cow, economic allocation factor (both $P<0.001$) and diesel use per on-farm ha ($P < 0.005$), but no one indicator was a good predictor of CF. Effective sward management of white clover on a few farms appeared to lower the CF, but the signal was not all that clear because of other management tactics.

It was concluded that a combination of multiple tactics would determine CF of milk production on commercial dairy farms, and one of the most important indicators was milk output per cow, however this could not be used as a parsimonious predictor of CF on its own. The overall management efficiency of each farm is critical to achieving low CF for Irish milk.

References

Casey, J.W., Holden, N.M., 2005. The relationship between greenhouse gas emissions and the intensity of milk production in Ireland. J. Environ. Quality 34, 429-436.

GGELS, 2010. Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions. Joint Research Centre, European Commission.

Statutory Instruments (SI) No. 610, 2010. European communities (good agricultural practice for protection of waters) regulations 2010. Government Publications Sale Office, Sun Alliance House, Molesworth Street, Dublin 2, Dublin, Ireland.

47. Carbon footprint of the Australian dairy industry

Sven Lundie[1], Jake McLaren[2,*], Neil van Buuren[3], Sebastian Gollnow[1,2], Andrew Moore[2], Larisa Maya Altamira[2], Daniel Thylmann[1], Barbara Nebel[2], Hazel v. Rowley[4], Manfred Lenzen[5] , Karen Christie[6], Torsten Rehl[1], Sabine Deimling[1]

[1] PE International AG, Stuttgart, Germany, [2] PE Australasia Ltd., Wellington, New Zealand, [3] Dairy Australia, Melbourne, Australia, [4] Sustainability Assessment Program, Water Research Centre, The University of New South Wales, Sydney, Australia, [5] Integated Sustainability Assessment Group, University of Sydney, Australia, [6] Tasmanian Institute of Agricultural Research, University of Tasmania, Hobart, Australia, [*] Corresponding author. E-mail *j.mclaren@pe-international.com*

This abstract describes an overview of a carbon footprint project conducted (2010-12) by PE International, PE Australasia and a consortium of research partners. The project sponsor is Dairy Australia acting on behalf of the Australian national dairy industry. This abstract outlines the project aims, methodology, approach adopted and lessons learnt.

Overall the study aims to assist Australian dairy companies to understand the carbon footprint of their products, and develop long term strategies to reduce hot spots in the carbon footprint of dairy products. The overarching project aims are to:

- Quantify the carbon footprint of the major Australian dairy products produced for export, i.e. butter, cheese, fresh products, milk powders, nutritional and UHT-products, whey and lactose, from farm-to-distributor's warehouse or export harbour. Altogether, 12 products have been investigated.
- Represent the weighted Australian average product carbon footprint for selected products based on annual production figures by region and by farm practices;
- Establish an auditable monitoring system and framework for a reproducible carbon footprint reporting that can be updated and expanded to include other environmental impacts in the future.

Meeting the goals of the project requires collection of primary LCA data from approximately 150 farms and and 15 dairy representative processing sites across Australia operated by the major dairy companies.

The methodology for this study is in line with ISO standards on Life Cycle Assessment and the sector specific Carbon footprinting guidelines of the International Dairy Federation (2010). Following the IDF guidelines (IDF, 2010) raw milk intake and transportation is allocated on the basis of the milk solids content of the final product. Operations within the processing plant are modelled as detailed as the data availability allow. Three modelling and approaches are possible for addressing allocation issues at the processing plants.

The study includes all relevant activities from the growing of grass on farm to feed the cows, to the delivery of dairy products to warehouse or export harbour. On farm site this includes emissions from mechanical as well as non-mechanical sources. For the product processing the system boundary contains all relevant activities from collection of the milk from the farm, product processing, through to export harbours for export products, and for fresh products to the retailers' distribution warehouse.

The scale of data collection and complexity of modelling approach for this project is extensive. The software solution utilised is a linked solution, combining both GaBi and SoFi software packages from PE International (Fig. 1). Data was collected using customised web-based questionnaires in the SoFi software which allow individual producers to submit their data in a secure and auditable environment.

References

International Dairy Federation (2010) "A common carbon footprint methodology for the dairy sector", November 2010

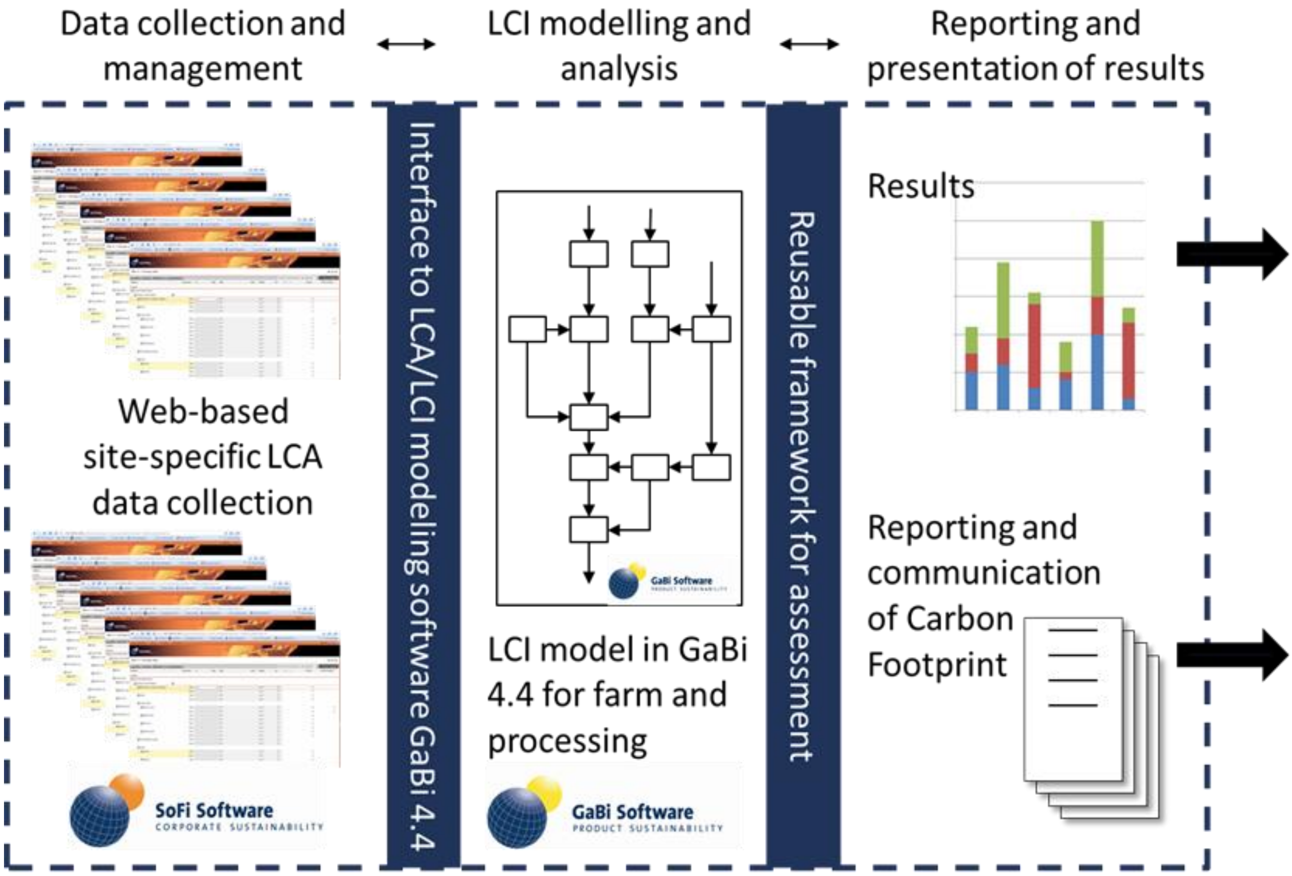


Figure 1. Integrated data collection and processing approach. Web-based data collection in SoFi (left), LCA/LCI modelling in GaBi, and final reporting and communications.

48. Contribution of packaging to the carbon footprint of canned tuna

Ngamtip Poovarodom*, Chularat Ponnak, Natthaphon Manatphrom

Department of Packaging and Materials Technology, Faculty of Agro-Industry, Kasetsart University, Bangkok 10900, Thailand, * Corresponding author. E-mail: *fagintp@ku.ac.th*

Packaging is an essential accompaniment of almost every food product. It performs multiple functions to ensure the safety and satisfactory condition of food products delivered to end users. Packaging helps minimise food losses, consequently reducing the environmental burden arising from excessive food production. In addition, packaging is a key element in creating new food products to meet consumer needs. For instance, canned foods discovered by a French chef, Nicholas Appert, more than 200 years ago have increasingly been presented in retort containers which are more convenient and attractive than metal cans.

Nevertheless, packaging has a negative impact on the environment, the extent of which depends significantly on the choice of materials and the efficiency of material recovery (Hospido et al., 2006; Humbert et al., 2009; Mungkung et al., 2010). These circumstances present an additional challenge to the food industry: selecting a packaging system which could increase consumer satisfaction while having less impact on the environment.

This study was thus initiated to identify and compare the carbon footprint of different packaging systems used to provide one single-serve meal of canned food which tuna was selected as a study product. It also aimed at enhancing the application of LCA study at the early stage of a new product development and a product improvement. The study was conducted in accordance with publicly available specification PAS 2050:2008 (BSI, 2008), with respect to LCA methodology covered by ISO 14040 and ISO 14044.

Single-serve packages selected were: (1) two-piece cans made of chrome-coated steel, with an aluminium pull ring tab on the top; (2) retort pouches made of polypropylene (PP), aluminium foil (Al) and oriented nylon (ON), referred to as PP/Al/ON/PP; and (3) retort cups made of PP and ethylene vinyl alcohol (EVOH), referred to as PP/EVOH/PP with lids made of polyethylene terephthalate (PET), aluminium foil, ON and PP, referred to as PET/Al/ON/PP. The system boundary, Figure 1, covered the production of tuna meat, the production of packages, product assembly, processing, packing, transport, and disposal, excluding consumption. Primary data were collected for each main activity in the production line, including filling, closing or sealing, sterilising, cooling, labelling, and unitising. As sterilisation is an energy-intensive and batchwise process, primary data on energy use and steam consumption were carefully measured.

The life cycle GHG emissions associated with a single serving of tuna using different packaging systems are shown in Figure 2. The manufacturing process of retort pouches and cups produced 60% and 70% less GHG emissions, respectively, than that of metal cans. However, the overall carbon footprint of canned tuna in retort cups was 10% and 22% less than when packaged in metal cans and retort pouches, respectively. Packaging and its associated processing constituted significant fractions of the product's carbon footprint, ranging from 20-40%. These findings show that the advantage of low GHG emissions embodied in plastic packaging might vanish if the associated processes are not optimally managed. To reduce a product's carbon footprint, the choice of food packaging thus depends not only on the materials but also on the further processing involved. Hotspots in the life cycle assessment of canned foods are packaging production and disposal, and product sterilisation. The improvement of retort operation in terms of capacity and energy utilisation, and the efficiency of post-consumption packaging material recovery, are the key factors responsible for the reduction of a product's carbon footprint. These issues present a challenge to both the food industry and local authorities.

References

British Standards Institute (BSI), 2008. PAS 2050:2008: Specification for the Assessment of the Life Cycle Greenhouse Gas Emissions of Goods and Services. British Standards Institute, London.

Hospido A, Vazquez ME, Cuevasc A, Feijoo G, Moreira MT., 2006. Environmental assessment of canned tuna manufacture with a life-cycle perspective. Resour. Conserv. Recycl. 47(1): 56-72.

Humbert S, Rossi V, Margni M, Jolliet O, Loerincik Y., 2009. Life cycle assessment of two baby food packaging alternatives: glass jars vs. plastic pots. Int. J. Life Cycle Assess. 14(2), 95-106.

Mungkung R, Gheewala SH, Kanyarushoki C, Hospido A, van der Werf H, Poovarodom N, Bonnet S, Aubin J, Moreira MT, Feijoo G., 2010. Practical issues of product carbon footprinting: experiences from Thailand. In Proc. of the 7th Int. Conf. on LCA in the Agri-Food Sector. Bari, Italy.

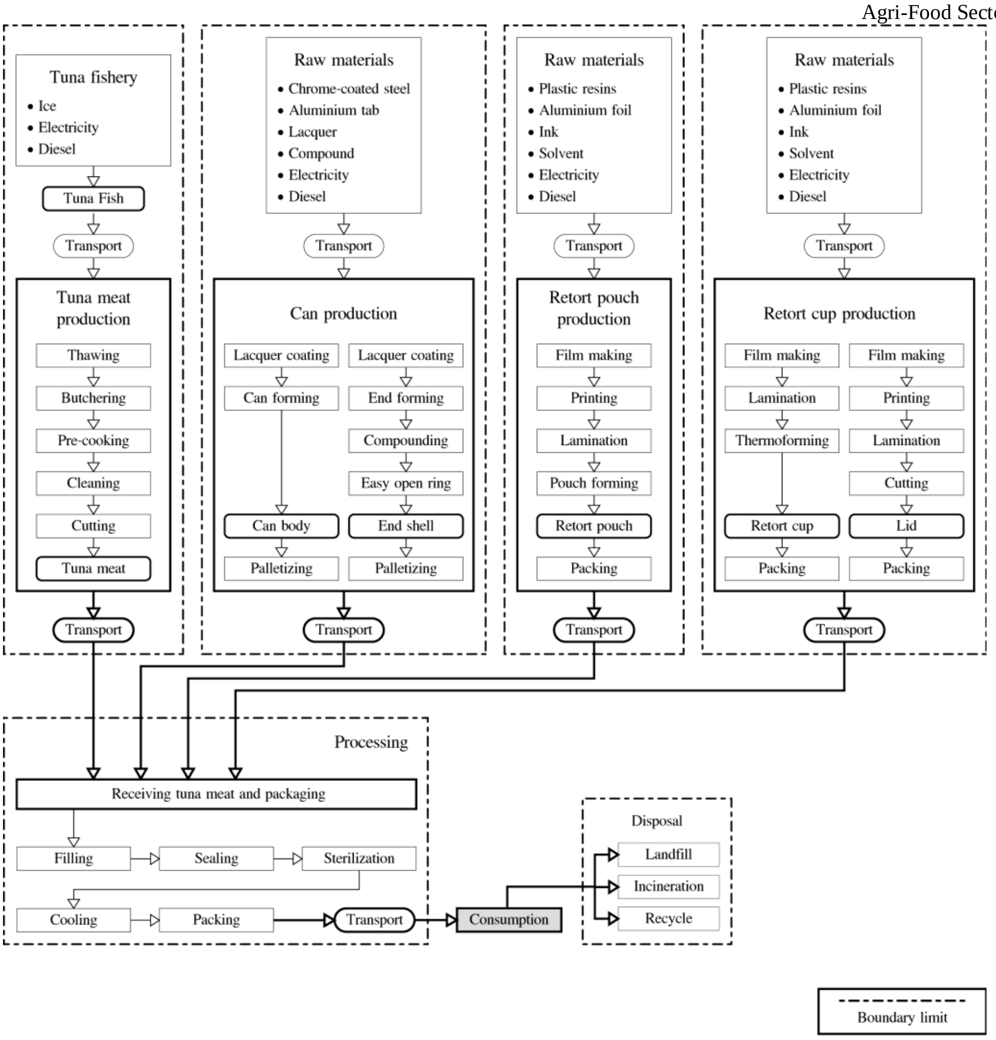


Figure 1. System boundary of carbon footprint assessment for canned tuna products, excluding consumption.

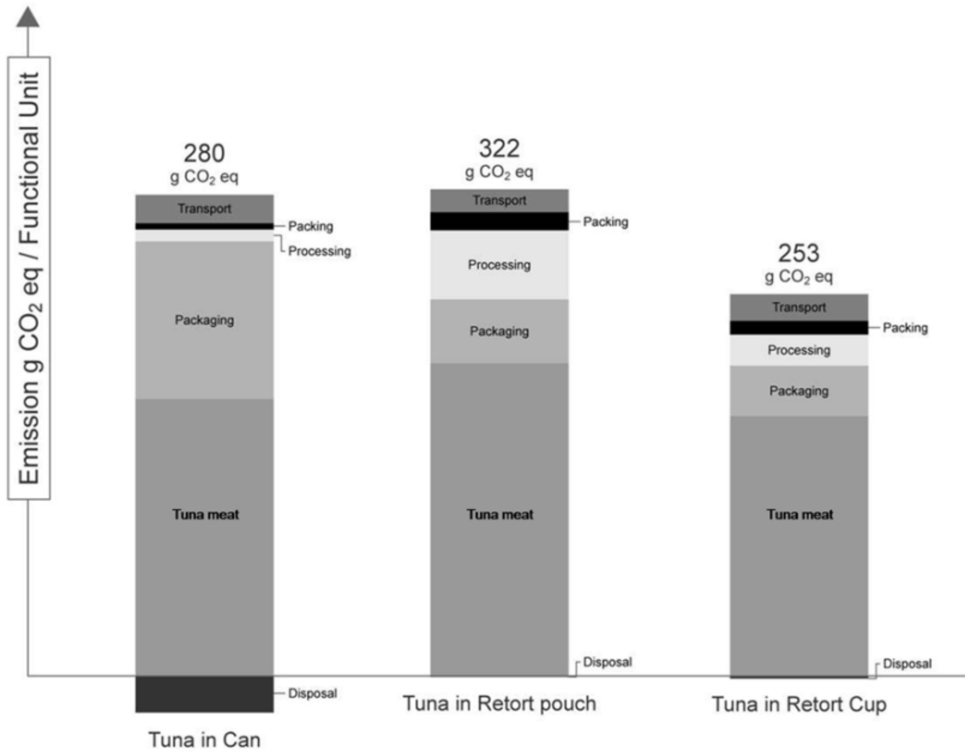


Figure 2. Total carbon footprint of different packaging systems for tuna.

49. Carbon footprint of Canadian dairy products: national and regional assessments

Xavier P. Vergé[1], Dominique Maxime[2], Raymond L. Desjardins[3,*], Yves Arcand[4], Devon Worth[3]

[1] Agro-environmental consultant, Ottawa ON, Canada, [2] CIRAIG, École Polytechnique de Montréal, Montréal QC, Canada, [3] Agriculture and Agri-Food Canada, Eastern Cereal and Oilseed Research Centre, Ottawa ON, Canada, [4] Agriculture and Agri-Food Canada, Food Research and Development Centre, Saint-Hyacinthe QC, Canada, [*] Corresponding author. E-mail: *ray.desjardins@agr.gc.ca*

The production of dairy products such as milk, cheese, butter and yoghurt results in a wide range of greenhouse gas (GHG) emissions per kg of product. A calculator, which was recently developed to estimate the magnitude of the GHG emission intensities of dairy products from cradle to the exit gate of the processing plants, at the provincial and national scales, will be presented (Vergé et al., 2012). Estimates based on a regional assessment of farming and processing systems will be given for eleven dairy products for 2006 (Fig. 1). The on-farm GHG emission estimates, which are based on the IPCC Tier 2 methodologies adapted for Canadian conditions (e.g. Vergé et al., 2006*;* Rochette et al., 2008) also account for GHG emissions associated with farm inputs (Dyer and Desjardins, 2003a, 2003b). Much of this work is based on previous work from Vergé et al. (2007) and has been improved 1) to account for co-products allocation where relevant; 2) for a more comprehensive assessment of background processes and to be fed by yearly statistics instead of the five-year Census of agriculture data. For the processing phase, a top-down approach (e.g. for energy inputs gathered from yearly statistics) is used, and data gaps (e.g. packaging) have been filled with North-American generic data (Vergé et al., 2011). The dairy plant is seen as a multi-product output plant and co-products allocation is performed using the physico-chemical approach described by Feitz et al. (2007), which has been modified to incorporate the solids characteristics (i.e. protein and fat content) of Canadian dairy products (Maxime et al., 2011). The contribution of each step to the overall carbon footprint of dairy products will be discussed. Information on the magnitude of the carbon footprint is likely to become important information for dairy producers for selling their products on the international scene.

References

Dyer, J., and R.L. Desjardins. 2003a. Simulated farm field work, energy consumption and related greenhouse gas emissions in Canada. Biosystems Engineering 85(4), 503-513.

Dyer, J.A. and R. L. Desjardins. 2003b. The impact of farm machinery management on the greenhouse gas emissions from Canadian agriculture. Journal of Sustainable Agriculture 22(3), 59-74.

Feitz A.J., Lundie, S., Dennien, G., Morain, M., Jones M., 2007. Generation of an Industry-specific Physico-chemical Allocation Matrix. Application in the Dairy Industry and Implications for Systems Analysis. Int. J. LCA 12(2), 109-117.

Maxime D., Vergé, X., Arcand, Y., Desjardins, R., 2011. A cradle-to-gate assessment of the carbon footprint of Canadian dairy products. LCA XI Conference, October 4-6, 2011, Chicago IL (USA).

Rochette P., Worth, D.E., Huffman, E.C., Brierley, J.A., McConkey, B.G., Yang, J., Hutchinson, J.J., Desjardins, R.L., Lemke, R. and Gameda, S., 2008. Estimation of N_2O emissions from agricultural soils in Canada. II. 1990 2005 inventory. Can. J. Soil Sci. 88, 655-669.

Vergé, X. P. C., Worth, D., Hutchinson, J. J. and Desjardins, R. L. 2006. Greenhouse gas emissions from agro-ecosystems in Canada. Methane emissions: Methodology for calculations. Pages 38 Technical Report. Agriculture and Agri-Food Canada, Ottawa, ON.

Vergé X.P.C., Dyer, J.A., Desjardins, R.L., Worth, D., 2007. Greenhouse gas emissions from the Canadian dairy industry in 2001. Agricultural Systems, 94(3), 683-693.

Vergé X.P.C., Maxime D., Dyer, J. A., Desjardins, R.L. and Arcand Y., 2011. Carbon footprint model for Canadian livestock production - Presentation and application to the dairy sector. Technical report (August 2011); Contract # 01E86-110116/001/SS. 37p.

Vergé X.P.C., Maxime D., Dyer, J. A., Desjardins, R.L. and Arcand Y., 2012. Carbon footprint calculators for Canadian livestock production – The ULICEES and (cafoo)² models. Technical report (January 2012); Contract # 01E86-110116/001/SS. 47p.

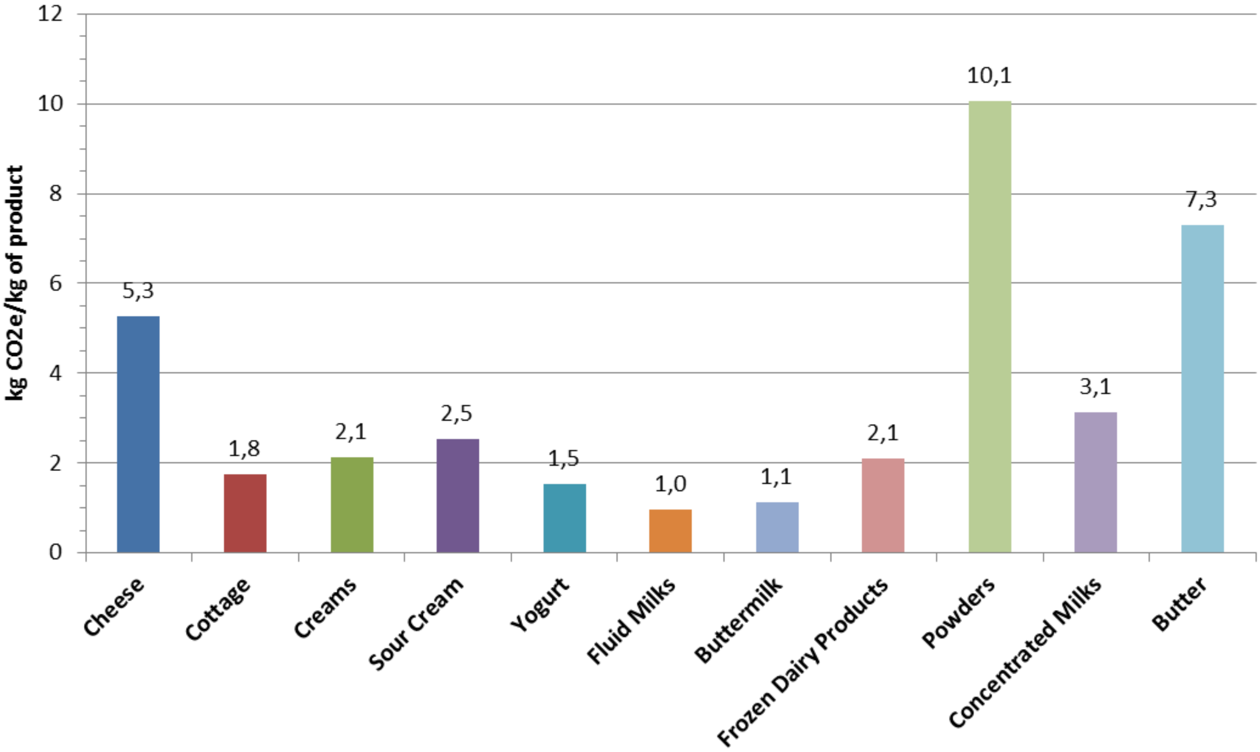


Figure 1. Cradle-to-gate climate change impact of Canadian dairy products in 2006.

50. Carbon footprint of Flemish livestock products

Ellen Bracquené*, Valerie Vandermeulen, Marianne Fernagut

Environment Resources Management, Brussels, Belgium, * Corresponding author. E-mail: *ellen.bracquene@erm.com*

The purpose of this study is to develop a monitoring tool for the carbon footprint of the Flemish livestock farming produce. A carbon footprint quantifies the climate change impact of an activity, product or service. Considering the current interest to mitigate the consequences of climate change, it is important to have a clear understanding of this impact. When developing a monitoring tool related to the impact of food production systems, horticulture and fishery products should also be taken into account, however this is not covered by the scope of this study. Currently there are several international carbon footprint calculation initiatives being developed. However, there is no one unambiguous international standard or specific rules for agricultural products.

Using the available international standards, a functional and transparent monitoring tool has been developed to determine the carbon footprint of the Flemish livestock farming products. This study focuses on the livestock industry and more specifically on the supply chain of beef, pork and milk. A method was formulated and applied to these product groups, revealing the influencing parameters and 'hotspots' within the parts of the food chain this study focused on.

Where the results show the most potential for reducing the emissions, the 'hotspots' identified will have recommendations on how these can be implemented. This can help the Flemish government or the stakeholders involved to develop a strategy for the reduction of greenhouse gas emissions. In this way, a carbon footprint may contribute to make the agro food chain in Flanders as well as the livestock industry in particular more sustainable.

For this the general standards about carbon footprinting such as the PAS2050 and the ISO14067 have been used. For milk, there was the additional use of the guidelines of the International Dairy Federation (IDF). Critical points of the methodology that have an influence on the carbon footprint are the choice of functional unit, the system boundaries and the allocation method (how to allocate the emission of greenhouse gases among the various co-products emerging from a single process). All these were addressed during the study.

Data and knowledge has been collected from the faculty of bioscience engineering of Ghent University, data of Bemefa, the farmers union (Boerenbond) and the ILVO. The data regarding other levels in the supply chain (mainly the processing industry) have been primarily collected through in-depth interviews and primary activity data. Within the current scope of the study, no primary data were collected at the farm level.

In general it can be stated that the data used are of good quality and represent livestock farming within Flanders. The principal part of the activity data originate from reliable sources, however there is always some natural variability. This is dependent of external factors (e.g. differences in breeds, farming yields will differ yearly, the composition of feed concentrates can vary depending on the available commodities and market prices). Furthermore a number of data points cannot be easily measured under real life conditions (e.g. the consumption of feed mixtures per animal and the number of days on the pasture), and estimation is necessary based on expert opinion. The data used were also verified and validated by the members of the steering committee. The emission factors originate from acknowledged life cycle inventory databases and literature sources, and can be considered as representative for Flanders. Other available sources were used to cross-check the values and are reported in a transparent way.

The carbon footprint results depend on the data quality of both the collected activity data and the available emission factors. The data and calculations in the developed carbon footprint models of the current project are highly detailed. In some cases however, reference values and standard formulas from relevant literature or other existing models were applied. In future, the current carbon footprint models could be refined and improved if additional data is collected. The shortcomings and restrictions of the current dataset are reviewed below. Recommendations are made to solve the identified knowledge gaps in Flanders.

Table 1. Overview of emission sources within the covered system boundaries

Name	Greenhouse gasses	Description
Feed mixtures and material for bedding of stables (own production)	CO_2 and N_2O	Diesel is taken into account within energy consumption. Production and transport of fertilisers, pesticides en lime. Impact of fertilisers and the use of lime (direct and indirect) and the impact of crop residues are taken into account according to the IPCC method (Tier 2 calculation).
Feed mixtures (purchased)	CO_2 and N_2O	Farming, transport, processing en land conversion is taken into account in de covered emission factors.
Animal (stomach-intestine fermentation)	CH_4	The IPCC method is applied (Tier 2 calculation).
Manure storage and disposal	CH_4 and N_2O	The IPCC method is applied (Tier 2 calculation).
Manure application (not used for own feed mixtures)	CH_4 and N_2O	Allocation between animal (40%) and vegetable production system (60%) on the basis of nitrogen uptake by plants.
Energy and water consumption[1]	CO_2, CH_4 and N_2O	Energy consumption (electricity; diesel; red diesel; gas) Water consumption (tap and ground water)
Transport of goods	CO_2, CH_4 and N_2O	Assumptions are being made for the goods entering and leaving the farm.
Processing materials	CO_2, refrigerant	Use of cleansing products and refrigerants

1: water is not a source of emission; however the use of energy for processing and transporting tap water is taken into account.

Table 2.Overview of the applied allocation method

Process	Products	Allocation method
Farming of crops	Products for human consumption (like flower); Products for animal consumption (like wheat starch) Other products (like straw)	Economic allocation
Dairy sector	Milk and meat	Physical relation registered by the IDF
Dairy products	Low-fat (skimmed), medium-skimmed and whole milk, cream, milk powder, yoghurt, butter, …	Physical relation registered by the IDF
Manure production	Stock farming products and farming of crops	Physical relation
Slaughtering and deboning of carcass	Meat, bones, fat, skin/hide, hart, blood, etc.	Economic allocation

51. Carbon footprint of organic crop production in Sweden

Christel Cederberg*, Magdalena Wallman, Maria Berglund, Jenny Gustavsson

SIK – The Swedish Institute for Food and Biotechnology, Sustainable Food Production, Box 5401, SE-402 29 Göteborg, Sweden, * Corresponding author. E-mail: *Christel.Cederberg@sik.se*

The aim of the study was to assess the carbon footprint (CF) of Swedish organic crop production, and to explore the need of better statistics on organic crop production to improve environmental system analyses. The CF of organic crop production was compared to conventional production when possible. Improvements were suggested on how to reduce greenhouse gas emissions from organic cultivation. The analyses were based on five organic crop rotations reflecting the current situation on organic farms with typical operations (milk and arable farms) in three regions in Sweden. In this study the whole crop rotation was included in the environmental analysis since organic nitrogen applications and weed control is managed over the whole crop rotation and not for an individual crop for a single year. There was a lack of reliable statistics; hence information regarding crop rotations, yields, fertiliser management etc. was collected in cooperation with agronomic advisors. Statistics were used for geographic allocation of different crops and the use of organic fertiliser products to reflect the current Swedish situation. Greenhouse gas emissions were estimated according to IPCC (2006) and the functional unit was 1 kg crop at farm gate/farm storage.

The CF per kg organic crop was mainly affected by yield levels and nitrogen management strategy. High yield levels combined with moderate applications of organic fertiliser reduced the CF for organic cereal crops. Organic silage production (grass and clover) production had a lower CF than conventional silage due to relatively high yield levels and good crop management. An individual crop's sequence in a given crop rotation had also a substantial influence on the overall CF. The distribution of nitrogen emissions between individual crops in the rotation must be considered. In this study, the allocation was made between the nitrogen fixing crop and the subsequent crops as the N fixing crops are cultivated to provide other crops with nitrogen. Crop rotational effects are more evident in organic agriculture than in conventional and therefore there is a need to develop a uniform methodology to estimate the CF of organic production.

References

Cederberg, C., Wallman, M., Berglund, M., Gustavsson, J. 2011. Klimatavtryck av ekologiska jordbruksprodukter [In Swedish; Carbon footprint of organic crop production]. Report 830, SIK – The Swedish Institute for Food and Biotechnology, Göteborg.

IPCC. 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Vol. 4. Agriculture, Forestry and Other Land Use. Chapter 11. N2O Emissions from Managed Soils, and CO2 Emissions from Lime and Urea Application. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

52. Guidelines on inclusion of land use change emissions in carbon footprints of food

Hannele Pulkkinen*, Kristoffer Krogerus, Juha-Matti Katajajuuri

MTT Agrifood Research Finland, Biotechnology and Food Research, FI-00790 Helsinki, Finland, * E-mail: *hannele.pulkkinen@mtt.fi*

Currently there exist several methodologies for the calculation of greenhouse gas emissions resulting from a land use change, but no methodology has been commonly accepted for use in LCA. The methodologies produce differing results, because, for example, their emission boundaries differ (e.g. only some include the change of gas flux direction and magnitude). In Finland a harmonised national methodology for calculating carbon footprints of food is developed in 2009-2012. As a part of it a more practical methodology for the Finnish food industry to calculate emissions from land use change on a product level was also developed.
In Foodprint, the methodology for estimating carbon stock changes is based on the IPCC 2006 Guidelines. FAOSTAT statistics on trends of crop area expansion are used for assessing which land use changes have occurred, based on the methodology originally presented by Blonk et al. (2009). The trends of crop area expansion are calculated based on the mean of all annual crop area changes from the last 20 years, instead of using linear regression to predict the crop area trend, since it was realised that when large changes in crop area had occurred during a studied period, the predicted land use changes would be distorted.
The carbon storages and their changes due to land use change are to be evaluated according to the method described in European Commission guidelines (EC 2010), which itself is based on the IPCC 2006 methodology. FAO's *Global Forest Resource Assessment* is the preferred source for evaluating carbon stocks in above-ground, root and litter biomass. For assessing the change in carbon stocks, the GHGV approach (Anderson-Teixeira, 2011) was also considered, but it was seen that it did not provide remarkably more accurate results, while at the same time being much more work intensive.
The method used in the Foodprint guidelines for allocating emissions resulting from land use change to agricultural products, using the trends of annual crop area expansions and reductions according to FAOSTAT statistics, is specified in more detail in Ponsioen and Blonk (2010). The same method is also presented in GHG Protocol Product Standard. Other allocation methods were also analysed, such as using scientific articles and their results for estimations, e.g. one of Prudêncio da Silva (2010), which can be more accurate, but impractical for harmonising footprint calculations in Finnish food industry.
Further land allocation was studied, which utilises the Ponsion and Blonk methodology and an assumption that the deforested crop area does not remain as agricultural land, but ultimately ends up as pasture and secondary forests as well. It was though seen, that it can be too complicated for food companies to find such data from literature, and therefore this is optional in the Foodprint guidelines.
The Foodprint methodology was tested by assessing the increase in carbon footprint of a processed broiler product caused by land use changes. In the production chain, the methodology predicted that land use changes occurred only in the cultivation of Brazilian soya. The initial carbon footprint was 3.6 kgCO_2e/kg product, while the new (including emissions from land use changes) one was 4.1 kg CO_2e/kg product.

References

Anderson-Teixeira, K., DeLucia, E., 2011. The greenhouse gas value of ecosystems. Glob. Change Biol. 17, 425-438.
Blonk, H., Ponsioen, T., Scholten, J., 2009. Carbon Footprinting of Horticultural Products. Available at: http://www.tuinbouw.nl/files/page/carbon%20footprint.pdf.
EC, 2010. Commission Decision of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC.
Ponsioen, T., Blonk, H., 2010. Allocating greenhouse gas emissions from land conversion. Proceedings of LCA food 2010. 1, 551-556.
Prudêncio da Silva, V., van der Werf, H., Spies, A., Soares, S., 2010. Variability in environmental impacts of Brazilian soybean according to crop production and transport scenarios. J. Environ. Manage. 91, 1831-1839.

53. A simplified tool for the estimation of N and P emissions due to the use of fertiliser in a LCA study

Filippo Sessa[1,*], Massimo Marino[1], Pierluigi Meriggi[2], Matteo Ruggeri[3], Michele Zerbini[4]

[1] Life Cycle Engineering, [2] Horta srl, [3] Univerisità Cattolica del Sacro Cuore, Piacenza, [4]Barilla G. R. Fratelli S.p.a, [*] Corresponding author. E-mail: *sessa@studiolce.it*

Emissions of nitrogen and phosphorus compounds from soil are among the most relevant sources of environmental impact in the life cycle of most agricultural products. The most important emissions in air are N_2O, which has a high Global warming potential and NO and NH_3 which contribute to acidification. The most important emissions in water are nitrates and phosphates, which contribute to eutrophication of water. Agricultural practices such as fertilisation and tillage could seriously affect these kinds of emission.
It is often difficult to estimate these emissions because they depend on a multitude of factors such as climate conditions, soil characteristics and cultivated crops. There are models in literature that enable to estimate them with a certain precision but in many cases they are too complicated and require input which would imply money and time efforts that are not always available in a life cycle study. On the other hand it is possible to estimate emissions using emission factors or equations which however are too simplistic and do not permit for example to appreciate differences among different farming systems. The aim of this study was to identify models that may be easy to handle and in the meanwhile enough accurate to take into account the main factors that influence the emissions. This is important in particular when performing a Life Cycle Assessment on a farm or on a territory level. With the individuated models a tool was then developed to permit the life cycle analyst to get the values of emissions by inserting a few required inputs. Particular attention in the choice of the models has been given to the responsiveness to different types and dosage of fertilisers.
The models individuated are Bouwman et al. (2002) for direct N_2O emissions, IPCC (2006) for indirect N_2O emissions, Bouwman et al. (2002) for NH_3 emissions from mineral fertilisers, Dohler et al. (2002) for NH_3 emissions from organic fertilisers, Stehfest et al. (2006) for NO emissions, De Willigen (2000) for nitrates emissions and Prahsun (2006) for phosphorus emissions.
Most of the input required can be obtained by asking the farmer and by soil analysis. Other data required are the quantity of annual precipitation and the type of climate, which can be easily obtained from literature or from some meteorological database. Some factors required by the models are taken from literature and provided in the tool.
The result of the work was a tool that enables the life cycle analyst to easily get quite accurate estimates of nitrogen and phosphorus emissions from agricultural soils. The development prospect for the tool could enrich the internal database to make it more specific for determined crops or geographical areas.

References

Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002. Estimation of global NH_3 volatilization loss from synthetic fertilizers and animal manure applied to arable lands and grasslands. Global Biogechem. Cycles 16(2), 1024.

Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002. Modeling global annual N_2O and NO emissions from fertilized fields. Glob Biogeochem Cycles. 16(4), 1080-1107.

IPCC, 2006. Guidelines for National Greenhouse Gas Inventories

Prasuhn, V., 2006. Erfassung der PO_4-Austräge für die Ökobilanzierung. SALCA-Phosphor. Zürich, CH, Agrocope FAL Reckenholz.

Stehfest, E., Bouwman, L., 2006. N2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modelling of global annual emissions. Nutrient Cycling in Agroecosystems. 74, 207-228.

54. Soil and biogenic carbon accounting in carbon footprint: potential and challenges

Edouard Périé[1,2,3,*], Sarah J. McLaren[1,3], Brent E. Clothier[2,3]

[1] Massey University, Palmerston North, New Zealand, [2] New Zealand Institute for Plant & Food Research Ltd, Palmerston North, New Zealand, [3]New Zealand Life Cycle Management Centre, New Zealand, [*] Corresponding author. E-mail: *Edouard.perie@plantandfood.co.nz*

Carbon sequestration in soil has the potential to counterbalance a significant proportion of life-cycle greenhouse gas (GHG) emissions associated with (at least) some horticultural products, as well as improving soil health and orchard productivity (Lal, 2010). Furthermore, potentially significant quantities of atmospheric CO_2 can also be stored in the standing biomass of perennial crops (Albrecht and Kandji, 2003).
However, the most widely used standard for GHG accounting, the PAS 2050 (BSI, 2011), currently does not include above-ground biomass and changes in soil carbon stocks as a result of land use (unless provided for in supplementary requirements). This is due to a lack of an agreed methodology, and uncertainty as to how to measure these parameters and integrate them into a carbon footprint. Indeed, at the inventory phase, it is difficult to measure accurately a change in the soil carbon stock over short time periods, whilst satisfying statistical significance and power levels (Post et al., 2001) because of the spatial variability of carbon stocks in soils and their small change with time. Measurement methods are costly and time consuming – and thus not easily implementable. Regarding methodology, the grower's potential to store carbon is site-dependent due to the variability in the carbon storing capacity of different types of soil; arguably, this should be reflected in a carbon footprint calculation. Furthermore the timeframe adopted for measurement of carbon stock changes can have an important impact on the carbon footprint results (Milà i Canals et al., 2007), because changes are often not linear over time. Lastly, maintenance of soil carbon is also important and it may be desirable to account for this aspect. In this poster, we describe the challenges and the requirements for the development of a reliable and practical methodology to measure soil and biogenic carbon stocks changes over time in apple orchards, summarise methodological issues related to their integration into carbon footprint, and discuss potential solutions.

References

Albrecht A, Kandji ST, 2003. Carbon sequestration in tropical agroforestry systems. Agriculture Ecosystems & Environment 99(1-3):15-27.

BSI, 2011. PAS 2050:2011 Specification for the Assessment of the Life Cycle Greenhouse Gas Emissions of Goods and Services. London, UK: British Standards Institute.

Lal R., 2010. Beyond Copenhagen: mitigating climate change and achieving food security through soil carbon sequestration. Food Security 2(2):169-77.

Milà i Canals L., Romanyà J., Cowell SJ, 2007. Method for assessing impacts on life support functions (LSF) related to the use of "fertile land" in Life Cycle Assessment (LCA). Journal of Cleaner Production 15:1426-1440.

Post WM, Izaurralde RC, Mann LK, Bliss N., 2001. Monitoring and verifying changes of organic carbon in soil. Climatic Change 51(1):73-99.

55. Using LCA to inform policy for biochar in New Zealand

Ruy Anaya de la Rosa[1], Sarah J. McLaren[1,2,*], Jim Jones[1], Ralph Sims[1]

[1] Massey University, Palmerston North, New Zealand, [2] New Zealand Life Cycle Management Centre, New Zealand, [*] Corresponding author. E-mail: *S.McLaren@massey.ac.nz*

Biochar is carbonised biomass; the focus of this study is biochar obtained from sustainable sources and sequestered in soils to sustainably enhance their agricultural and environmental value under present and future management. Biochar has attracted international attention as a carbon sequestration strategy since it can take hundreds or thousands of years to decompose. Moreover, biochar offers opportunities in the energy, soil management, and end-of-life biomass (ELB) recycling sectors.

The quantity of ELB feedstocks in New Zealand that could potentially be used to produce biochar was assessed. In a highly optimistic scenario in which 80% of the available ELB is sourced to make biochar, over 1 million tonnes of CO_2 could potentially be sequestered every year. This translates to about 1.5% of NZ's total greenhouse gas emissions (based on 2009 data). Although this percentage is small, the relative contribution of using biochar on net greenhouse gas emissions may be much more significant when considering products in particular economic sectors from a life cycle perspective e.g., in determining the carbon footprint of agricultural products.

Although a number of biochar systems have been evaluated from a life cycle perspective, only two studies (Roberts et al. 2010; Hammond et al., 2010) and one life cycle inventory analysis (Kameyama et al., 2010) seem to have followed LCA methodology – and they all focus on just one impact category (climate change). Also, one LCA study of ethanol produced from a hectare of corn includes stover-derived biochar in the analysis (Kauffman et al., 2011). Using an LCA approach, the methodological issues concern the goal, scope and decision-context of the study; functional unit; multiple functions; system boundaries and allocation; choice of impact categories; indirect consequences; and reference scenario with which the biochar system is compared. At the forefront of these variables, it is not clear when and how to conduct attributional versus consequential LCA, and so results can vary considerably.

Therefore, particularly when considering future policy options to encourage or discourage production and use of biochar, it is important to carefully consider the different variables and their influence on the final results of an LCA study of biochar. This paper presents the results of a life cycle study on three different future management options for the woody ELB from apple orchards in the Hawke's Bay region in NZ undertaken with the goal of informing stakeholders and policy makers on the best use of biomass to mitigate climate change. Three different scenarios are compared: i) reference scenario, in which the woody ELB is mulched and left on orchard soils; ii) energy scenario, in which the ELB is used for energy generation; and iii) biochar scenario, in which the ELB is used for biochar production and application into the same area. The results show that the fuller trade-offs associated with alternative end uses of biomass need to be explored using a more complete system expansion perspective and representation of alternatives.

References

Hammond, J., Shackley, S., Sohi, S., Brownsort, P., 2011. Prospective life cycle carbon abatement for pyrolysis biochar systems in the UK. Energy Policy, 39, 2646-2655.

Kameyama, K., Shinogi, Y., Miyamoto, T., Agarie, K., 2010. Estimation of net carbon sequestration potential with farmland application of bagasse charcoal: life cycle inventory analysis through a pilot sugarcane bagasse carbonisation plant. Aust. J Soil Res.,48, 586-592.

Kauffman, N., Hayes, D., Brown, B., 2011. A life cycle assessment of advanced biofuel production from a hectare of corn. Fuel, 90,3306-3314.

Roberts, K.G., Gloy, B.A., Joseph, S., Scott, N.R., & Lehmann, J., 2010. Life Cycle Assessment of biochar systems: estimating the energetic, economic, and climate change potential. Environ. Sci. Tech., 44, 827-833.

56. Soil quality aspects in food LCA

Taija Sinkko*, Merja Saarinen

MTT Agrifood Research Finland, * Corresponding author. E-mail: *taija.sinkko@mtt.fi*

Soil quality is a crucial issue in food production and consumption from a sustainability point of view. However, there is no commonly accepted soil quality impact indicator in food LCA. Soil quality can be defined as "The capacity of soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health" (Doran and Parkin 1994). Soil properties are generally divided into three categories; biological, chemical and physical quality characters. The methodology of soil quality assessment should adequately reflect all the main soil characters and/or their interface.

The aim of this on-going study is to 1) review suggested soil quality indicators for LCA, 2) assess their practical applicability in process-based food LCA, 3) adapt or further develop the methodology for soil quality assessment in process-based food LCA, and 4) (preliminarily) assess soil quality of case-products. Through a literature survey we compiled some methods suggested for the assessment of land use impacts on soil quality in LCA (e.g. Muys and García Quijano 2002, Achten et al. 2008, Wagendorp et al. 2006). The methods include numerous indicators, and lack of data is one of the major challenges for practical application. Therefore, Milà i Canals et al. (2007) proposed a sole indicator, soil organic matter (SOM), to assess land use impacts on soil quality. However, SOM of arable land changes rather slowly from an annual measurements point of view. This reduces its flexibility as an indicator, restricting feedback for improvements. According to the ISO 14040 series, LCA impact category implies how production of a certain product affects nature; i.e. the reference situation is a natural stage. The suggested methods in the literature represent this approach, but the approach of sustainable development challenges it. Actually, the field should remain a field, and farming practices should maintain or improve soil quality so that farming can continue (within ecosystem boundaries). A concept of ecosystem service takes this aspect into account as it is based on the concept of sustainable use of natural resources. It was decided to take this as the theoretical basis for developing a soil quality indicator in this study.

Based on reviewing and assessing methods we concluded that a new methodology (incl. theoretical background) is needed. We initiated an interaction between specialists from different branches of soil science to establish which aspects should be included in the soil quality indicator to ensure that it is amenable to follow-up measurements and, for example, is sensitive to differences between organic and conventional cultivation methods.

The study is included in the project "Towards Sustainable Food Choices – Consumer Information on Nutrition and Environmental Impacts of Food in the Context of Sustainability" funded by the Finnish Ministry of Agriculture and Forestry and MTT Agrifood Research Finland.

References

Achten, W.M.J., Mathjis, E., Muys, B., 2008. Proposing a life cycle land use impact calculation methodology. Proc. 6th Int. Conf. on LCA in Agri-Food Sector, Zurich, pp. 22-33.

Doran, J.W., Parkin, T.B., 1994. Defining and assessing soil quality. In: J.W. Doran et al. (eds.) Defining Soil Quality for a Sustainable Environment. Soil Science Society of America, Madison, WI, Special Publication 35, pp. 3-22.

Milà i Canals, L., Romanyà, J., Cowell, S.J., 2007. Method for assessing impacts on life support functions (LFS) related to the use of "fertile land" in Life Cycle Assessment (LCA). J Clean Prod 15, 1426-1440.

Muys, B., García Quijano, J., 2002. A new method for Land Use Impact Assessment in LCA based on ecosystem exergy concept. Internal report.

Wagendorp, T., Gulinck, H., Coppin, P., Muys, B., 2006. Land use impact evaluation in life cycle assessment based on ecosystem thermodynamics. Energy 31, 112-125.

57. The influence of methodology on the water footprint of selected UK produced and consumed products

Rob Lillywhite[1], Ashok Chapagain[2], Tim Hess[3]

[1] University of Warwick, Wellesbourne, Warwick, CV35 6JQ, UK, [2] WWF-UK, Goldaming, UK, [3] Cranfield University, Bedford, UK, [*] Corresponding author. E-mail: *robert.lillywhite@warwick.ac.uk*

Water footprinting is still evolving as a discipline and various methodologies and approaches exist to quantify the volumes of water withdrawn, used and consumed for food production and to assess any associated environmental impacts. The choice of methodology will influence the results and interpretation, a situation which has resulted in considerable variation in reported estimates on water use and its impacts for different agricultural and food products. Mindful of this, the UK Government commissioned research to explore the different approaches using selected UK produced and consumed products (potato, lamb, milk, strawberry, sugar and winter wheat) as case studies.

The study used three approaches:

1. The water footprint accounting framework developed by The Water Footprint Network (Hoekstra et al., 2011). This approach estimates the volume of green and blue water consumed during the different stages of production (evapotranspiration, irrigation, crop protection, livestock drinking, cleaning etc.)
2. The stress-weighted water footprint (Ridoutt and Pfister, 2010). This approach estimates the volume of blue water consumed during production and converts it to an assessment of local water stress using a water stress index (WSI) (Pfister at al., 2009)
3. The normalised water footprint, H_2Oe (Ridoutt et al., 2012) This is a development of the stress-weighted approach in which the stress-weighted value is divided by the global average WSI to give an assessment of water consumption relative to a global average.

The results show that the considerable influence that methodology has on the estimated values. The volumetric results are not directly comparable to the stress-weighted and normalised results since they are based on different criteria, but they are all commonly referred to as water footprints and clearly illustrate the potential for confusion that can arise in this discipline. Table 1 shows the results for UK and Israeli potato and allows the following interpretation.

In terms of volumetric water consumption, Israeli potatoes use one and a half times more water than UK production. However, where the impact of that water consumption is considered, Israeli potatoes can have a ten-fold greater impact on local water resources and a nineteen-fold greater impact at a global scale. The key to understanding and using these results is the omission of green water from the stress-weighed and normalised approaches which principally reflects the difference in evapotranspiration (and therefore climate) but also of irrigation practice, between the two countries.

The volumetric water footprint has been very successful in raising awareness of the use of water and is invaluable for water auditing purposes but we conclude that it has limited value for determining the local water stress of globally sourced products. A more balanced approach, especially within the LCA framework, is possible using the normalised water footprint alongside other environmental indicators, such as eutrophication and acidification. This will provide a more consistent and robust approach for environmental and sustainability studies.

References

Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M. (2011) The water footprint assessment manual: Setting the global standard, Earthscan, London, UK.

Pfister ,S., Koehler, A., Hellweg, S. (2009) Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Tech 43, 4098-4104.

Ridoutt, B.G., Pfister, S. 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environ. Change 20, 113-120.

Ridoutt B,G., Sanguansri, P., Freer, M., Harper, G.S. (2012) Water footprint of livestock: comparison of six geographically defined beef production systems. International Journal of LCA 17: 165-175.

Table 1. The influence of methodology on the water footprint of UK and Israeli potato

Country	Volumetric (litres/kg)	Stress-weighted	Normalised (H_2Oe/kg)
UK	107	10	9
Israel	147	103	171

58. Water footprints of wheat and maize: comparison between China's main breadbasket basins

Jing Huang[1,2], Bradley G Ridoutt[2], Feng Huang[3], Fu Chen[1,*]

[1] China Agricultural University, College of Agriculture and Biotechnology, Beijing 100193, China, [2] Commonwealth Scientific and Industrial Research Organisation (CSIRO), Sustainable Agriculture National Research Flagship, Private Bag 10, Clayton South, Victoria 3169, Australia, [3] China Agricultural University, College of Resource and Environmental Sciences, Beijing 100193, China, * Corresponding author. E-mail: *chenfu@cau.edu.cn*

As the most populous country in the world, China's food security has become an issue of broad concern. A critical factor which threatens food production is water scarcity. In recent decades, the water-scarce northern part of China has become the most important agricultural production area, supported in small part by water transfers from the South. Substantial volumes of food are being transferred from the water-scarce north to the water-rich south. In the context of China's food security, this complex arrangement has been the focus of much debate. However, science-based evidence to support an environmentally-sustainable increase in China's food production is rarely found in the literature.

This study compares the water footprints of cereal production (wheat and maize) in China's main breadbasket basins. Water footprints were calculated using an LCA-based water footprinting method (Ridoutt and Pfister, 2010) which uses a Water Stress Index (WSI; Pfister et al., 2009) to express the environmental relevance of water use. The water footprints are presented in the units H_2Oe (equivalent), where 1 L H_2Oe represents 1 L of consumptive freshwater use at the global average WSI.

Wheat grown in the Huang and Hai basins had much higher water footprints (1,262 L H_2Oe kg^{-1}) compared to wheat grown in the Chang basin (31 L H_2Oe kg^{-1}). The water footprints of maize grown in the Huang, Huai and Hai basins (515 L H_2Oe kg^{-1}) were also much higher than maize grown in the Chang and Songliao basins (35 and 44 L H_2Oe kg^{-1} respectively).

These results demonstrate a huge spatial differentiation of water use for cereal production in China's main breadbasket basins. It is suggested that the variability in crop water footprint between production systems should be taken into consideration in strategic decisions related to China's food production. National-scale cropping pattern adjustment and technological upgrade at the basin level are considered as important interventions to alleviate water stress from agriculture.

References

Ridoutt, B.G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environ. Chang. 20, 113-120.

Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Tech. 43, 4098-4104.

59. Environmental impact of green beans: the relevance of water use

Thomas Kägi*, Silvan Rüttimann, Fredy Dinkel

Carbotech AG, Switzerland, * Corresponding author. E-mail: *t.kaegi@carbotech.ch*

About 80% of the world fresh water consumption is attributed to agriculture. Excessive water use can lead to water stress, one of the major environmental challenges in the future. As fresh water is getting scarcer due to climate change and due to increase irrigation amounts, the role of environmental impact due to water scarcity is growing in life cycle impact assessments.

The goal of this study (Kaegi et al. 2011) was to compare different green bean production and processing scenarios and to derive the contribution of water use to the total environmental impact. The functional unit was 1kg of green beans, ready to eat. All relevant life cycle phases were considered including cultivation of beans (field processes, fertiliser and chemical inputs, irrigation, direct field emissions), transportation, further processing (such as washing and then drying or canning or freezing) and cooking of the beans. Data for bean production was based on Lattauschke (2002) and data for direct field emissions were derived from Nemecek & Kägi (2007). The ecoinvent inventory V2.2 database (Swiss Centre for LCA 2009) was used for other secondary data (fertiliser production, transportation and other) and emission factors. For valuation of the different environmental impacts (such as global warming-, acidification-, eutrophication-, ozone depletion potential, ecological and human toxicity etc.) the ecological scarcity method (Frischknecht 2007) was used including regional water scarcity factors for the water use.

The high water scarcity in the Spanish region where the beans are grown leads to a large environmental impact. Beans grown in greenhouses show a similar high environmental impact due to fossil energy use for heating. Fresh beans from Egypt are flown to Switzerland which explains the high contribution from transportation. Further processing such as canning, freezing, drying lead to higher environmental impacts compared to fresh beans. However, when compared to irrigated beans from Spain, this seems to be irrelevant.

If the whole life cycle of ready to eat beans is considered the water impact can play a very important role. It contributes up to 85% of the total environmental impact in the case of beans from Spain. The implemented water impact in nowadays life cycle assessment methods covers a crucial and important environmental topic and helps to improve environmental consulting in the field of agriculture and water consumption.

References

Dinkel et al. (2006). Calculations for „Der kluge Einkaufswagen“ Unterrichtseinheit zu den Themen Umwelt, Konsum, Ökobilanzen. Bundesamt für Umwelt (BAFU)

Frischknecht et al. (2008) Frischknecht R., Steiner R., Jungbluth N. (2008) Methode der ökologischen Knappheit - Ökofaktoren 2006. Bundesamt für Umwelt (BAFU)/ÖBU.

Lattauschke, G, 2002. Hinweise zum umweltgerechten Anbau von Freilandgemüse im Freistaat Sachsen – Buschbohnen. Sächsische Landesanstalt für Landwirtschaft. http://www.landwirtschaft.sachsen.de/LFL

Nemecek T. & Kägi T. (2007). Life Cycle Inventories of Swiss and European Agricultural Production Systems. Final report ecoinvent V2.0 No. 15a. Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for Life Cycle Inventories, Zurich and Dubendorf, CH, retrieved from www.ecoinvent.ch

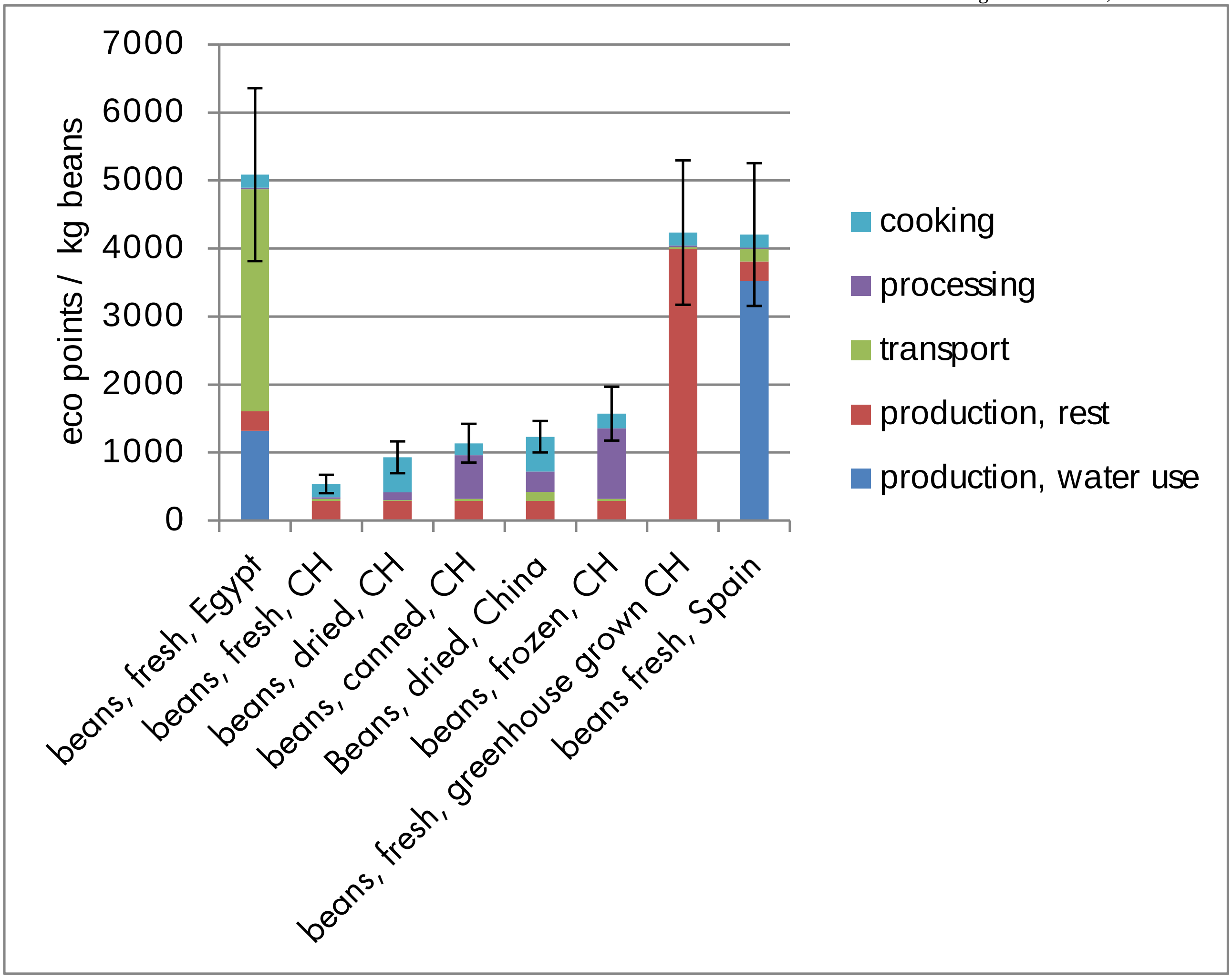


Figure 1. Environmental impact per kg beans, ready to eat.

60. Hands-on water footprinting: putting the assessment of agricultural fresh water use into practice

Daniel Thylmann*, Annette Köhler, Sabine Deimling

PE INTERNATIONAL AG, Leinfelden-Echterdingen, Germany, * Corresponding author. E-mail: *d.thylmann@pe-international.com*

Freshwater scarcity is now recognised to become one of the main environmental issues in the future. However, the consideration of fresh water consumption in life cycle modelling is still very young. The fact that no standardised method has evolved so far has led to a hesitant reaction of stakeholders towards the generally rising interest in the subject.

Agriculture contributes 80% to worldwide freshwater use (WWAP 2009), thus its key importance in sustainable water management practices is undoubted. But agricultural systems are particularly difficult to assess in an LCA framework, water use being no exception. PE INTERNATIONAL has implemented the latest methods developed by Pfister et al. (2009, 2011) in their balancing approaches and is now conducting complete water footprint assessments with consideration of regional water availability. The LCA software and database *GaBi* was updated (GaBi 2012) and contains complete and consistent water inventory data now, allowing assessment of fresh water use and consumption in an LCA framework using software solutions.

This paper describes how assessments of fresh water use in agricultural products can be put into practice using the latest software solutions. More than 100 agricultural products contained in the GaBi 5 database were updated to contain data on water use and consumption. Important lessons learned from this update process are presented. Results from a case study on cotton cultivation in the US are shown, including differentiation of water availability in four different cultivation regions in the US (Fig. 1). These results are considered to be a representative example for a variety of other agricultural products as well.

The experiences gained underline the relevance of agricultural processes, especially of irrigation, for the water footprint results of the complete value chain. It can also be seen that regional water availability needs to be considered in order to derive meaningful conclusions from water use assessments. Another important aspect is the necessity of consistent inventory data on water use and the difficulties to obtain these.

Water is an important aspect when considering the environmental impact of a product and should not be ignored any longer in LCAs. With the methods on hand, it is now possible to account for fresh water consumption in a LCA framework, also for agricultural products. However, large challenges lay ahead. Further advancements are needed in the development of a harmonised and standardised method, especially for the impact assessment phase. Finally, LCA is not meant to be a self-contained art. Not until companies, policy makers, civil society and private people understand the necessity of a responsible use of fresh water resources, the final goal of water footprinting will be reached.

References

GaBi, 2012. GaBi 5 - Software and Databases for Life-Cycle-Engineering, PE INTERNATIONAL AG, Copyright, TM. Stuttgart, Echterdingen. 1992 - 2012.

Pfister, S.; Koehler, A.; Hellweg, S., 2009. Assessing the environmental impact of freshwater consumption in LCA. Environ Sci Technol 43(11), 4098-4104.

Pfister, S; Bayer, P; Koehler, A; Hellweg, S, 2011. Environmental impacts of water use in global crop production: hotspots and trade-offs with land use. Environmental Science & Technology 45, 5761-5768.

WWAP, World Water Assessment Programme, 2009. The United Nations World Water Development Report 3: Water in a Changing World, UNESCO, Paris and Earthscan, London.

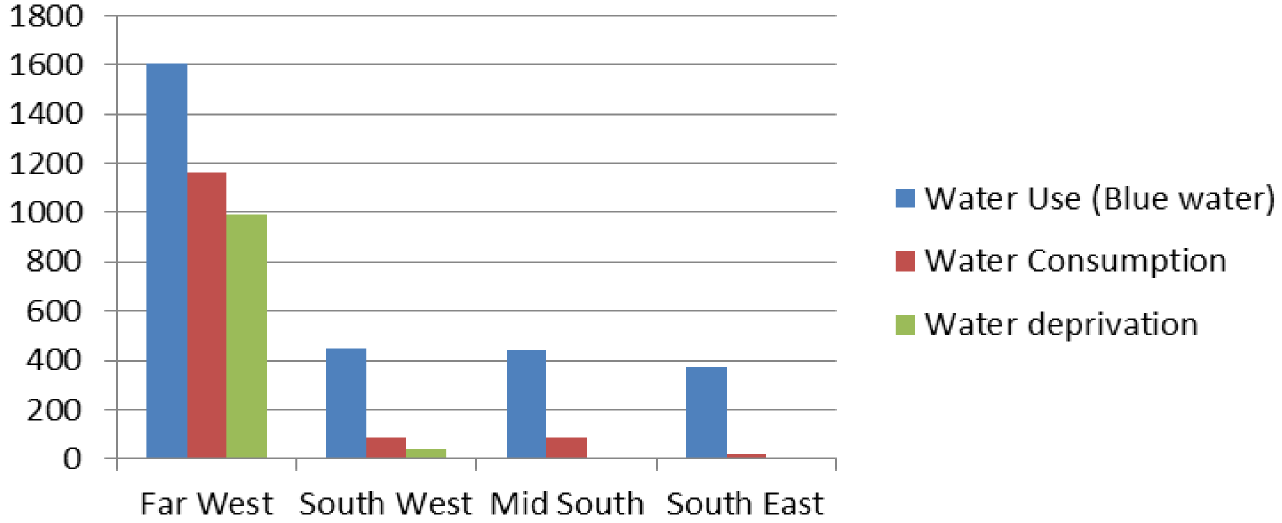


Figure 1. Example of a complete water footprint (in litres) of a processed agricultural product (cotton T-shirt) considering regional water availability

61. A method for estimating water use in food supply chains: liquid milk as an example

Stefan Hörtenhuber[1,2,*], Rainer Weißhaidinger[1], Thomas Lindenthal[1,3],Werner Zollitsch[2]

[1] Research Institute of Organic Agriculture (FiBL) Austria, [2] University of Natural Resources and Life Sciences Vienna, Department of Sustainable Agricultural Systems, [3] University of Natural Resources and Life Sciences Vienna, Centre for Global Change and Sustainability, [*] Corresponding author. E-mail: *stefan.hoertenhuber@fibl.org*

A water-use (WU) method, based on Hoekstra et al. (2009), was modified and applied to Austrian agriculture, particularly livestock production. To meet the requirements of a life cycle assessment (LCA) approach, system boundaries include additional demand for water, e.g. from dairies, trade, supply of production inputs (mineral fertilisers) or for industrial processing of feedstuffs. The WU method accounts for effective or so called 'blue' water demand which is needed for irrigation, cleaning, livestock's drinking water or for cooling systems in dairies. Precipitation water which is evapotranspirated is summarised within the 'green' water, including a potential loss of precipitation in the case of preceding clearing of tropical forests. The WU method provides regionally differentiated water demand for effective evapotranspiration per kg yield for roughage, concentrates (grains, grain legumes, oilseeds or co-products from oil mills and distilleries) or bedding material, which is not only based on precipitation inputs but also reflects climatic and soil conditions, groundwater recharge and run-off. 'Grey' water partly integrates an eutrophication potential into the water footprint. For derivation of the grey water (i.e. dilution below nitrate limits 45 mg NO_3 per litre in drinking water), a detailed nitrogen (N) cycle model was used, including various N-inputs and outputs from agricultural production and its upstream and downstream processes. Co-products (beef from cull cows and calves) and water required for the rearing phase of dairy cow were also considered (see Hörtenhuber et al., 2010). Generally, results for livestock products' WU mainly depend on type (i.e. composition) and quantity of the diet needed to produce one unit of product (kg milk).

The result for an alpine, grassland-based production system shows an overall water demand of about 940 litres per kg liquid milk at the supermarket (Fig. 1). This WU result agrees with findings from previous studies for milk, e.g. 800 and 990 litres of water demand (global scale) as reported by Chapagain and Hoekstra (2003) and Hoekstra and Chapagain (2006), respectively. However, some differences between these sources and our study are obvious, such as (1) a higher proportion of grey water and (2) a smaller proportion of green water in our result; (3) additional processes were included, which require water along the entire supply chain. A potential for the reduction of water demand was identified particularly for 'grey water' by implementing the following measures: (i) greening and catch crops instead of bare fallow, (ii) application of manure/fertiliser according to the requirement of crops at the optimum point of time, (iii) decreasing the input of external production factors (mineral fertilisers) or (iv) preferring organic over mineral fertilisers. Because of the limited water supply in many parts of the world, comprehensive WU or water footprint methods need to be developed and integrated into sustainability assessment schemes for agricultural products.

References

Chapagain, A.K., Hoekstra, A.Y., 2003. Virtual water flows between nations in relation to trade in livestock and livestock products. Value of Water Research Report Series No. 13, UNESCO-IHE.

Hoekstra, A.Y., Chapagain, A.K., 2006. Water footprints of nations: Water use by people as a function of their consumption pattern. Water Resources Management. 21(1), 35-48.

Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2009. Water footprint manual: State of the art 2009. Water Footprint Network, Enschede, Niederlande.

Hörtenhuber, S., Lindenthal, T., Amon, B., Markut, T., Kirner, L., Zollitsch, W., 2010. Greenhouse gas emissions from selected Austrian dairy production systems – model calculations considering the effects of land use change. Renewable Agriculture and Food Systems 25(4), 316-329.

Hörtenhuber, S., Lindenthal, T., Schmid, E., 2011: Water Footprint – Ein Beitrag zur Nachhaltigkeitsbewertung am Beispiel der Milcherzeugung, in: Proceedings of the 66th ALVA Conference, May 23-24 2011. Graz, Austria, pp. 87-89.

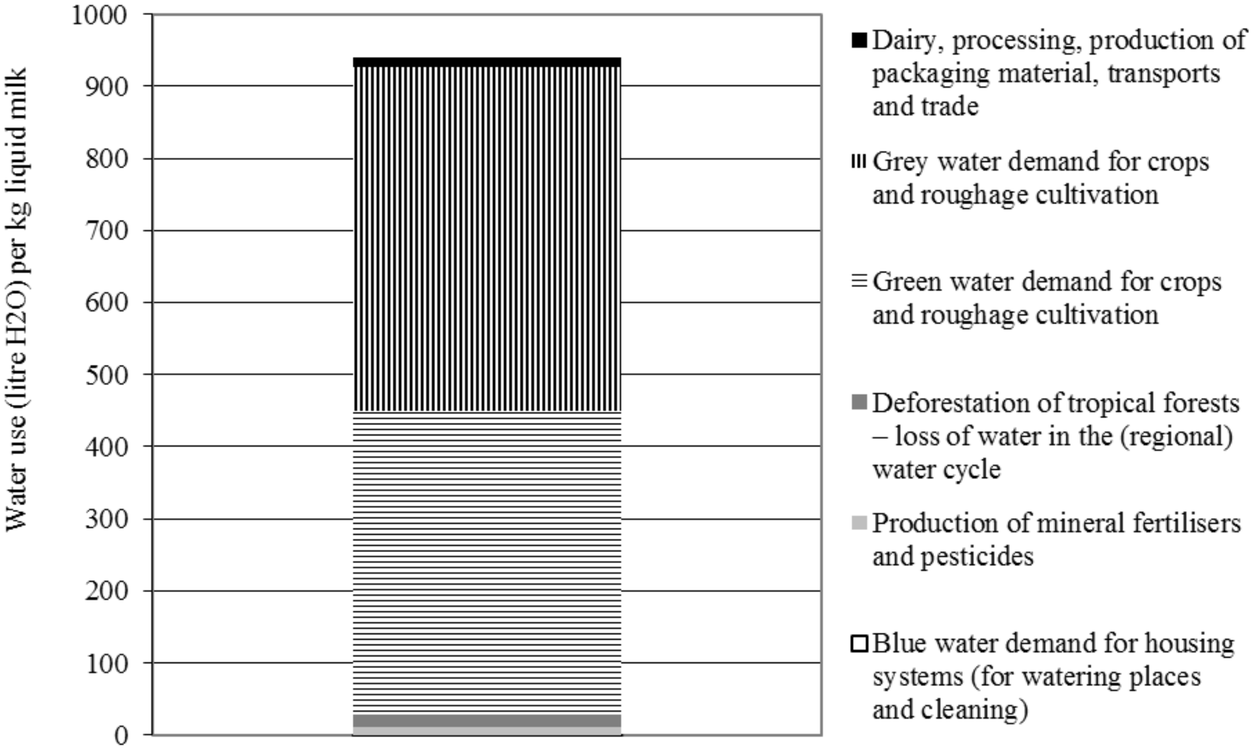


Figure 1. Results for a comprehensive evaluation of water use (litres H_2O) of 1 kg liquid milk from an alpine, grassland-based production system at the retailer level.

62. Irrigation systems used in water scarce locations: LCA of three contrasted scenarios for watermelon growing

Ludivine Pradeleix[1,*], Sami Bouarfa[1], Philippe Roux[2], M. Gueguen[2], V. Bellon-Maurel[2]

[1] Cemagref/Irstea, Research Unit: G-Eau, Water management, stakeholders and uses, 361 rue JF Breton, 34196 Montpellier, France, [2] Cemagref/Irstea, Research Unit: Information & technologies for agro-processes, 361 rue JF Breton, 34196 Montpellier, France, [*] Corresponding author. E-mail: *ludivine.pradeleix@irstea.fr*

The agricultural productivity in arid and semi-arid regions greatly depends upon the access to irrigation. But irrigated areas are often charged with being water and chemical inputs intensive and consequently damaging the environment. In return, irrigated systems allow more diversified and intensified agricultural production. This is even more important when farmers shift from surface irrigation to drip irrigation that enable covered crops cultivation and use of mulch.

Environmental impacts arising from this twofold intensification process have to be addressed. Most of the LCA studies and guidelines have targeted temperate locations (Nemecek & Kägi, 2007) and this study contribute to adapt the method to southern contexts where the great variability of crop management practices leads to increased uncertainty (Basset-Mens, et al., 2006).

Based on a case study located in the Tunisian central Irrigated Plain of Kairouan, we choose to study the impacts per kg and per ha of three contrasted cropping systems of watermelon growing. They were identified among a typology of eight systems in total. The least intensive cropping system relies on surface irrigation rather than drip irrigation for the middle and high inputs intensive cropping systems. The most intensive cropping system combines plastic mulches and row covers. Because it was not possible to measure irrigation water volumes, they were modeled with PILOTE, a crop-soil model (Mailhol, 2004).

The most impacting cropping system per kg is the middle input intensive for almost all the impacts categories apart from those related to toxicity and ecotoxicity. Then comes the high input intensive cropping system whose impacts are balanced by the relative high yield obtained. The low input intensive system shows the smallest impacts either per kg or ha.

The drip irrigation equipment at field scale is drawn on Fig. 1, and Fig. 2 displays the environmental impacts of 1 ha of drip irrigated middle input intensive watermelon growing. The most impacting field operation are fertilisation then irrigation and lastly soil preparation. **More specific results of impacts caused by the drip irrigation device show the relatively high impact of energy for water pumping and thus suggest the related improvements be done.**

References

Basset Mens, C., Van der Werf, H.M.G., Durand P., Leterme P., 2006. Implications of uncertainty and variability in the life cycle assessment of pig production systems. International Journal of Life Cycle Assessment, 11(5), 298-304.

Mailhol, J. 2004. Analysis of irrigation systems and irrigation strategies for durum wheat in Tunisia. Agricultural Water Management, 70(1), 19-37.

Nemecek, T., Kägi, T. 2007. Life Cycle Inventories of Agricultural Production Systems, (15).

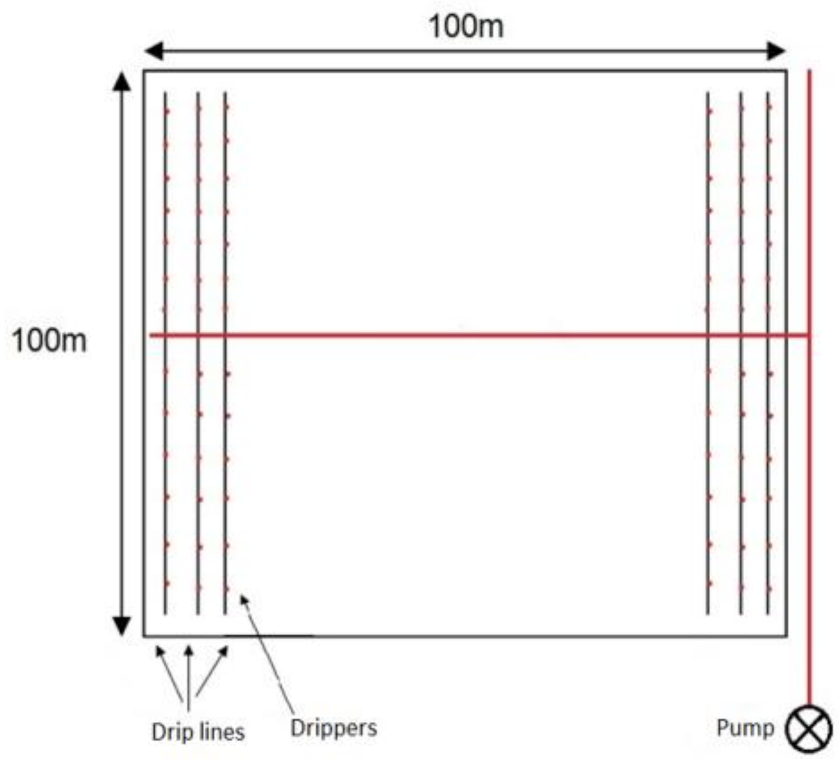


Figure 1. Field-scale drip irrigation system: irrigation elements for mid- and high-input intensive cropping systems.

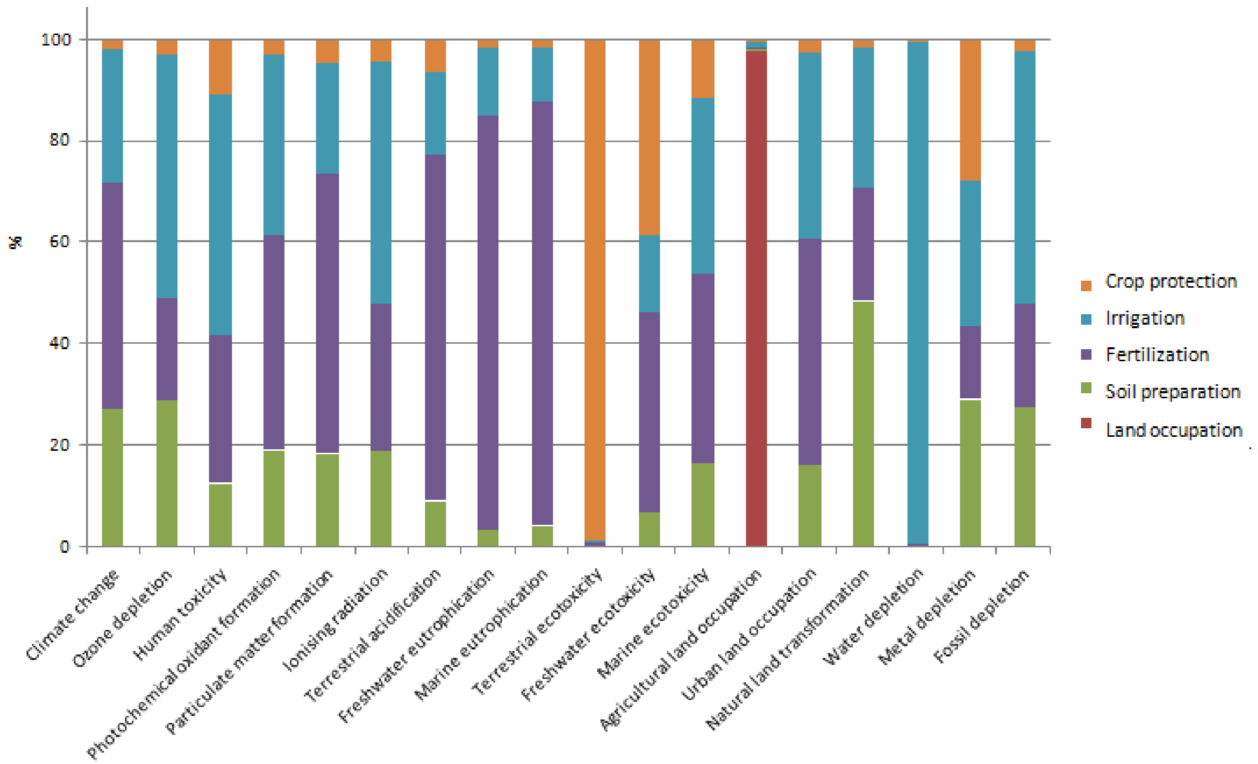


Figure 2. Characterisation of environmental impacts of the cultivation of 1 hectare of mid-input intensive watermelon, Recipe Midpoint (H).

63. Green, blue, and grey water use of Canadian wheat and maize

Brian McConkey[1,*], Chao Chen[1], Dominique Maxime[2], Suren Kulshrestha[3], Xavier Vergé[4], Ray Desjardins[5], Devon Worth[5], Jim Dyer[6]

[1] Agriculture and Agri-Food Canada, Swift Current, SK, Canada, [2] Centre interuniversitaire de recherche sur le cycle de vie des produits, procédés et services, Montréal, QC, Canada, [3] Department: Bioresource Policy, Business and Economics, University of Saskatchewan, Saskatoon, SK, Canada, [4] Consultant, Ottawa, ON, Canada, [5] Agriculture and Agri-Food Canada, Ottawa, ON, Canada, [6] Consultant, Cambridge, ON, Canada, [*] Corresponding author. E-mail: *brian.mcconkey@agr.gc.ca*

Wheat (*Triticum aestivum* L.) and maize (*Zea mays* L.) are the major cereal crops of Canada and their use for producing bioethanol has heightened interest in their environmental footprint. In semiarid climates of western Canada, both green and blue water supply are a constant concern. Rainfed agriculture periodically requires "drought disaster" financial support from government. Irrigation is limited as water in major streams is fully allocated and water restrictions can occur during drought years. In contrast, water excesses are more often a concern than deficiencies in the more humid climates of central and eastern Canada.

We used the DSSAT model v4 to estimate 30-yr average green (evapotranspiration of *in situ* precipitation) and blue (from water withdrawals) use and yield across a range of important cereal production situations (Table 1). The model was validated with field data. Estimated blue water use embodied in inputs (pesticide, fossil fuel, machinery manufacture, fertiliser) was about 1 m^3 (t grain)$^{-1}$ based on withdrawals. Grey water use was calculated from P losses from the literature. The emerging Canadian guideline for total P in water to prevent eutrophication (0.02 mg L^{-1}) is 500 times lower than drinking water standard for N (10 mg L^{-1} as N-NO_3). Since total N losses are about 2 to 10 times greater than those of P, roughly 50-250 times more water is needed to dilute the P than the N (divide by about 100 to compare with N-based studies).

The total cereal water use in Canada was dominated by grey water (Table 1). Grey water use is higher in humid climates due to larger P loss in runoff and artificial drainage. Grey water use usually exceeds any precipitation surplus and this explains why P is the major environmental concern since there is often insufficient water to dilute the P to a desired concentration in cropland dominated watersheds. Total (green+blue+grey) water use decreased moving from humid to more arid climate. In contrast and as expected, the blue + green water use t^{-1} tended to increase moving to drier climate (e.g. London to Winnipeg, Lethbridge or Winnipeg to Swift Current). However, the relationship between climate and green+blue water use was less clear than total water use, probably due to confounding effects of other weather factors (temperature, rain timing, and sunlight) on production. As expected, maize was more water efficient than wheat under rainfed conditions but efficiencies between these cereals was similar under irrigation. In the most semiarid areas, summer fallow is still widely practiced where a crop is purposefully not grown in one year so as to use the soil-stored precipitation from that year to reduce drought risk for the crop grown the next year. A systems level calculation that considers the fallow year as an inseparable part of wheat production on summer fallow has the highest cereal water use t^{-1} in Canada while the simplistic non-systems analysis that considers the crop year only would indicate that summer fallow actually decreases water use t^{-1}; we believe only the systems-level analysis is valid. Excluding summer fallow production, green +blue wheat water use t^{-1} did not vary much across the diverse climates or with and without irrigation.

If lower green+blue+grey water use t^{-1} were used as the sole indicator of greater water security for cereal production in Canada, it would result in the nonsensical conclusion that production should be increased in the region with regular water shortages and decreased in regions with abundant water. This would exacerbate the impact of droughts in water-stressed regions to the whole of Canada. Inter-regional comparisons of green+blue water use were more difficult to interpret although there was an inconsistent trend of higher water use t^{-1} as climate became drier. Intra-regional comparisons of water use t^{-1} help identify crops and production methods that would, if selected, reduce water requirements for cereal production and lower natural resource requirements for biofuels produced from those cereals.

Table 1. Land use and green, blue, and grey water use for important cereal production situations in Canada across a range of climates.

		Land Use	Water Use (m^3 t^{-1})				
Crop	**Production Method**	**(ha t^{-1})**	**Green**	**Blue**	**Grey***	**Green + Blue**	**Green +Blue + Grey**
Semiarid (Swift Current: P=352 mm, PET=931 mm, MAT=3.9°C)**							
Spring Wheat	Rainfed on stubble	0.53	968	1	5260	969	6230
Spring Wheat	Rainfed on summerfal-low excluding fallow area and evapotranspiration	0.38	804	1	3850	805	4650
Spring Wheat	Rainfed on summerfal-low including fallow area and evapotranspiration	0.77	3850	1	15400	3850	19200
Semiarid (Lethbridge: P =385mm, PE=917 mm, MAT=5.8°C)							
Maize	Irrigated	0.17	407	356	2540	763	3310
Spring Wheat	Irrigated	0.22	400	378	3330	778	4110
Spring Wheat	Rainfed on stubble	0.45	818	1	4550	819	5370
Subhumid (Winnipeg: P =514 mm, PE=716 mm, MAT=5.0°C)							
Maize	Rainfed	0.19	821	1	4720	822	5540
Spring Wheat	Rainfed	0.38	908	1	9620	909	10500
Humid (London, Ontario: P =987 mm, PE=662 mm, MAT=7.5°C)							
Maize	Rainfed	0.11	449	1	6842	450	7290
Spring Wheat	Rainfed	0.23	707	1	14800	708	15500
Humid (Charlottetown: P =1173 mm, PET=512 mm, MAT=5.3°C)							
Spring Wheat	Rainfed	0.34	997	1	18600	998	19600

*based on desired P concentration to meet Canadian environmental objectives, divide by about 100 to compare with other studies based on drinking water NO_3 objectives.

**P=annual precipitation, PET=annual potential evapotranspiration, MAT=mean annual temperature

64. Virtual water of sugar production in Spain

J. Soler-Rovira*, P. Gómez de la Calle, J.M. Arroyo-Sanz

Escuela Universitaria de Ingeniería Técnica Agrícola, Universidad Politécnica de Madrid, Ciudad Universitaria, s/n 28040 Madrid, Spain, * Corresponding author. E-mail: *jose.soler@upm.es*

Food production and consumption contributes to water use and abstraction, mainly during the phase of cultivation. Water footprint of agricultural products is made up of blue, green and grey water. Green water is the rainfall water evapotranspirated from cultivated soils. Blue water is the fresh water used in irrigation, taken from water bodies that is used and not returned. Grey water is the volume of water required to dilute pollutants to such extent that the water quality reaches acceptable standards. Irrigated sugar beet crop in Spain accounts for 93% of the 70,000 ha of cultivated surface. Beet is the main source of sugar and every Spanish inhabitant consumes 5.5 kg per year, although 50% is imported.

The aim of this work is to evaluate the virtual water content of sugar beet crop and industrial sugar in Spain. The main provinces of sugar beet cultivation were considered. Virtual water content of the beet crop was calculated taking into account the root and sugar yield and the evaporative and non-evaporative water used for crop production. The water consumed in evaporation was made up of green and blue water. The green one was computed from rainfall and crop evapotranspiration plus soil evaporation computing a soil water balance with site specific soil data, climatic data and crop growth cycle. Reference evapotranspiration was computed with both Penman-Monteith and Hargreaves method. Blue water was obtained from soil water balance as the difference of crop evapotranspiration and rainfall and the efficiency of the irrigation system (gravity or sprinkler). Seedling emergence water applications were also accounted in sprinkler irrigated crops. Grey water was considered as the polluted water, and was calculated with the site specific fertilisation rate of the crop, estimated nitrate leaching and water quality standards.

The estimated water footprint per surface unit in Burgos and Valladolid provinces is shown in Figure 1. The volume of water is higher than 1,000 L per m^2. Total water footprint of Valladolid province is greater than that of Burgos. Blue water (irrigation requirements) is higher in Valladolid because the increased ETo values and the decreased rainfall in that province. However, green water is lower due to the less rainfall. As nitrogen fertilisation rates are higher in Valladolid than in Burgos, grey water is also higher. The water footprint is larger for gravity irrigation systems than for sprinkler ones, because their lower water application efficiency.

Water footprint estimated per kg of sugar is more than 800 L (Fig. 2). The most important component of sugar water footprint is the blue one, because the sugar beet is sown in spring and the maximum canopy development and water transpiration is during summer, when ETo is high and the rainfall is low. Green water is less than 35% of total water footprint, and it is lower in provinces with decreased rainfall values. Grey water account for 100-200 L per kg of sugar, and it depends on nitrogen fertilisation rates.

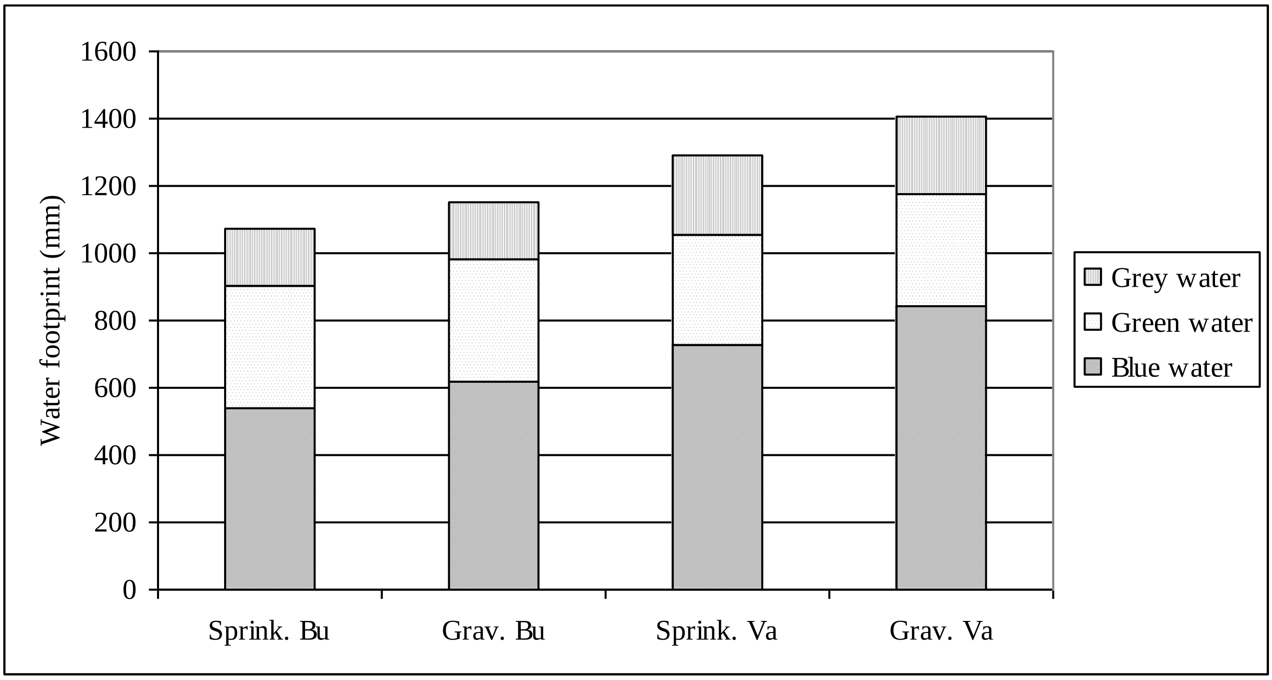


Figure 1. Estimated water footprint of sugar beet crop in two Spanish provinces, Burgos (Bu) and Valladolid (Va), with two different irrigation systems, sprinkler (sprink.) and gravity (grav.).

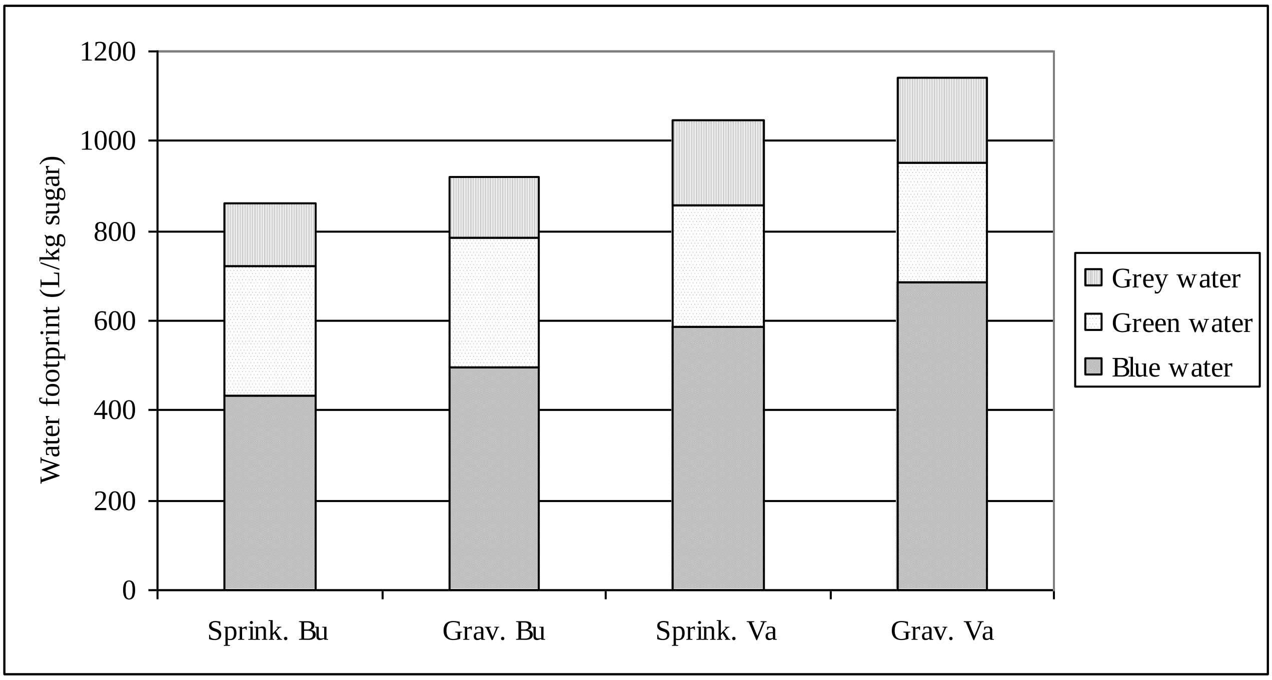


Figure 2. Estimated water footprint of sugar in two Spanish provinces, Burgos (Bu) and Valladolid (Va), with two different irrigation systems, sprinkler (sprink.) and gravity (grav.).

65. Modelling sugar beet water use in Spain

J. Soler-Rovira*, P. Gómez de la Calle, J.M. Arroyo-Sanz

Escuela Universitaria de Ingeniería Técnica Agrícola, Universidad Politécnica de Madrid, Ciudad Universitaria, s/n 28040 Madrid, Spain, * Corresponding author. E-mail: *jose.soler@upm.es*

Crop production has a great contribution to water use and abstraction. Sugar beet is an important crop in irrigated land in Spain and covers 70,000 ha. Crop and resources management are key factors for a sustainable agriculture. The aim of this work is to model the sugar beet crop growth and water consumption in order to quantify crop water use and virtual water content in different growing conditions.

In site daily meteorological data from SIAR (Magrama, 2011) automatic weather stations network were downloaded for the time period 2001-2010 for the main sugar beet growing provinces in Spain. Average farmers' data for sowing date were considered. Seedling emergence was computed using a thermal time model from literature data (120 ºC.day with a base temperature of 0 ºC). Biomass production of the crop was then estimated computing leaf and canopy development as a function of a thermal time model and the subsequent photosynthetically active radiation interception. Radiation use efficiency was estimated as a function of meteorological daily values of maximum temperature and vapour pressure deficit, despite neither water nor nutrient limitation (Arroyo-Sanz, 2002). Sugar yield was estimated considering an average harvest index (Arroyo-Sanz, 2002). Crop growth was then modelled with daily values of mean temperature and mean solar radiation until farmers' average harvest date for the 10 year period (2001-2010). A soil water balance was modelled and then green and blue water were estimated. Soil water balance included crop ET, drainage, rainfall, irrigation and soil moisture content. Crop ET was the product of reference ETo, from daily meteorological data (Penman-Monteith method), and evaporation or crop coefficients. Evaporation coefficients were estimated before and after harvest as a function of rainfall frequency and ETo. Crop coefficients were estimated considering canopy development. Rainfall daily values were included. In site soil texture defined readily available soil moisture as the difference of soil water content at 10 and 45 kPa (Arroyo-Sanz, 2002). A daily soil water balance was computed considering the drainage water the excess of rainfall over field capacity, effective rainfall that stored in the soil and used by the crop and irrigation water as the amount of water applied to refill the soil moisture until field capacity. Blue water was estimated as the irrigation needs divided by the system application efficiency. Green water was estimated as the sum of soil evaporation before sowing and after harvest, crop evapotranspiration from emergence until the first irrigation and effective rain during the irrigation period. Grey water was considered as the polluted water, and was calculated with the site specific nitrogen fertilisation of the crop, estimated nitrate leaching and water quality standards. Virtual water content was computed adding daily blue, green and grey water for the 10 year period (2001-2010).

The modelled biomass accumulation in Valladolid province in the period 2001-2010 is shown in Figure 1. The temporal trend of crop growth during the growing season shapes a sigmoid curve. Biomass at harvest is the last value of the curve. The most producing years are 2005 and 2007, so the value of the water footprint is relatively lower. The estimated value of sugar virtual water content is shown in Figure 2. The largest values are reached in year 2002 and 2003, the two wettest years. So, green water footprint (mm) is positively and highly correlated with annual rainfall. This relatively high rainfall does not affect the water consumption in irrigation nor the blue water footprint.

References

Arroyo Sanz, J.M. (2002). Influencia del riego sobre el crecimiento, el rendimiento, la calidad industrial y las eficiencias en el uso del agua de riego y de la radiación PAR en el cultivo de la remolacha azucarera (*Beta vulgaris* L.). Ph.D. Thesis. Universidad Politécnica de Madrid, Spain.

Magrama (2011). Sistema de información climática para el regadío. Ministerio de Agricultura, Alimentación y Medio Ambiente. In: http://www.magrama.es/es/agua/temas/observatorio-del-regadio-espanol/Sistema-de-Informacion-Agroclimatica-para-el-Regadio/default.aspx

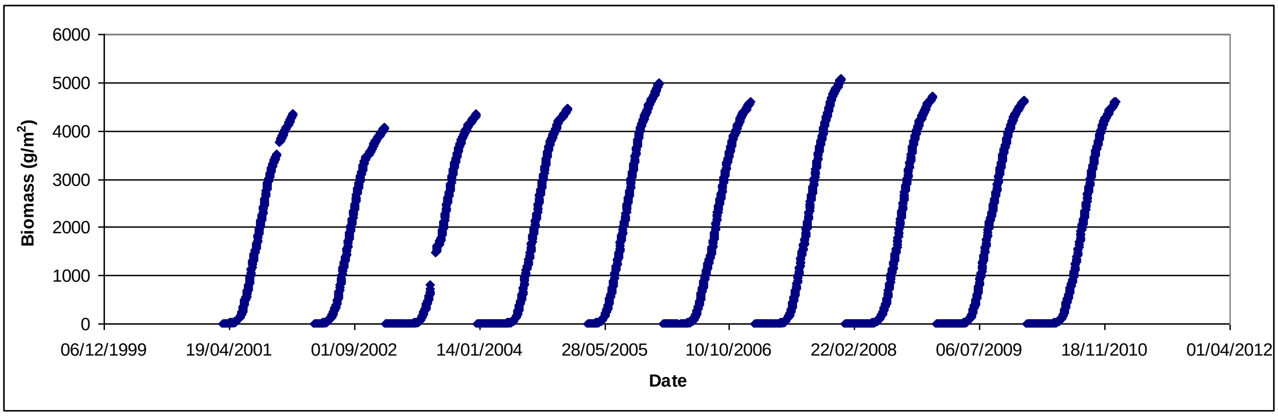


Figure 1. Simulated biomass accumulation of sugar beet crop in Valladolid province (Spain) from 2001-2010.

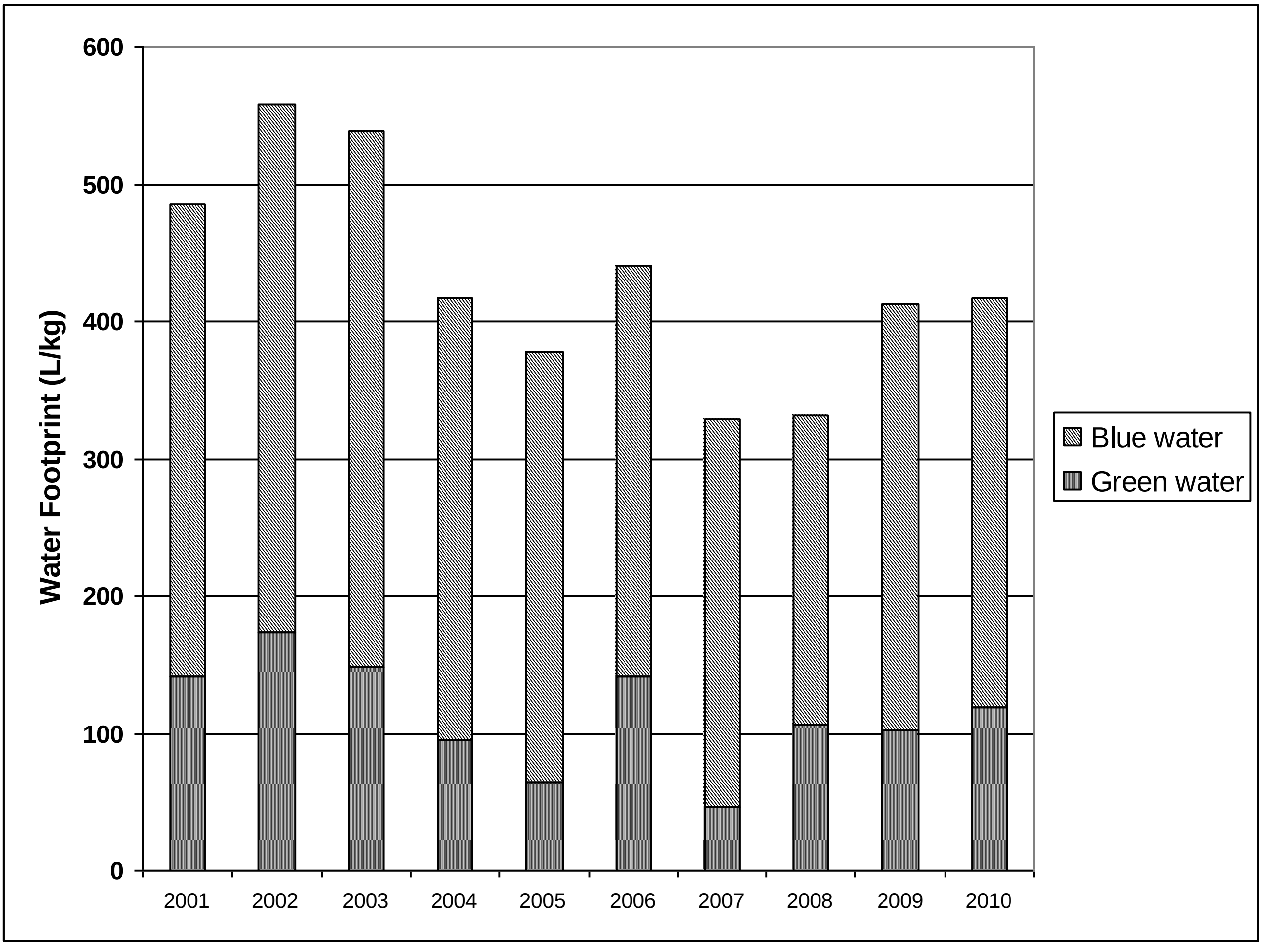


Figure 2. Estimated yearly virtual water content of sugar in Valladolid province (Spain) during the period 2001-2010.

66. Water footprint of milk at dairy farm

Armelle Gac[1,*], Sindy Moreau[1], Didier Moreau[2], Samuel Vionnet[3]

[1] Institut de l'Elevage, Building and Environment Department, BP 85225, F-35652 Le Rheu, France, [2] Danone Dairy Division WorldWide, Environment Direction, F-75000 Paris, France, [3] Quantis International, Parc Scientifique EPFL, Bat-D, 1015 Lausanne, Switzerland, [*] Corresponding author. E-mail: *armelle.gac@idele.fr*

The water resource preservation is becoming a new challenge for agriculture at local and global scales. It is assessed through water footprinting, an emerging methodology. The Water Footprint Network was a precursor for defining a water footprint methodology (Hoekstra et al. 2011) and provides figures focusing on livestock productions (Mekonnen & Hoekstra, 2012). The WFN methodology focuses only on specific types of water flow whereas the WULCA project of the UNEP SETAC LCA Initiative (Koehler & Aoustin 2008) and the future ISO 14046 Water footprint (under development) support that water footprinting should follow the same framework as LCA (ISO 14040 and 14044).
This study presents an application of the water footprint to dairy farms, adapted from Ridoutt and Pfister (2010). This stress-weighted water footprint is based on an assessment of the amount of freshwater withdrawn, consumed water and grey water generated by the production activity (Figure 1). A regionalised characterisation factor, the water stress index, is then multiplied with the inventory to obtain the Water Stress Assessment (WSA) (Figure 2) that reflects the potential for water uses to contribute to water scarcity.
This method has been tested in collaboration with Danone in 3 farms in 3 different countries (Spain, Poland and Saudi Arabia). The functional unit is to produce one kg of whole milk, at the farm gate. The physical flows of water inside the system were described to identify the different sources of freshwater use and consumption (withdrawal water, consumed water). The four main sources (Fig. 3) gather the animals (drinking water, transpiration of animals, water in milk and evaporation from manure), crops and pastures (irrigation and evapotranspiration of plants), the milking parlour (washing and evaporated water) and the farm inputs (diesel, electricity, fertilisers and off-farm feed production). Concerning the flows occurring on the farm assumptions, references and models were used to assess the water requirements and losses. Data about the farm inputs for different countries were provided by databases (ecoinvent v2.2, Quantis Water Database, access December 2011). The Grey water is calculated for the whole farm, based on N leaching potential. Only nitrogen is considered assuming that generally the quantity of water needed to assimilate produced nitrogen will be enough important to assimilate produced phosphorus as well as other pollutants.
The preliminary results show the importance of off-farm processes, especially feed production, on the total WSA of the milk. It occurs that the options for farmers to reduce the water footprint of their products would focus on decreasing the use of some inputs. Nevertheless, the consideration of physical flows at farm level remains important for appropriation of results by the farmers and also because actions to reduce water footprint can more easily undertaken at this level than at the supply chain level. The study also underlines the fact that some methodological issues have to be improved: characterisation of the withdrawal water and assessment of the grey water, as well as implicit weighting of consumed water with grey water.

References

Hoekstra AY, Chapagain AK, Aldaya MM, Mekonnen MM. 2011. The water footprint assessment manual: Setting the global standard. London: Earthscan. p 203

ISO 14040: 2006. Environmental Management - Life Cycle Assessment - Principles and Framework

ISO 14044: 2006. Environmental Management - Life Cycle Assessment - Requirements and guidelines

Koehler A, Aoustin E (2008) Assessment of use and depletion of water resources within LCA. SETAC Europe, Warsaw

Mekonnen, M.M. and Hoekstra, A.Y., 2012. A global assessment of the water footprint of farm animal products, Ecosystems, doi: 10.1007/s10021-011-9517-8.

Ridoutt, B., Pfister, S., 2010. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environmental Change 20, 113-120.

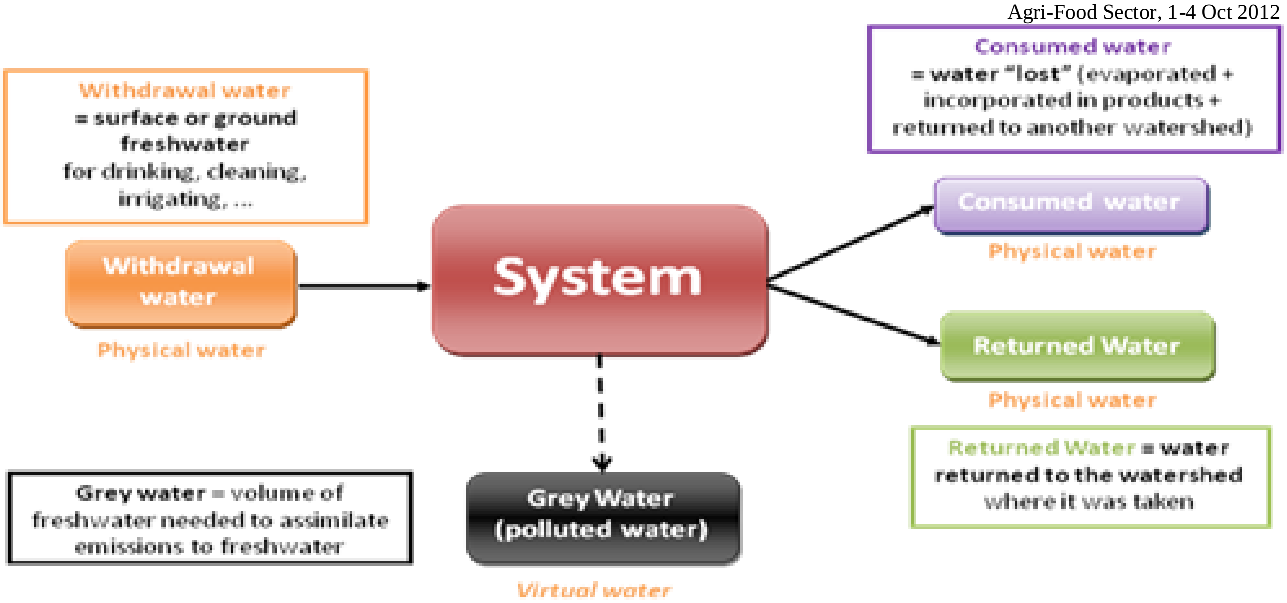


Figure 1. Type of water considered in the WSA methodology (from Ridoutt and Pfister, 2010)

$$\boxed{\begin{aligned} \mathrm{WSA}\,(\mathrm{m}^3) &= \text{Consumed water}\,(\mathrm{m}^3) && *\ \mathrm{WSI} \\ &+ \text{Grey water}\,(\mathrm{m}^3) && *\ \mathrm{WSI} \\ &+ \text{Withdrawal water}\,(\mathrm{m}^3) && *\ 5\%\ *\ \mathrm{WSI} \end{aligned}}$$

Figure 2. Calculation of the Water Stress Assessment (adapted from Ridoutt and Pfister, 2010)

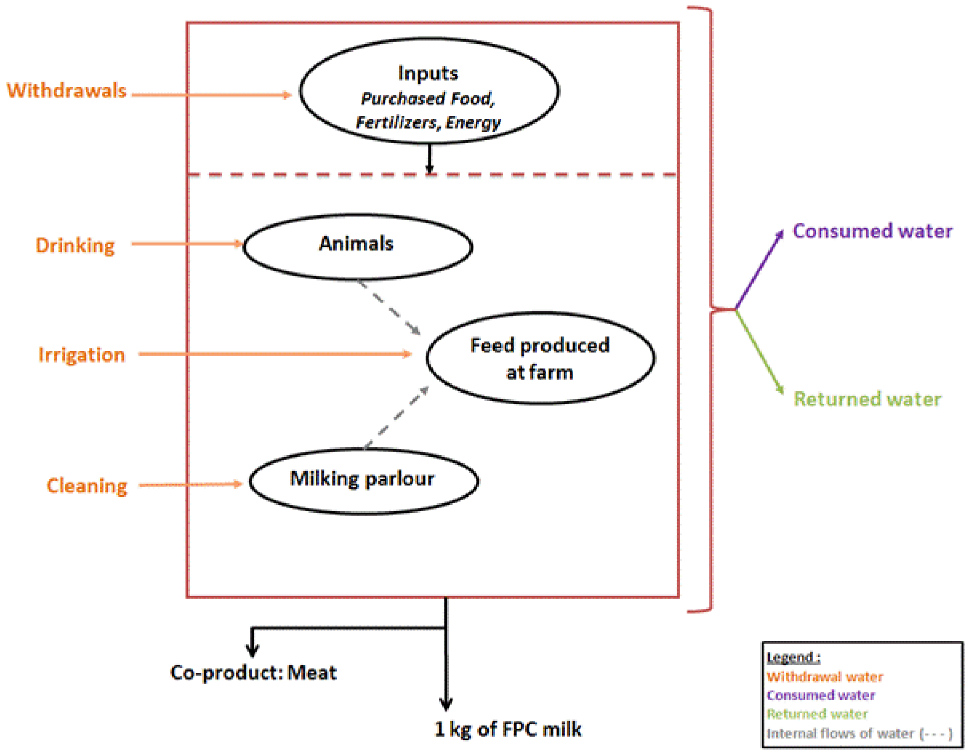


Figure 3. Physical flows of water (withdrawn, consumed and returned water) on a dairy farm

67. Food production in Brazil: challenges for water footprint accounting

Leda Coltro

Institute of Food Technology – ITAL/CETEA, Brazil, E-mail: *ledacolt@ital.sp.gov.br*

Due to the increasing pollution of the rivers as a consequence of industrial and agricultural activities, the importance of evaluating water use throughout the production chains in order to identify hot spots and then to implement improvements for water use reduction has gained more and more attention. That is why in the last decade water use methodologies have being developed by several researchers (Chapagain, Hoekstra, 2007; Milá i Canals.et al. 2009; Ridout, Pfister, 2010). So, nowadays there are more than 30 methodologies to account water use in the life cycle of products or processes.

In 2009, International Organization for Standardization – ISO launched the discussion on a new standard - ISO 14046 on the subject “Water footprint – Requirements and guidelines”. This standard has the aim of harmonising the water footprint criteria adopted to account the use of this resource, as well as the consistency with carbon footprint and other LCA approaches among others (ISO, 2012). By June 2012 the standard was submitted as a vote for CD (committee draft) by SC5 members and it is planned to be published in 2013-2014. According to ISO 14046, the water footprint can be represented by one or more parameters which quantify the environmental impacts of a process, a product or an organisation related to water as follow: 1) the water footprint indicator result (single impact category); 2) the water footprint profile (several indicator results) and 3) the water footprint parameter (weighted result).

ISO 14046 will establish what must be done, but not how to do it. So, the several methodologies for water footprint should follow the requirements of this standard. Despite many questions must be solved until the publication of this standard, one requirement already established by ISO 14046 is that even the single indicator category should consider both the quantity and quality of water resources.

Brazil has 12% of freshwater available in the world. However, approx. 74% of this water is in the Amazon hydrographical region, while the main food producing regions are located at Paraná, São Francisco, Atlântico Leste, Atlântico Sul and Atlântico Sudeste hydrological regions that have only 12.9% of the freshwater available in Brazil.

The Brazilian System for Controlling the Water Resources – SINGREH has an information system on water quantity well established, being the Brazilian Water Agency – ANA responsible for approx. 30% of the controlling points (ANA, 2007). Nevertheless, the water quality control is made only by approx. 12% of this amount of control points (Table 1).

Concerning water quality, only four basic quality parameters are controlled by ANA which are not proper to evaluate the water quality. However, the state and municipal environmental agencies from Minas Gerais, Paraná, Pernambuco, Rio de Janeiro, Rio Grande do Sul and São Paulo control a higher number of quality parameters for some of their main rivers (Table 1) (IBGE, 2010).

It is possible to conclude that the main challenges to account the water footprint of Brazilian food products are: 1) lack of quality data for many Brazilian rivers, and 2) many of quality water data are kept by the state and municipal environmental agencies as private data. So, much work should be done in order to get reliable water footprint of Brazilian products.

References

Agência Nacional de Águas – ANA, 2007. Evolução da rede hidrometereológica nacional. v. 1, n.1, 15p.

Chapagain, A.K., Hoekstra, A.Y., 2007. The water footprint of coffee and tea consumption in the Netherlands. Ecological Economics 64, 109 – 118.

ISO - International Organization for Standardization. ISO/CD 14046: Water footprint – Requirements and guidelines, 2012. (Working Draft 2)

Instituto Brasileiro de Geografia e Estatística – IBGE. Indicadores de desenvolvimento sustentável Brasil 2010. Rio de Janeiro. 442p.

Milá i Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., Cliff, R., 2009. Assessing freshwater use impacts in LCA. Part I – inventory modeling and characterization factors for the main impact pathways. Int. J. LCA, 14, 28-42.

Ridoutt, B.G., Pfister, S., 2010. A revised approach to water footprinting to make transparent the consumption and production on global freshwater scarcity. Global Environ. Change, 20, 113-120.

Table 1. Information system on water resources in Brazil (ANA, 2007; IBGE, 2010).

Aspects	Control points	Parameters evaluated
Water quantity	11,260	Discharge
Water quality [a]	1,340	pH Dissolved Oxygen Conductivity Temperature
Water quality [b]	----	BOD Water Quality Index - WQI [c]

[a] Brazilian Water Agency
[b] State and Municipal Environmental Water Agencies
[c] WQI calculated from pH, Temperature, Dissolved Oxygen, BOD, N, P, total soluble solids, turbidity.

68. A life cycle C3 commons: communicate, collaborate, connect

Shelly Grist*, Mark Toleman

University of Southern Queensland, Toowoomba, Queensland, Australia, * Corresponding author. E-mail: *shelly.grist@usq.edu.au*

Practitioners conducting life cycle assessments (LCA) are uniquely dependent on methodology, databases, and tools, as wells as applications of LCA developed by their peers (Gnansounou et al., 2009; Suh et al., 2004). Conferences such as LCA Food 2012 serve as a global forum. They provide a *physical* meeting place for LCA practitioners from around the globe to share knowledge, discuss topics of interest to the community and connect with peers. Unfortunately international conferences like LCA Food are annual at best and not all who would benefit are able to attend. There is an on-going need for a meeting place, with relevant community resources 'held in common'. In essence a virtual community commons where LCA practitioners can continue to communicate, collaborate and connect.

At LCA Food 2010 the authors presented a poster proposing an LCA community website. The proposal outlined a *virtual* meeting place for community members: to find and share information through searchable publication and data knowledgebases; to learn from each other by asking questions of authors, watching online video tutorials and participating in discussion forums and blogs; and to connect with colleagues with whom to collaborate.

The proposed LCA community website is being designed and developed now. The website is based on SilverStripe, an open source framework and configuration management system (CMS). An open source solution was selected in keeping with the design philosophy of a community commons. Once the community site is completed both the site and its software will be available as a community resource. It's hoped that the LCA C3 Commons website on its own will prove to be a valuable resource for the global LCA community. The solution in whole or in part will also be available for LCA organisations who wish to download it and host it for their own use.

A working prototype will be available for LCA Food 2012 conference attendees to trial. Feedback will be requested on existing features as well as suggestions for additional features.

References

Gnansounou E, Dauriat A, Villegas J, Panichelli L. Life cycle assessment of biofuels: Energy and greenhouse gas balances. Bioresource Technology, 100: 4919-4930.

Suh S, Lenzen M, Treloar GJ, Hondo H, Horvath A, Huppes G, et al. System boundary selection in life-cycle inventories using hybrid approaches. Environmental Science & Technology, 38: 657-664.

69. Assessment of the environmental impact of dietary intake habits in the UK

Janet Kyle[1,*], Jennie Macdiarmid[2] and Tony Craig[3]

[1]Public Health Nutrition Research Group, Institute of Applied Health Sciences, University of Aberdeen, Foresterhill, Aberdeen, AB25 2ZD, UK, [2]Rowett Institute of Nutrition and Health, University of Aberdeen, UK, [3]James Hutton Institute, Craigiebuckler, Aberdeen, AB15 8QH, UK, [*] Corresponding author. E-mail: *j.kyle@abdn.ac.uk*

The food system is estimated to account for 18-20% of total UK greenhouse gas (GHG) emissions, increasing to 30% factoring land use change (Audsley et al 2009), and food consumption habits make an important contribution to these emissions. With the introduction of legally binding commitments to reduce GHGE; the UK Climate Change Act (2008) set targets to reduce GHGE by at least 34% by 2020, this will require not only efficiency savings in food production and processing, but also adoption of lower impact diets by consumers. However, in the UK, few studies have assessed the impact of actual dietary habits and the environmental impact, measured in the form of GHG emissions. The current study aimed to investigate the potential of assessing the environmental impact of dietary intake habits using a novel approach to measuring and ranking the environmental impact of food consumption into high, medium and low levels of emissions and compare this with the nutritional quality of the diet.

A random sample to people living in the South West of Scotland were asked to completed an anonymised postal survey, 527 people, average age 58y (range 20-90y) returned the questionnaires. Habitual diet was measured using a food frequency questionnaire (FFQ) (www.foodfrequency.org), which asks for the frequency of consumption of 170 food and drink items and has been previously validated for dietary assessment. Food and nutrient intake and environmental impact were determined by linking, the FFQ response information to an in-house food nutrient composition and food related GHG emissions database. Estimation of average daily GHG emissions acted as a marker of environmental impact.

GHG emissions values for food items ‘as eaten’ were calculated from raw food commodities data from one of the most comprehensive lists of GHG emissions in the UK (Audsley *et al* 2009). These food commodity data did not represent the full ‘cradle to grave’ life cycle analysis (LCA), but rather the average GHG emissions for the production of raw food commodities up to the regional distribution centre (RDC). The RDC was described as a nominal boundary of agricultural and food ingredient production up to the point of distribution in the UK and estimated to account on average for 56% of the total LCA GHG emissions. The later stages of processing raw basic ingredients into edible food products ready for consumption were not included. These data were harmonised with the nutrient composition data to reflect food as eaten, with adjustments for edible portion, cooking gains and losses, and production of composite dishes and food products, such as lasagne, crisps and cakes, employing a proportion of ingredients approach,. It is important to note for example that GHG emissions for cooked meat will in relative terms increase due to weight loss whilst cooking, while the emissions for rice will fall with cooking due to weight gains through hydration. However it is important to highlight there is a lack of complete ‘cradle to grave’ LCA for commonly consumed foods.

Preliminary results indicate that though, as expected, increasing energy intake and levels of GHG emissions are closed linked, the actual quality of the diet from a health perspective does not appear to diminish. For example, the relative contribution of fat (34% of total energy), protein (16% of total energy), to total energy intake remains unchanged with increasing levels of GHG emissions from the diet. This study has lead to deeper insights into the interactions between diet, environment and health which will contribute to development of the population based approaches to reducing the environmental impact of dietary intake.

References

Audsley E, Brander M, Chatterton J, Murphy-Bokern D, Webster C, Williams A., 2009. How Low Can We Go? An assessment of greenhouse gas emissions from the UK food system and the scope for reduction by 2050. WWF-UK.

70. Fish or meat? Is this a relevant question from an environmental point of view?

Benedikt Buchspies[1], Niels Jungbluth[2,*]

[1] *b.buchspies@gmx.de*, [2] ESU-services Ltd., CH-8610 Uster, Switzerland, [*] Corresponding author. E-mail: *jungbluth@esu-services.ch*

Nutrition accounts for about 30% of environmental impacts caused due to the final consumption of Swiss households (Jungbluth et al., 2011). It is thus the most important consumption sector from an environmental point of view. Out of this meat is the most important product group accounting for about 26% of impacts of food consumption (Jungbluth et al., 2012). Therefore, it is necessary to investigate and understand the environmental impacts of food consumption and possibilities for the reduction of environmental impacts. One option discussed for this is a reduction of meat consumption. Fish might be considered as a possible replacement by consumers.

Within a recent study we assessed the environmental impacts of different fish products sold in Swiss supermarkets (Buchspies et al., 2011). The life cycle inventory for different types of fish is based on published work by different authors. These data have been harmonised and implemented in the EcoSpold format. The defined functional unit is one kg of frozen cod, canned mackerel, canned herring or smoked salmon. The former three are caught and processed in Denmark; the latter is farmed and processed in Norway. Data for the production of different meat products were available from earlier studies (Jungbluth, 2000; Jungbluth et al., 2011). To evaluate environmental impacts, the ecological scarcity method 2006 and global warming potential 2007 are used.

When comparing the results with the ecological scarcity method 2006, high sea fish is at the lower end of range for all compared products. Fishing and packaging are main determinant in regard to environmental impacts of high sea fishing. Salomon's environmental impacts are nearly as high as those of veal. Feed production and the nutrient emissions into the sea are quite important for the total environmental impacts. In regard to the global warming potential, fish offers an alternative to meat. Depending on the type of fish, emission per kg of filet range between 3.7 and 6.6 kg CO_2-eq. For farmed salmon indirect N_2O emissions from nutrient emissions need to be considered. Fish cannot be regarded generally as a more environmentally friendly food product than meat, because environmental impacts of different fish products might be quite variable and be even higher than these of meat.

References

Buchspies B, Tölle SJ, Jungbluth N. Life Cycle Assessment of High-Sea Fish and Salmon Aquaculture [Practical training report]. Uster, CH: ESU-services Ltd.; 2011.

Jungbluth N. Umweltfolgen des Nahrungsmittelkonsums: Beurteilung von Produktmerkmalen auf Grundlage einer modularen Ökobilanz [Dissertation Nr. 13499]. Berlin, D: Eidgenössische Technische Hochschule Zürich, Umweltnatur- und Umweltsozialwissenschaften, dissertation.de; 2000.

Jungbluth N, Flury K, Büsser S, Stucki M, Frischknecht R. Life cycle inventory database on demand: EcoSpold LCI database of ESU-services. Uster, CH: ESU-services Ltd.; 2011.

Jungbluth N, Itten R. Umweltbelastungen des Konsums in der Schweiz und in der Stadt Zürich: Grundlagendaten und Reduktionspotentiale. Zürich: ESU-services GmbH im Auftrag der Stadt Zürich (unpublished); 2012.

Jungbluth N, Nathani C, Stucki M, Leuenberger M. Environmental impacts of Swiss consumption and production: a combination of input-output analysis with life cycle assessment. Bern, CH: ESU-services Ltd. & Rütter+Partner, commissioned by the Swiss Federal Office for the Environment (FOEN); 2011. p. 171.

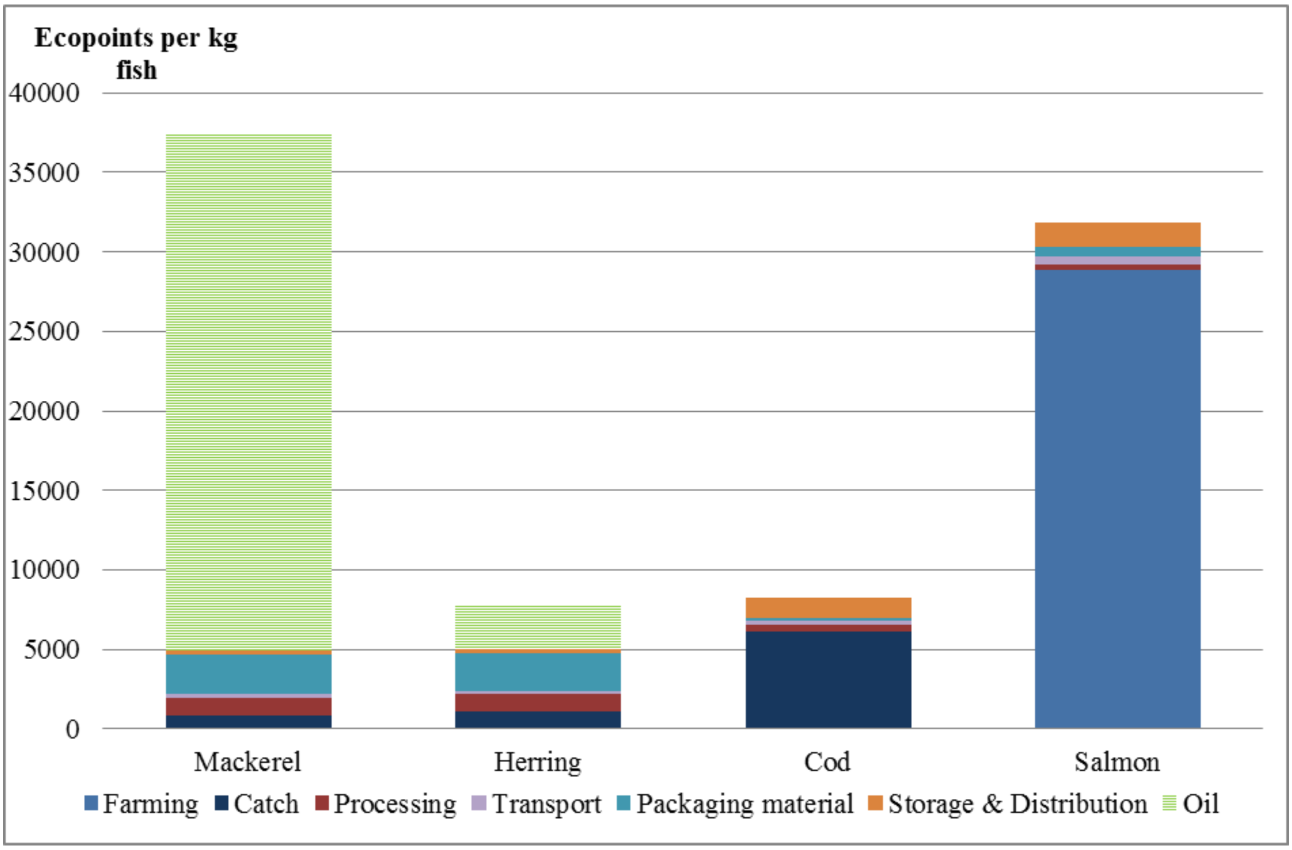


Figure 1. Distribution of eco-points for different life cycle stages

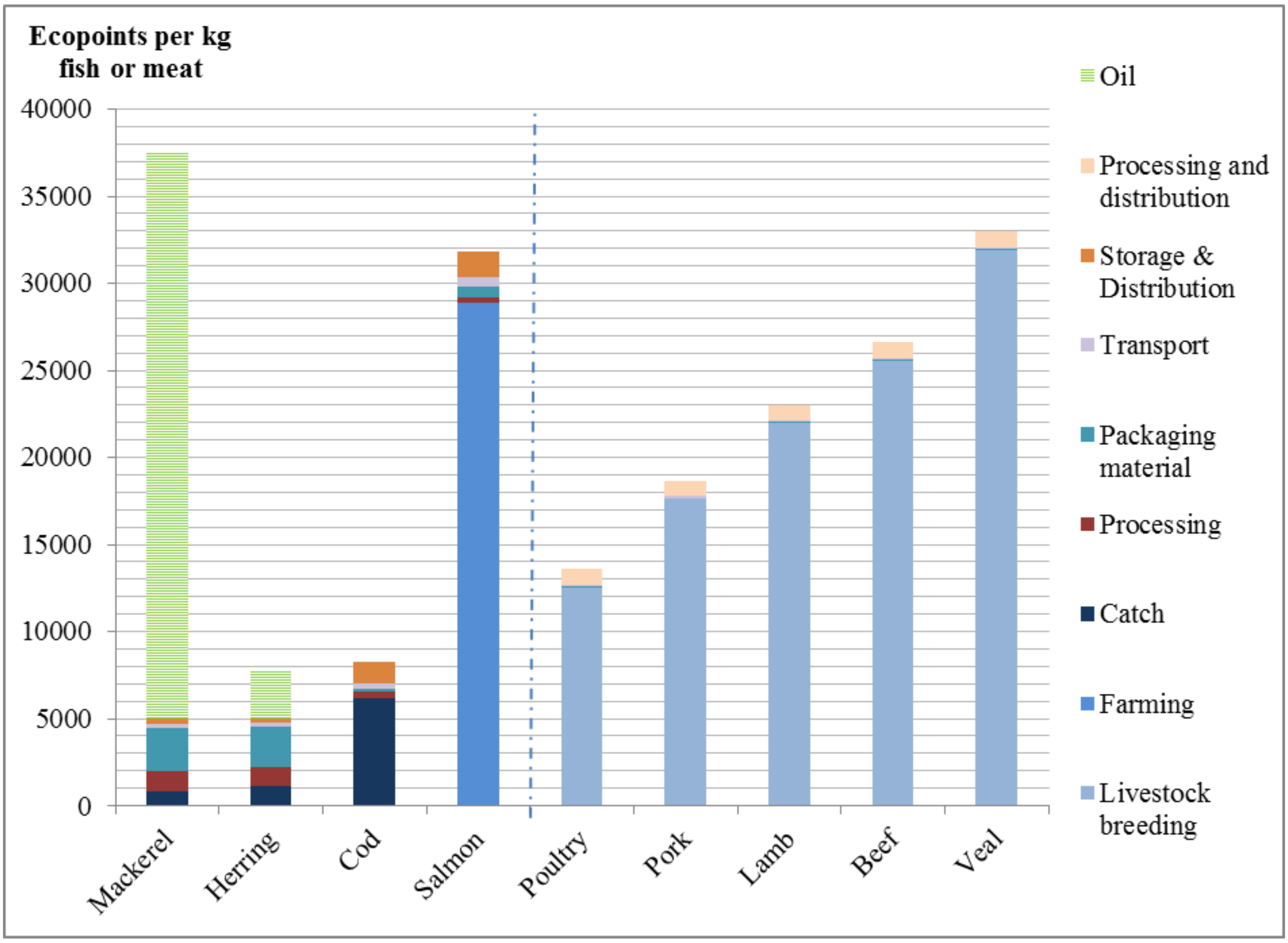


Figure 2. Comparison of fish and meat – ecological scarcity method 2006

71. The impacts of food choices on the state of the Baltic Sea - example of the EIOLCA study

Virpi Vorne[1,*], Yrjö Virtanen[2], Sirpa Kurppa[2], Sanna Hietala[1], Matti Verta[3], Juha Grönroos[3], Liina Laumets[4], Elina Līce[5]

[1] MTT Biotechnology and Food Research, P.O.Box 413, FI-90014 University of Oulu, Finland, [2] MTT Biotechnology and Food Research, FIN-31600 Jokioinen, Finland, [3] Finnish Environment Institute (SYKE), P.O.Box 140, FI-00251 Helsinki, Finland, [4] University of Tartu, Ülikooli 18, 50090 Tartu, Estonia, [5] University of Latvia, 19 Raina Blvd., Riga, LV 1586, Latvia, [*] Corresponding author. E-mail: *virpi.vorne@mtt.fi*

The Baltic Sea is a small and relatively shallow brackish body of water located in northern Europe. It is the second largest brackish water basin in the world and is also considered to be the most polluted. Low salinity also makes the Baltic's unique ecosystems sensitive to changes resulting from human activity. One of the biggest problems is accelerating eutrophication caused by nutrient inflow and deposit.
The food production chain is one the most resource demanding and polluting sectors. Agriculture and the food chain are largely responsible for eutrophication and pollution of waterways. Food consumption represents a significant part of the environmental load of households and, in addition, food can contain hazardous compounds resulting, for example, from farming and livestock production and traces of harmful chemicals, like those in fertilisers.
An average 3 600 tonnes of phosphorus and 78 000 tonnes of nitrogen were leached into the Baltic Sea from Finland annually between 2000 and 2006. Approximately 28% of the phosphorus and 36% of the nitrogen load were from natural sources. The runoffs of the food chain were estimated at 2 320 tonnes of phosphorus and 34 680 tonnes of nitrogen in 2005, corresponding to about 80% of the diffuse phosphorus load, and about 70% of the diffuse nitrogen load from socio-economic activities. Raw material production governs the total environmental load of the domestic food chain. Its contribution to the eutrophic emissions is 83% on average. About 95% of nitrogen and phosphorus leaching stems from raw material production.
Based on the results of the EIOLCA food chain model, the eutrophication intensity varies among different foodstuffs: beef has the highest eutrophication intensity of all meats, about three times higher than that of pork, and seven times that of poultry. The eutrophication impacts of plants also vary among species: grain has the highest intensity of the plant-based raw materials.
Eutrophication intensity was estimated for Estonian and Latvian food raw materials using the Finnish EIOLCA model, which was modified for the emission factors of the raw material production sectors. For Estonia eutrophication intensity estimates appeared higher than for Finland. For the Latvian cereals the estimate was considerably lower than for the Finnish ones. This reflected through grain fodder to pork, poultry and eggs.
The effects of diet were studied with help of the EIOLCA food chain model as part of a project on the coherency assessment of other policies with the environmental policies in Finland. The modelling results indicated that eutrophication caused by the food chain could be reduced by about 7% if the recommended diet were to have full effect on private food consumption. The eutrophication intensities, which are the gradients of the changes and are much higher for animal protein foods than for carbohydrate foods, explain the change.
Human activity and land-based agricultural operations exert key effects on the nutrient contents of the Baltic Sea. The most important factors are the total area of agricultural land, its local distribution, diversity and volumes of different crops produced, use of fertilisers, and other agricultural operations. Nutrient load into the Baltic Sea can be reduced by improving crop yields, by optimising crop selection and fertiliser use, and by practising efficient nutrient recycling.

72 Potential contribution of dietary changes to improvements in the environment and human health

G.A. Rood[1,*], H. Westhoek, A. de Marco[2], J.P. Lesschen[3], A. Leip[4], D. Nijdam, S. Wagner[3,1]

[1] PBL Netherlands Environmental Assessment Agency, Bilthoven, The Netherlands, [2] ENEA, Rome, Italy, [3]Alterra, Wageningen-UR, Wageningen, The Netherlands, [4]European Commission - Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy, [*] Corresponding author. E-mail: *trudy.rood@pbl.nl*

The consumption of animal products has a large environmental impact, both within and outside Europe. Livestock production is a large user of land and a source of greenhouse gas and nitrogen emissions. At the consumption side, European diets are characterised by a high intake of protein, dominated by animal protein and a relatively high intake of saturated fat, also mainly originating from animal products. Reducing meat and dairy consumption could have various beneficial effects and offer a scope for change on a European level.

For our study, we based consumption on supply data from the FAO. Health impacts were assessed by calculating the intake of proteins, saturated fats and red meat, and comparing these intakes against health recommendations. The recommendations were obtained from the World Health Organization (WHO) and the World Cancer Research Fund (WHO, 2003; WCRF, 2007). Environmental impacts – from land use, and greenhouse gas and nitrogen emissions – were assessed using a review of LCAs of animal products and the MITERRA-Europe model. The MITERRA-Europe model calculates annual nitrogen flows and greenhouse gas emissions from agriculture, following a life-cycle approach that reaches 'up to the farm gate' (Velthof et al., 2009; Lesschen et al., 2011).

We assessed scenarios with reductions of 25% and 50% in beef and dairy consumption (S1, S4) and other scenarios with similar reductions in pork, poultry and egg consumption and production (S2, S5). Furthermore, scenarios with a 25% and 50% reduction in all meat and dairy were assessed (S3, S6). The energy intake was kept at a constant level in all scenarios. Only the protein intake was decreased, taking into account the minimum amount of protein recommended by the WHO.

Our study showed that reductions in meat and dairy consumption in fact decreased the environmental impacts, due to the large differences in land use, and carbon and nitrogen footprints between food products. Currently, European diets contain more proteins than necessary as well as more saturated fat which mainly originates from animal products, than the maximum recommended amount by the WHO. Also, current intake of red meats is higher than recommended by the WCRF. Diets with lower meat and dairy products have been found to reduce the risk of various diseases. It was concluded that a decrease in the consumption of animal products in the EU27 may result in a large reduction in environmental impacts and could be beneficial to human health.

References

Lesschen J.P., van den Berg M., Westhoek H.J., Witzke H.P. and Oenema O. 2011. Greenhouse gas emission profiles of European livestock sectors. Animal Feed Science & Technology 166-167, 16-28.

Velthof G.L., Oudendag D., Witzke H.P., Asman W.A.H., Klimont Z., Oenema, O. 2009. Integrated assessment of nitrogen emissions from agriculture in EU-27 using MITERRA-EUROPE. J. of Environ. Quality 38, 402-417.

WCRF& AICR (2007), Food, nutrion, physical activity and the prevention of cancer: a global perspective. 2nd Expert report ed. World Cancer Research Fund, Washington D.C.

WHO (2003a). Diet, Nutrition and the prevention of chronic diseases. WHO technical report series 916, WHO, Geneva.

WHO (2003b). Food based dietary guidelines in the WHO European region. WHO-Europe, Copenhagen.

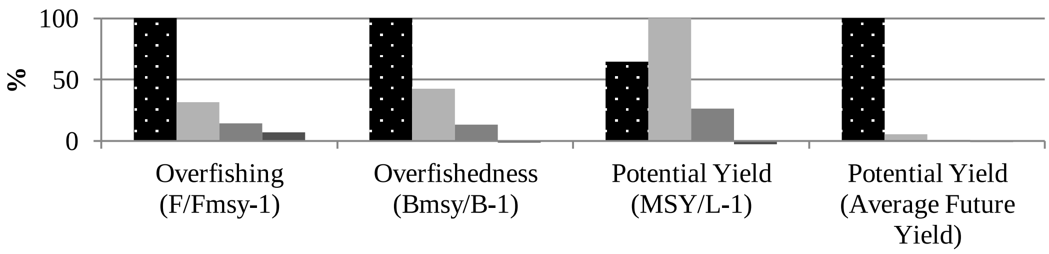
73. Benchmarking Swedish cod and herring products by spatial-temporal life cycle assessment

Andreas Emanuelsson[1,2,*], Friederike Ziegler[1], Leif Pihl[2], Mattias Sköld[3], Sara Hornborg[1], Ulf Sonesson[1]

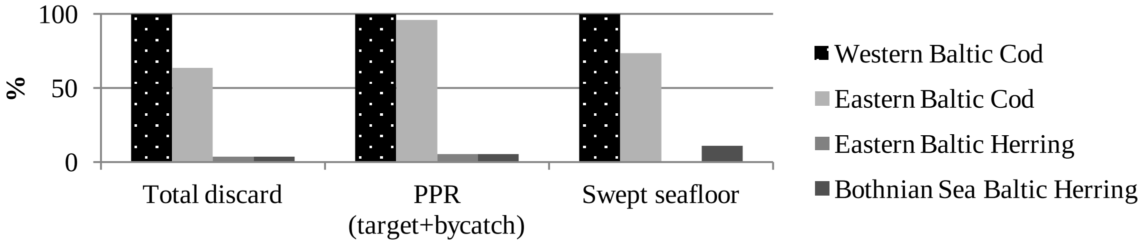
[1]SIK – The Swedish Institute for Food and Biotechnology, Department of Sustainable Food Production, 402 29 Gothenburg, Sweden, [2]University of Gothenburg, Department of Biological and Environmental Science, Kristineberg, 451 78 Fiskebäckskil, Sweden, [3]Swedish University of Agricultural Sciences, Department of Aquatic Resources, 453 30 Lysekil, Sweden, [*] Corresponding author. E-mail: *ae@sik.se*

Fisheries today affect the environment and its ecosystems with a broad range of impacts. Methods to quantify and compare these lay within the Life Cycle Assessment (LCA) framework when traditional emission based impact categories are complemented with biological indices covering internationally acknowledged aspects of direct target stock damage (i.e. overfishing) and direct ecosystem damage. This study demonstrates the application of seafood LCA's by midpoint benchmarking the environmental performance of the Swedish trawl fleet in the Baltic Sea. It was done by comparing fishing activities on two stocks of cod and two stocks of herring in 2008 (spatial resolution) in terms of average products. In addition one stock of each species was compared over time in terms of key drivers, i.e. between 2002 and 2008 (temporal resolution). Newly developed/refined impacts categories of Overfishing, Overfishedness, Wasted Potential Yield, Primary Production Required and Swept Area were applied together with a full set of ReCiPe midpoint impact categories. The results showed major differences between stocks (Fig. 1), and positive or negative trends were highlighted by defined impact groups of target stock, ecosystem and emission impact. Temporal variation was found substantial in all categories in relation to key drivers. The case study also demonstrated the weakness of generally using low fuel consumption as an indicator for good stock status due to technical differences influencing the catchability. Data availability was discussed as a limiting factor for applicability and a sensitivity analysis of the model assumptions performed. Trade-offs were discussed and final scores evaluated in relation to decision support and the future role of seafood LCA's. With the inclusion of biological impact categories in the methodology, LCA´s are concluded to be a useful complementary tool for fisheries managers, seafood industry or seafood labelling/consumer guides. But without directly addressing and quantifying the biological effects on target stock and ecosystem any future seafood LCA could easily be misinterpreted or even deliberately misused as a biased proxy for holistic "environmental" damage.

A – Target Catch



B – Ecosystem



C – Emissions

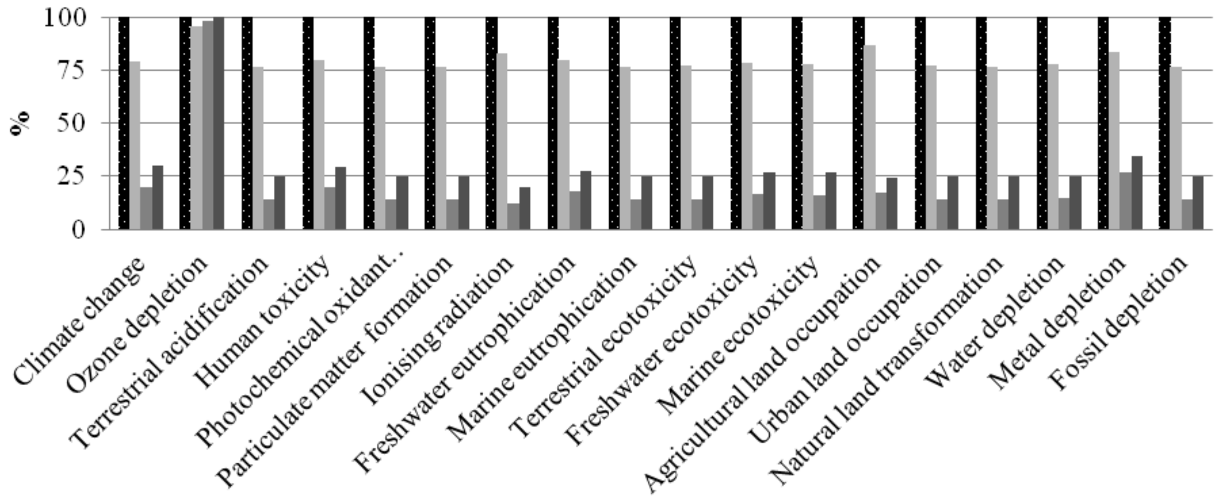


Fig. 1a-c. Comparison between environmental performances of four Baltic fish stocks during 2008 over three groups of impact categories: A – Direct target stock impact, B – Direct ecosystem impact. C- Indirect emission impact.

74. Linking geographical certification and environmental declaration of food products: the apple Mela Rossa Cuneo (PGI) case study

Alessandro K. Cerutti[1,*], Andrea Sacchetto[1], Sander Bruun[2], Dario Donno[1], Gabriele L. Beccaro[1], Giancarlo Bounous[1]

[1] Department of Arboriculture, University of Torino, Via Leonardo da Vinci, 44 – 10095 Grugliasco (TO), Italy, [2] Department of Agriculture and Ecology, University of Copenhagen, Thorvaldsensvej 40, 1871 Frederiksberg, Copenhagen, Denmark, [*] Corresponding author. E-mail: *alessandro.cerutti@unito.it*

Protected Geographical Status (PGS) is a legal body defined in European Union law in order to protect the names of regional foods. In this framework the Protected Geographical Indication (PGI) is a well known name used on certain products originating from a specific geographical location (e.g. a region or a country). The use of a PGI is also a certification that the product possesses certain qualities or is made according to local traditional methods. As PGI foods enjoy a certain reputation, due to their geographical origin, they are of higher commercial value both on local markets and in wholesale retail systems, in which PGI foods from several national areas are usually collected under specific brands in order to differentiate them from other products.

PGI foods are connected to specific quality traits without considering impact on the environment during the production. However, as the PGI seal reflects a standardised production, it may also be associated to standardised environmental impacts. Therefore it may be possible to combine the geographical certification with some sort of environmental declaration which may thus result in a positive effect of the commercial performance of such product. Even though the production of PGI foods are standardised to a certain extent, the application of an environmental assessment methods to a PGI food is complicated because of the variability of the production process (e.g. diverse production systems may merge to the same PGI) and production sites.

Therefore, the aim of this research is to evaluate the suitability and the methodological requirements for the application of the Environmental Product Declaration (EPD®) system to a PGI food case study. The focus will be the red apple "*Mela Rossa Cuneo*", which is a PGI product from the province of Cuneo (Piedmont, Italy). This apple is characterised by an intense red colouring of the peel and particular bright and shiny shades of colour. The PGI is constituted by a small list of apple varieties and their clones deriving from a strict quality selection of the varieties still grown today.

According to the EPD® system, the specific Product Category Rules (PCRs) for fruit and nuts products where applied to the *Mela Rossa Cuneo* in order to quantify the environmental impacts of the production. A cradle-to-gate LCA has been performed in accordance with the guidelines and requirements of the ISO 14040. Data regarding agricultural inputs, consumption and orchard management have been obtained directly from the growers, using a questionnaire for the season 2010-2011, and by consultation of the Italian protocols for such production. The environmental aspects of the production of fertilisers and pesticides have been included within the boundaries.

75. Eco-labelling and information asymmetry: independent consumer information through eco-labels in Bulgaria

Elka Vasileva*, Daniela Ivanova, Stilian Stefanov, Nina Tipova

Department of Economics of Natural Resource, University of National and World Economy, Sofia, Bulgaria,
* Corresponding author. E-mail: *elkav@unwe.acad.bg*

Third party eco-labelling can serve three functions in the marketplace: 1) it can provide independent evaluation and endorsement of a product; 2) it can act as a consumer protection tool; and 3) it can be a means of achieving specific environmental policy goals (Boström and Klintmann, 2008; Nilsson et al., 2004; Rubik and Frankl, 2005).

Eco-labelling is a means to narrow the information gap: independent third parties assure the consumer that the producer has complied with published, transparent, environmentally friendly standards. In recent years a number of critical research studies have been published which evaluate the reliability of eco-labels (Amstel van Saane et al., 2008; D'Souza et al., 2007; Koos, 2011; Nilsson et al., 2004).

An identification and analysis of third parties eco-labelling foods schemes in Bulgaria is presented. Discussion focuses on the coverage, promoters/owners and stakeholder inclusion in such schemes, and consideration of their impartiality, accessibility, independence and transparency. The disclosure of information to consumers by third parties eco-labelling certification (organic labels) is explored. Conclusions are drawn regarding their potential role in future shifts towards efficient consumer information practices in Bulgaria.

References

Amstel van Saane M., Driessen P., Glasbergen P., 2008. Eco-labelling and Information Asymmetry: a Comparison of Five Eco-labels in the Netherlands. Journal of Cleaner Production 16 (3), 263-276.

Boström M., Klintmann M., 2008. Eco Standards, Product Labelling and Green Consumerism, Houndmills: Palgrave Macmillan.

D'Souza C., Taghian M., Lamb P., Peretiatko R., 2007. Green Decisions: Demographics and Consumer Understanding of Environmental Labels. International Journal of Consumer Studies. 31 (4), 371 – 376.

Koos S., 2011. Varieties of Environmental Labelling, Market Structures, and Sustainable Consumption Across Europe: A Comparative Analysis of Organizational and Market Supply Determinants of Environmental-Labelled Goods. Journal of Consumer Policy. 34, 127 - 159.

Nilsson H., Tuncer B., Thidell A., 2004. The Use of Eco-labelling like Initiatives on Food Products to Promote Quality Assurance—is There Enough Credibility?. Journal of Cleaner Production. 12, 517–526.

Rubik F, Frankl P., 2005. The Future of Eco-labelling. Making Environmental Product Information Systems Effective, Sheffield: Greenleaf Publishing Ltd.

76. SENSE: harmonised environmental sustainability in the European food and drink chain

Guillermo Pardo[1,*], Aintzane Esturo[1], Jaime Zufía[1], Ulf Sonesson[2], Thorkild Nielsen[3], Erling Larsen[4], Guðrún Ólafsdóttir[5], David Barling[6], Begoña Perez Villarreal[1]

[1,*]AZTI-Tecnalia, Spain, [2] Swedish Institute for Food and Biotechnology, Sweden, [3] Aalborg University Centre, Denmark, [4] DTU-Aqua, Denmark, [5] University of Iceland, Iceland, [6] The City University London, UK, [*] Corresponding author. E-mail: *gpardo@azti.es*

The food and drink industry in Europe, of which 99% are SMEs, is highly fragmented, and food chains are very complex. Hence, to assess the environmental impact of a product there is a need for applying integrated, harmonised and scientifically robust methodologies, together with appropriate communication strategies for making environmental sustainability understandable to the market. However, there are difficulties in developing a commonly agreed methodology for environmental impact assessment that still need to be overcome, notably: complexity of food chains, large number of agents involved, different indicators depending on the business sector, regional differences, high data intensity, costs and expertise required.

Considering the previous difficulties, SENSE project will deliver a harmonised system for environmental impact assessment of food and drink products. The research will evaluate existing relevant environmental impact assessment methodologies, and consider socio-economical, quality and safety aspects, an approach that have been rare up till now, to deliver a new integral system that can be linked to monitoring and traceability data.

By means of incorporating a simplified data gathering system, a matrix of key environmental performance indicators and a certification scheme into the new methodology the project will provide a tool to effectively reflect the sustainability profile of any product. The e-information will allow food and drink chain actors, and especially industrial SMEs, to set realistic environmental sustainability goals and improve their competitiveness towards a more sustainable production culture to all levels of the production process.

The sustainability information collected along the production cycle of any food stuff will be finally reflected into an Environmental Identification Document (EID) which will contribute to enhanced environmental sustainability motivation of the usual purchasing behaviour of consumers and provide a competitive advantage to those products (and companies) which choose to use this approach.

The communication of the information will have a visual presentation that will be intuitively understandable by all the stakeholder of the food and drink sector, and especially the consumers. By means of a comprehensive environmental communication between the industry and consumers will lead those to choose for the food products communicated as being environmentally friendly.

The main results of SENSE will be:

- Standard key environmental performance indicators (KEPI) and specifically for three food and drink chains from three regions (Northern Europe, Mid-Eastern Europe and Mediterranean Europe)
- Harmonised methodology for environmental impact assessment
- SENSE-tool for simplified environmental data collection
- Environmental Identification Document (EID) and EID-Communication Platform
- Certification Scheme Concept (CSC) for sustainability based on EID
- Road map for policy and governance implementation

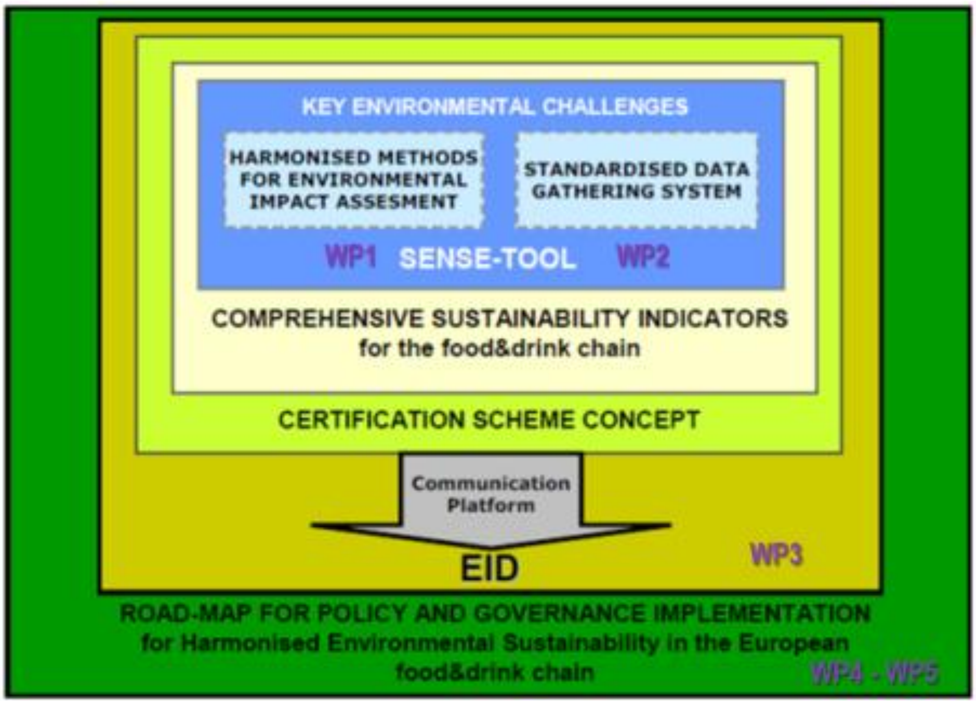


Figure 1. Diagram of objectives of SENSE

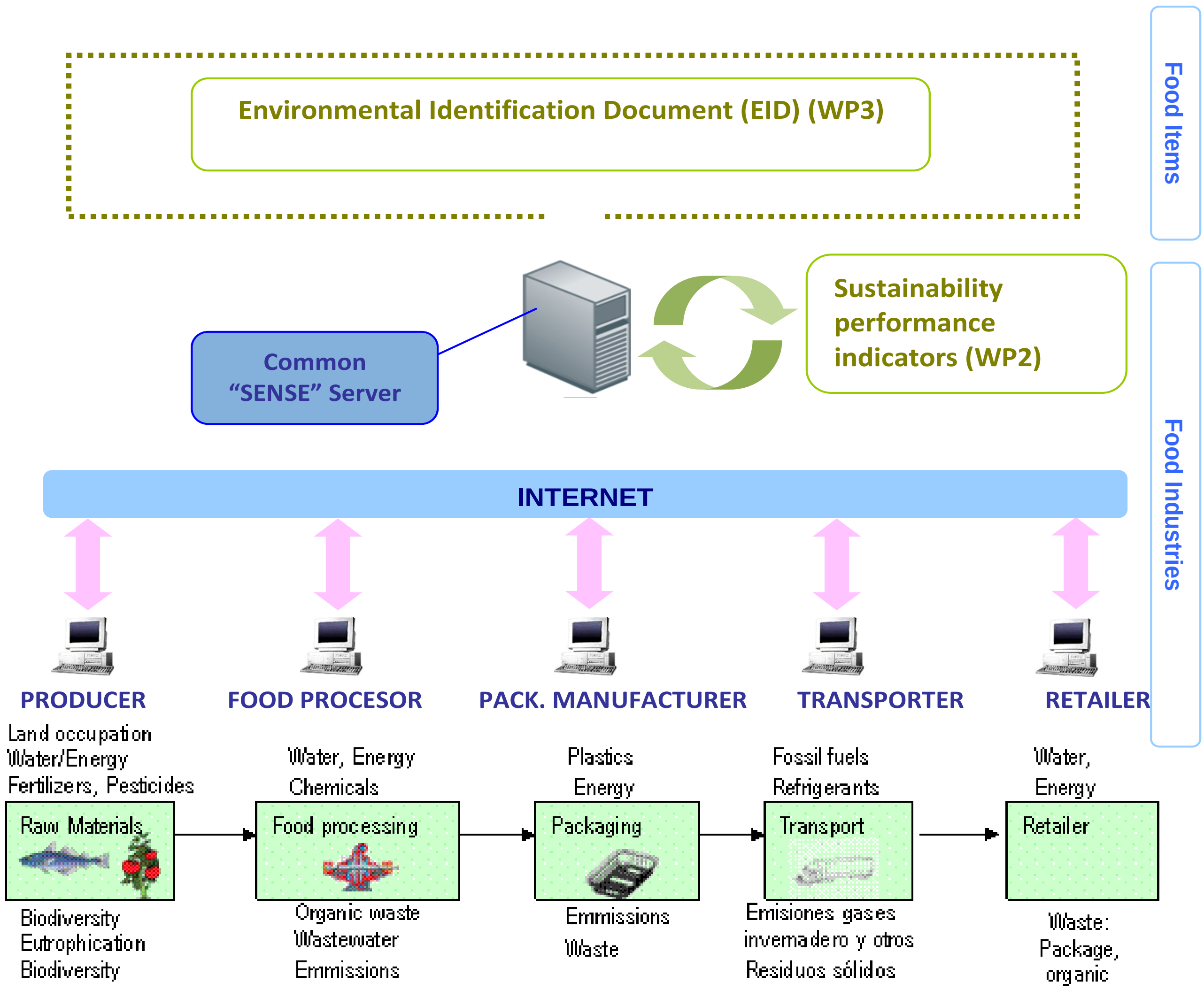


Figure 2. Proposed web-based system to enable users to input key environmental information from different supply chain stages in food chain following a data traceability approach.

77. Life cycle assessment of rapeseed and sunflower oils

Laureen Badey[1,*], Natie Lahitte, Francis Flenet[2], Fabrice Bosque

[1] ITERG, French Institute of Fats and Oils, [2] CETIOM, Technical Center for Oilseed Crops and Industrial Hemp, [*] Corresponding author. E-mail: *l.badey@iterg.com*

The aim of the study is to provide all necessary information to fit to the French national experimentation of environmental labeling for two standard oils: sunflower oil and rapeseed oil. The results of this study are a relevant overview of all sunflower and rapeseed oils produced in France, and are usable as standard values for both producers and users of vegetable oil. Industrials of oil sector use these values to compare to their own process values and to evaluate the improvement due to their ecodesign strategy. For example, the use of a biomass boiler, the reduction of packaging, different choices for the suppliers of the seeds lead to a lower LCA score.

The complete life cycle of oils has been studied from the seeds production to the end of life of the packaging. Only storage and use have been excluded from the study because of a lack of data. The life cycle inventory is illustrated in Fig. 1. In this graph, the cells in dotted line are the steps excluded from the study. All the other ones are included. An energetic allocation has been done to the co-products (with a grey frame in the graph).

The data were collected from different industrial sites belonged to Sofiproteol group that illustrate the diversity of all the French crushing, refining and packaging sites. About 4.5 million seeds were crushed in 2010 by Sofiproteol group, which represents 80% of French crushing. A focus has been made on the impacts of crushing and chemical refining. The industrial data are specific to edible oils. Note that refining of edible oil (chemical refining) differs from the refining of oil used for biofuels (physical refining).

For the agricultural step, data has been gathered from ADEME (2010) on first generation of biofuels used in France and has been rounded off by CETIOM expertise (water consumption). Life cycle inventories (LCI) come from the LCA database ecoinvent.

The indicators used are congruent with the French national experimentation of environmental labelling: greenhouse gases (GHGs) and water consumption. The most important point in the results (Table 1) is that the agricultural step is responsible of most impacts on the studied indicators (from 58-71% of GHG and 47-73% of water consumption). The other steps that contribute the more are the industrial step, the transport and the packaging.

References

ADEME - Direction Production et Énergies Durables, 2010. Analyses de cycle de vie appliquées aux biocarburants de première génération consommés en France. Final report. 236 pages.

ADEME, AFNOR, 2011. BPX 30-323 - Principes généraux pour l'affichage environnemental des produits de grande consommation - Partie 0: principes généraux et cadre méthodologique.

Shonfield, P.K.A., Dumelin E.E., 2005. A life cycle assessment of spreads and margarines. Lipid Technology, 17: 199-203.

Schmidt, J, 2007. Life cycle assessment of rapeseed oil and palm oil, Ph.D. thesis.

Wightman. P., Eavis, R., Batchelor, S., Walker, K., Bennett, R., Carruthers, P., Tranter, R., 1999. Comparison of rapeseed and mineral oils using life-cycle assessment and cost-benefit Analysis. OCL – Oleagineux, Corps Gras, Lipides, 6: 384-388.

Table 1. Results of LCA on rapeseed and sunflower oils

		Rapeseed oil		**Sunflower oil**	
Indicator	**Unit**	**Refining oil**	**Packaged oil**	**Refining oil**	**Packaged oil**
GHG emissions	g CO_2 eq	127	154	89	112
Water consumption	litres	0.7	1.0	1.7	1.9

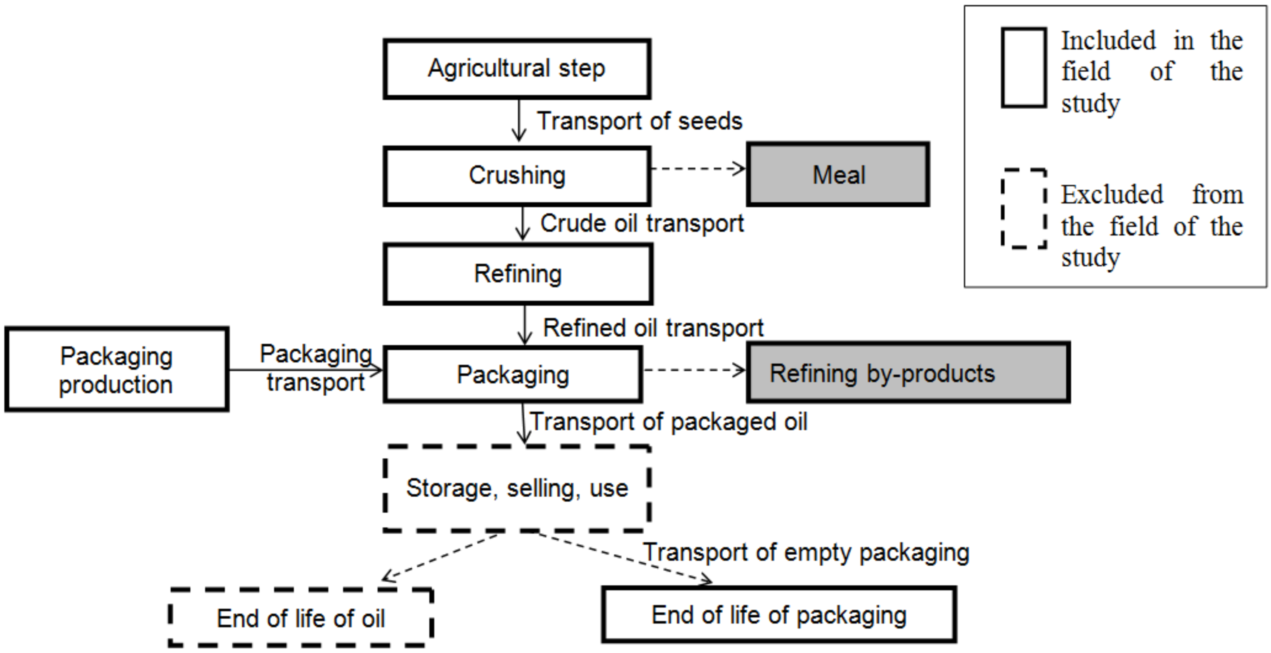


Figure 1. LCI considered for the LCA of rapeseed and sunflower oils.

78. Effect of carbon footprint label on food purchase behaviour

Aki-Heikki Finér[1,*], Marika Kelokari[1], Jussi Nikula[2], Mira Povelainen[1]

[1] Raisio Group, [2] WWF Finland, [*] Corresponding author. E-mail: *aki.finer@raisio.com*

Consumer interest on products' environmental profile is constantly increasing. Several eco-labels are published worldwide to inform consumers about products' environmental performance. Raisio Group, an international plant-based foods and feeds company, was one of the first companies to introduce a carbon footprint label for food products in 2008. Since, Raisio's carbon footprint label has been revised in compliance with customer feedback. Nowadays, Raisio has labelled more than 30 of its own products and the label is also in use in more than 15 other companies' food products including poultry and honey products.

A consumer web survey was organised together with Raisio Group and WWF Finland in July 2011 in Finland. The survey was organised to determine which aspects Finnish people value when making decisions concerning food. Information about the survey was spread through social media, Raisio's consumer newsletters and WWF Finland's newsletter. Most important questions were how Finnish consumers take food chains different sustainability aspects into account in their purchase decisions, the market penetration of Raisio Group's carbon footprint label and the relation of carbon footprint and other factors when making purchase decision.

The respondents were assumed to be environmentally conscious or aware of current discussion about food environmental impacts and sustainability as a whole in Finland. The results of the survey show that approximately 40% of 4960 respondents have seen Raisio Group's carbon footprint label on a food product package. Almost 54% of the respondents stated that a carbon footprint label should be mandatory for food products. More than 60% suppose that mandatory carbon footprint label would have effect on purchase decision. Other key factors in decision making are food products purity and safety, healthiness and price.

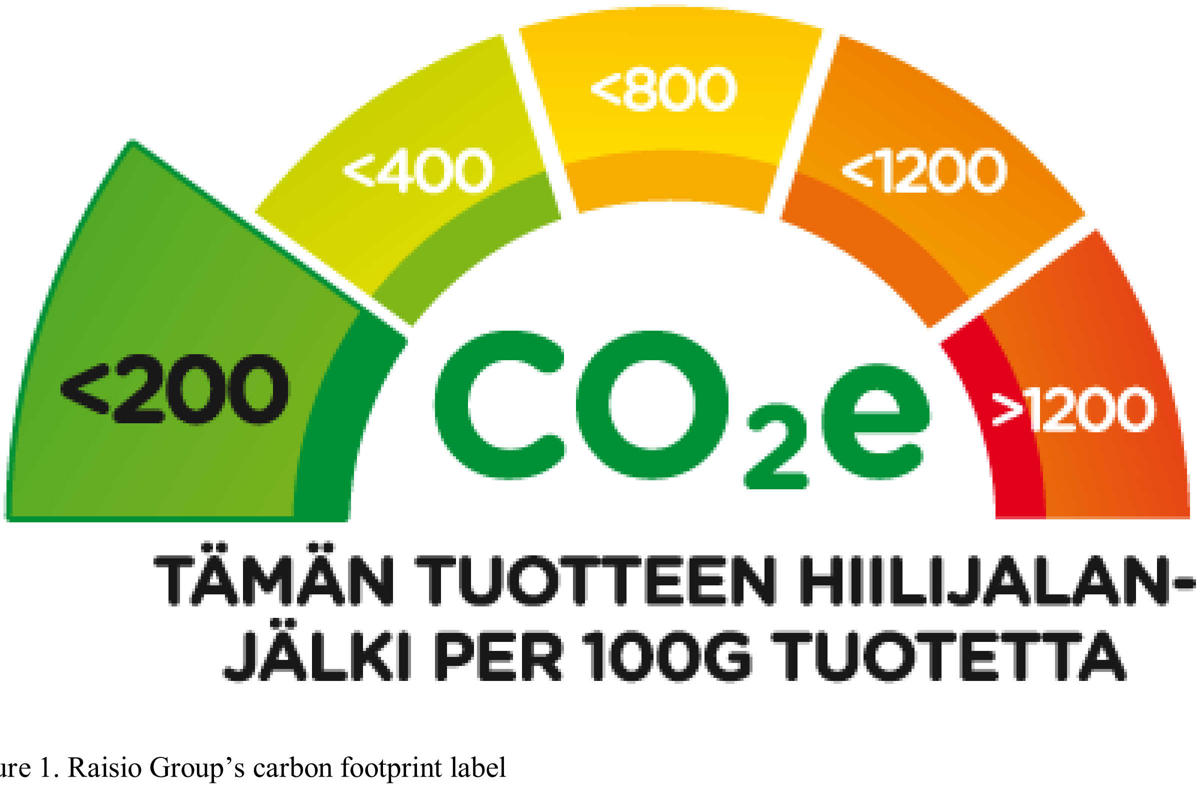


Figure 1. Raisio Group's carbon footprint label

79. Bittersweet comparability of carbon footprints

Silvia Marton

myclimate – The Climate Protection Partnership, Zürich, Switzerland, E-mail: *silvia.marton@myclimate.org*

Comparability of results from life cycle assessments is a critical topic, especially when results are used for communication and product labelling. The main sources of variation are (1) different system boundary definitions, (2) data handling and assumptions, (3) different databases for background data (Kägi et al. 2011). In our study of three types of chocolate, the carbon footprint is performed by the same person in order to eliminate the first and second cause for variation. The carbon footprint was performed using two different tools, Footprint Expert (FPX) of the British Carbon Trust Footprinting Company Ltd., and EMIS 5.6 from Carbotech AG. Both tools provide an integrated database for background data, the FPX Reference Database 3.3 and ecoinvent 2.2, respectively. The calculation of agricultural products was performed with the FPX Crop Calculator. In the EMIS model, direct emissions were calculated following the methodology used in ecoinvent (Nemecek & Kägi, 2007) and the IPCC 2006 Guidelines. The aim of this study is to find out if the results of the footprints are comparable or not. The study is performed according to PAS 2050. Primary data were collected for cocoa and sugar cane farming and the production stages. Secondary data were mainly taken from the databases of the tools.

The study shows that the final results from both tools are in the same magnitude order, with differences of less than 15 per cent. Larger differences could be observed for the single production stages (Fig. 1 and Table 1). The calculated greenhouse gas (GHG) emissions from cocoa beans were higher using the ecoinvent methodology compared to those from the FPX Crop Calculator. For sugarcane, the result was different, as the emissions calculated with the ecoinvent methodology were lower than those from the FPX Crop Calculator. GHG emissions from transport were higher when calculated with FPX compared to the transport emissions based on ecoinvent. Especially for sea freight, ecoinvent emission factors are much lower, which is in line with the results from other studies analysing sea freight (Emanuelsson et al. 2010).

Both tools, FPX and EMIS, have their advantages and disadvantages. FPX is a tool with a strict guideline and especially designed for carbon footprinting. Therefore it can be used by non-professionals, allowing them to calculate carbon footprints of good quality. The aim of FPX is to standardise carbon footprinting in order to enable comparable results, even if footprints are performed by different organisations. However, the tool is limited to carbon footprinting. EMIS is a tool for life cycle assessments (LCA). Therefore it can be used for other impact categories and includes more functionality such as input-output analyses or uncertainty analyses. It allows a more differentiated analysis of products. However, as it is more flexible in its use, comparability of studies performed by different people is lower.

Often product carbon footprint analyses are conducted mainly for marketing reasons. It has to be emphasised that the communication of results on product labels is a difficult topic. Indicating an exact number may be misleading to customers, as it pretends an accuracy that cannot be achieved by current methodologies. However, standardised tools like FPX together with an accredited certification body do provide comparable results that may help customers to choose climate friendly products.

References

Emanuelsson, A., Skontrop Hognes, E., Ziegler, F., Sund, V., Winther, U., Elligsen, H., 2010. Life cycle approach to Norwegian seafood logistics – new ways to model food transports, in: Notarnicola B., Settanni E., Tassielli G. Giungato, P. (Eds.), proceedings of LCAfood 2010, Bari, Italy

Kägi, T., Wettstein, D., Schmid, H., Azapagi, A., Amienyo, D., Contreras, A., Sanchez, M., Villegas Costa, L., Ménard, J.F., Bécaert, V., Kanzaki, M., 2011. LCA Comparative Study on Beer. 5th PCF World Summit, Zurich, 7-8 April 2011

Nemecek, T., Kägi, T., 2007. Life Cycle Inventories of Agricultural Production Systems. ecoinvent report No. 15, Zurich and Dübendorf, Switzerland.

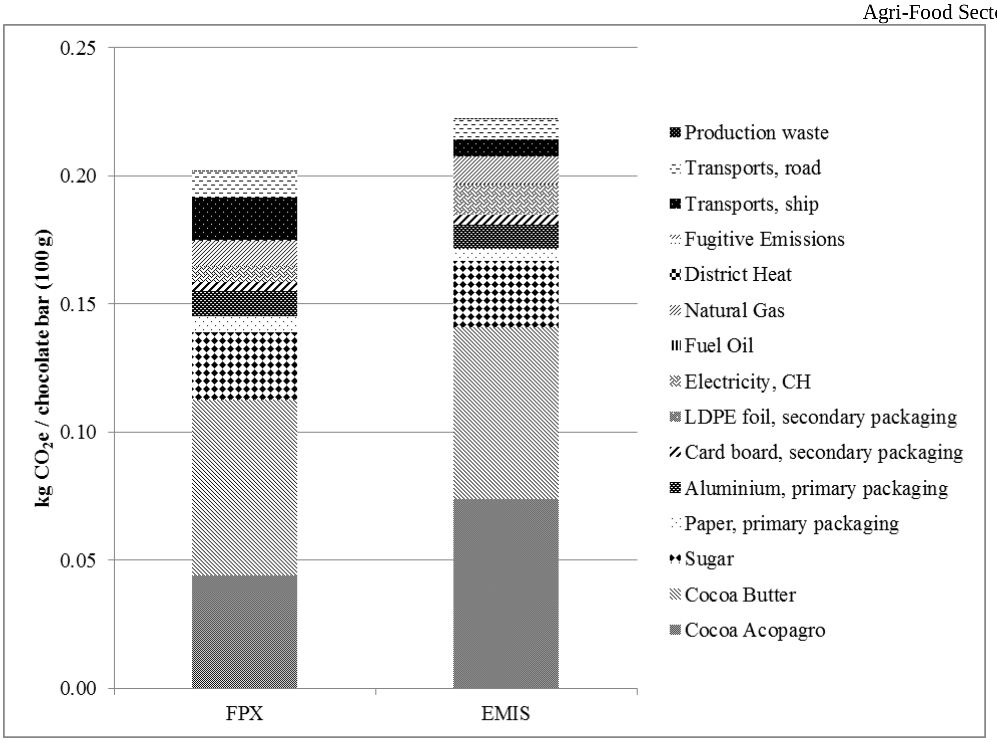


Figure 1. GHG emissions from the chocolate „Peruvian chocolate 60%“, calculated with EMIS and FPX.

Table 1. Emission sources and differences between EMIS and FPX results.

Emission source	**Difference of EMIS result compared to FPX result**
Cocoa Acopagro	+68.0%
Cocoa butter	-2.3%
Sugar	-2.3%
Paper, primary packaging	-23.0%
Aluminium, primary packaging	-5.7%
Cardboard, secondary packaging	+10.5%
LDPE foil, secondary packaging	+25.0%
Electricity, CH	+93.5%
Fuel oil	-0.5%
Natural gas	+9.3%
District heat	-
Fugitive emissions	-
Transports, ship	-59.8%
Transports, road	-22.5%
Production waste	+56.8%
Total	**+10.3%**

80. Life cycle assessment as a basis for eco-labelling of fish food products

Erwin M. Schau[1,2,*], Annik M. Fet[2], Harald Ellingsen[3], Matthias Finkbeiner[1]

[1] Department of Environmental Technology, Technische Universitaet Berlin, Berlin, Germany, [2] Department of Industrial Economics and Technology Management, Norwegian University of Science and Technology (NTNU), Trondheim, Norway, [3] Department of Marine Technology, NTNU, Trondheim, Norway, [*] Corresponding author. E-mail: *erwin.schau@tu-berlin.de*

Life cycle assessment (LCA) is a valuable tool for documenting environmental performance of food products. This is also true for fish food products – a commodity providing income for millions and food for billions of people worldwide (FAO Fisheries Department, 2010) – even though most effort has been invested in the sustainable harvesting of fish resources. The life cycle management in the food industry is mainly driven by policy makers, consumers and the industry itself on the emerging problem of climate change. In the fish industry, there is an urgent need for strategies, tools and communication with the market that emphasise the whole life cycle of the fish food product from catch to consumption. Taking this into account, carbon footprint declaration (after the planned ISO14067 (ISO/DIS 14067, 2012)), the coming European environmental footprint of products (European Commission, 2012), and the Environmental Product Declarations (EPD) - following the ISO14025 standard (ISO 14025, 2006) - are all suitable for communicating results from food products LCAs. The EPD, mainly developed for business to business communication, can also be used for communication to the other important stakeholders, like environmental organisations and consumers. However, the special challenges with declining fish resources need to be taken into account. This contribution presents results from a PhD-project on life cycle assessment of the fish food product with emphasis of the fishing phase (Schau, 2012). The results from the LCA of Atlantic herring landed in Norway and consumed in Germany, shows that the fishing vessel, and especially the diesel use is the main contributor to most impact categories investigated. Besides from presenting the LCA results, the EPD also gives information on the status of the fish stock the product comes from. In addition, this contribution investigates how life cycle environmental impacts from fish food products can be presented in the carbon footprint declarations and in the coming European environmental footprint of products.

References

European Commission, 2012. Environmental footprint of products. Retrieved 24 Jan 2012 from http://ec.europa.eu/environment/eussd/product_footprint.htm.

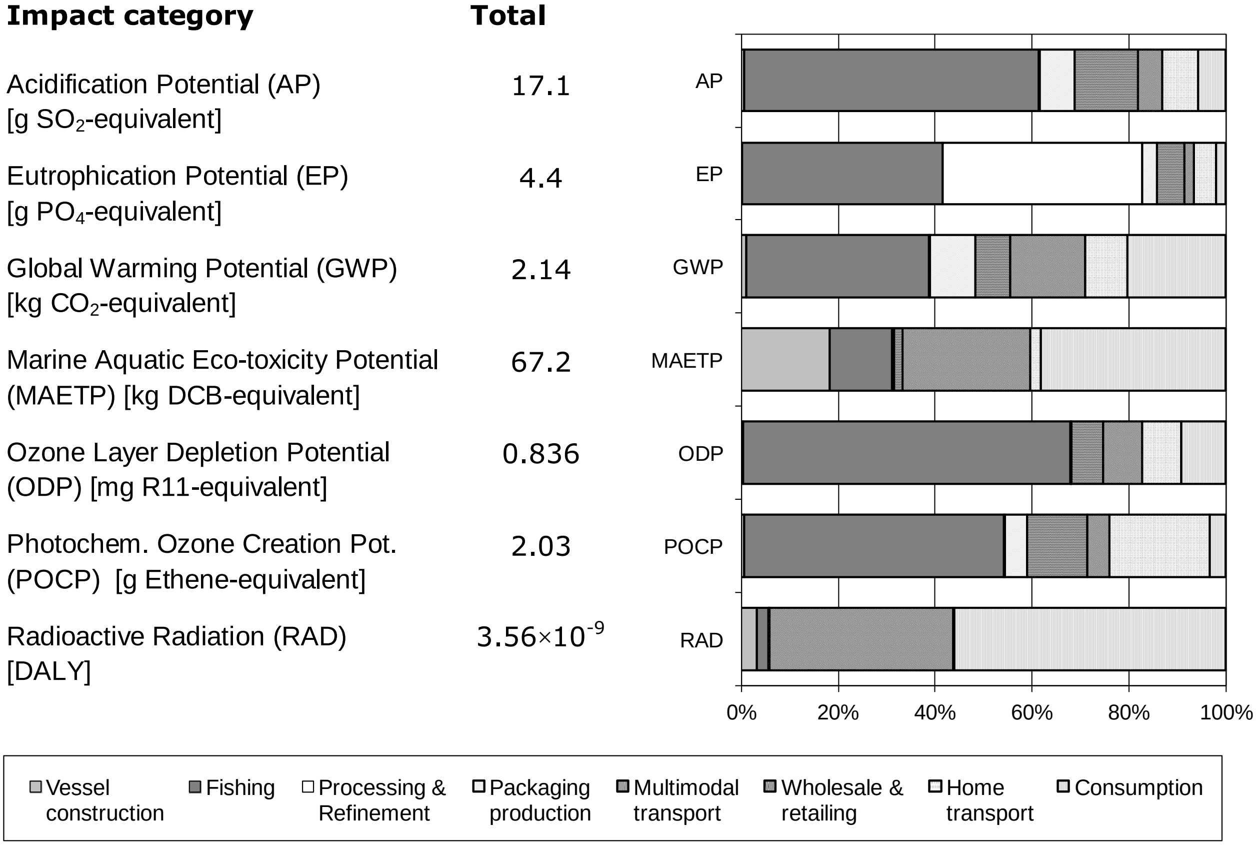
FAO Fisheries Department, 2010. The State of World Fisheries and Aquaculture - 2010. Rome, Food and Agriculture Organization of the United Nations.

ISO 14025, 2006. Environmental labelling and declarations - Type III environmental declarations - Principles and procedures (ISO 14025:2006). Geneva, ISO.

ISO/DIS 14067, 2012. Carbon footprint of products - Requirements and guidelines for quantification and communication. Berlin, Secretariat DIN**:** 52.

Schau, E.A.M., 2012. Environmental life cycle assessments of fish food products with emphasis on the fish catch process. Trondheim, Norway, NTNU, Norwegian University of Science and Technology.

Impact category	**Total**
Acidification Potential (AP) [g SO_2-equivalent]	17.1
Eutrophication Potential (EP) [g PO_4-equivalent]	4.4
Global Warming Potential (GWP) [kg CO_2-equivalent]	2.14
Marine Aquatic Eco-toxicity Potential (MAETP) [kg DCB-equivalent]	67.2
Ozone Layer Depletion Potential (ODP) [mg R11-equivalent]	0.836
Photochem. Ozone Creation Pot. (POCP) [g Ethene-equivalent]	2.03
Radioactive Radiation (RAD) [DALY]	3.56×10^{-9}



Abbreviations: DCB : 1,4-dichlorobenzene
DALY: disability-adjusted life year

Figure 1. LCA results from the Atlantic herring (*Clupea harrengus*) case study. The fishing life cycle phase dominates several impact categories (Schau, 2012).

81. Agrifood products: a comparison of existing carbon footprint systems in the world

Stéphane Le Pochat*, Evelyne Lauren, Roberto Bellino

EVEA, 56 Bd de la Fraternité, 44100 Nantes, France, * Corresponding author. E-mail: *s.lepochat@evea-conseil.com*

In 2011, EVEA and Savin Martinet Associés carried out a study on behalf of the French Food and Agriculture Ministry (MAAPRAT 2012). The aim of this study was first to compare the different product carbon footprint (PCF) systems of agrifood products around the world, and second to give a risk analysis of competing distortion from a world trade point of view.

Thirty-eight PCF systems were checked off. These systems are disseminated worldwide: Europe (France, Germany, England, Sweden, Switzerland, etc.), North America, Asia, Australia and New Zealand. Distinctions were made firstly by system category (PCF, labels, or tools), and secondly by origin (public or private, retailer). From these 38 systems, 14 of them providing existing PCF of agrifood products on the market were deeply analysed, 10 of which in Europe and 2 in Asia. This consisted in reporting and analysing general information (country, owner of the system, number of committed enterprises, number of PCF labelled products, etc.) and more specific information about the structure of the PCF system such as existing PCR, verification process, tools for computing, underlying database, etc. Furthermore, a comparison of PCF of several products was carried out. Around 300 PCF of products were reported from the 14 analysed systems and some detailed comparisons were made for 9 specific products (milk, wine, ham, bread, rice, olive oil, yogurt, potatoes, and French beans) both within a same system (intra-comparison) and between different systems (inter-comparison). Table 1 shows an example for the comparison of several milk footprints.

Each time, interpretation has been conducted from these comparisons confronting on one hand the PCF figures and on the other hand available information related to these PCF (calculation rules, allocation, etc). Three criteria were defined: level of available information, estimated level of methodological divergence, objective comparability. For each product and each PCF system, this analysis led to identify the main influencing parameters (packaging, organic production or not, origin, varieties, recipe, etc.) and their respective level of influence on the PCF result (as shown on table 1).

Moreover, a juridical analysis was carried out in order to confront the PCF systems to the regulation rules of the 4 main following international organisations: WTO, OECD, European Union, and FAO. Indeed, regarding the specific French context about environmental labelling, the issue was: can a mandatory environmental labelling system for consumer products be authorised on the French market by international organisations?

Regarding the risk of a competing distortion that would be a case of no-compliance with the rules of the world trade, the main conclusions from this juridical focus are that such systems cannot be imposed as mandatory by a country except if they fully respect three conditions:

- Transparency about the elaboration process of the standard (methodology for calculation): the elaboration process must reach an international consensus and must be opened to all stakeholders.
- No-discrimination about origin: results (PCF) must not lead to discriminating products based on their production process and geographic origin.
- Proportionality of the means regarding the claimed objectives (encourage a sustainable consumption): such a system must not engage costs that could be judged to be excessive for companies, in particular SMEs.

The study leads to the following conclusions:

- The different PCF systems present a great diversity regarding their respective features and in particular regarding the applied methodologies
- There is a huge variability of the PCF of products, and some methodological divergences have been demonstrated. Furthermore, very little information is usually available and most of the systems are absolutely not transparent and thus are not compliant with the only internationally recognised standard (ISO 14020). All these items lead to the conclusion that for a same product different PCF calculated within different systems are objectively not comparable. This can be problematic for customers when different systems co-exist in the same market (eg. in France).
- For now none of the existing labelling systems could be considered compliant with the rules of the world trade for a state to make it compulsory.

References

Bolwig, S., Gibbon, P., 2009. Emerging product carbon footprint standards and schemes and their possible trade impacts. Report, 2009, Technical University of Denmark, DTU Climate Center.

MAAPRAT, 2012. Comparaison des ACV et affichages carbone utilisés sur les produits agroalimentaires – Etude technique et juridique. Report, EVEA and Savin Martinet Associés, February 2012.

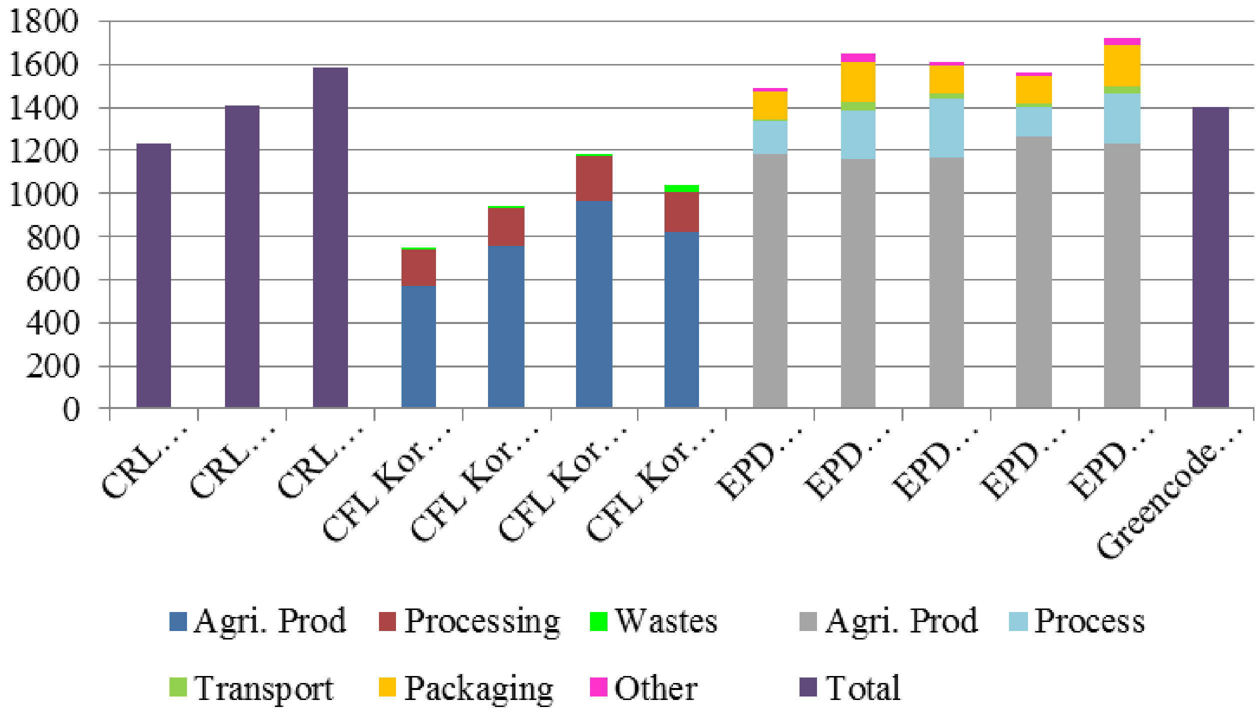


Figure 1. Comparison of milk carbon footprints (g CO_2 eq.) per 100 cl

Table 1. Influence of parameters on carbon footprints.

Parameter		Product/ PCF system	Level of influence
Packaging	Sale unit	Ham / Casino (France)	Strong
		Yaourt / Casino (France)	Strong
		Ham / CFP (Japan)	Weak
		Rice / CFP (Japan)	Strong ?
		Potatoes / CRL Tesco (UK)	Weak
		Milk / EPD® (Sweden)	Important
	Storage (ambient temperature or frozen)	Bread / EPD® vs. others	Weak
		French beans / Greenc. Info® (France)	Strong
Type of production	Organic vs conventionnal	Wine / H. de Carbono (Spain)	Strong (up)
		Wine / EPD® (Sweden)	Important (down)
		Olive oil / H. de Carbono (Spain)	Strong (down)
		Milk / EPD® (Sweden)	Weak
		Milk / CFL (Korea)	Strong (up)
		Rice / Climatop (Switzerland)	Strong ?
	Dry vs. immersed	Rice / Climatop (and CFP Japan ?)	Strong

82. Finnish carbon footprint protocol “Foodprint” for food products

Juha-Matti Katajajuuri*, Hannele Pulkkinen, Hanna Hartikainen, Kristoffer Krogerus, Merja Saarinen, Frans Silvenius, Kirsi Usva, Heli Yrjänäinen

MTT Agrifood Research Finland, Responsible Food Chain – Better Consumer Well-being, FI 00790 Helsinki, Finland * Corresponding author. E-mail: *juha-matti.katajajuuri@mtt.fi*

In the Foodprint - research project harmonised methodology for assessing carbon footprints of food products was developed in collaboration with the Finnish food sector in 2009-2012. Many international standards, protocols and guides are published but no common approved standard nor communication method evaluating food products’ climate impacts is available. In addition, the published ones are too generic and they do not give practical instructions for companies to produce comparable carbon footprints.

However, international standardisation, developments and best practices on evaluating climate impacts of the food products’ entire life cycle were taken into account when national specific guidelines were prepared. Some of the most challenging issues tackled in the methodology and protocol work are described in this paper.

There are many situations in assessing carbon footprints where decisions are often done case by case. These issues are critical as they affect comparability and the magnitude of carbon footprint results. Attributional approach was selected as s starting point of the protocol. The methodology development work was carried out as an iterative process between research, companies and other stakeholders to ensure practical methodology and implementation.

To harmonise methodology and future carbon footprint assessments, detailed instructions were given to different life cycle phases and many clear requirements have been established. All life cycle phases from raw material extraction to waste treatment shall generally be included. Different requirements were also made for cultivation and for animal production. Cut-off rules are also applied in the methodology and more detailed instructions for their use are given. Capital goods are excluded from the system boundaries.

Present data quality requirements, particularly requirements on primary data, in current and draft international guidelines are seen insufficient. Therefore, in the Foodprint protocol more detailed requirements were applied separately for each life cycle phase. Detailed instructions were given to each life cycle phase whether data shall be collected directly from a supply chain, or gathered from national statistics, databases etc., and which are adequate data sources. The intention was to harmonise the data requirements from agricultural phase in the protocol with the fairly comprehensive activity data, which is already collected by primary producers for other purposes in Finland.

New updated emission factors were also developed. National emission factors for N_2O emissions from agricultural soils derived from field measurements were launched in the project to describe better national circumstances. This means that new default will give considerable higher emissions to grains and vegetables (outdoor) grown in open fields due to the northern conditions.

Another area of improvement and generation of defaults were national emission factors for different electricity production types. The protocol proposes that specific emissions factors related to the actual electricity supplier should be used. This means that when the production profile of the supplier is known (as in Finland is the case by law), the new national defaults for different production types shall be used.

Different existing methodologies to include land use changes, especially deforestation, were also analysed based on the Finnish case studies. Emissions resulting from land use change were proved to have a large impact on the final carbon footprint of food products. Thus they shall be included in the assessment, and presented separately from total results. The methodology and some practical guides are presented in the protocol and additional material of the project.

It is seen that in the near future, when climate impacts are understood and tackled by companies and they have suitable tools for that, this also directs proactive companies to develop more comprehensive methods, to consider issues such as water and nutrient footprints.

83. How to measure sustainable agriculture? The GLOBAL 2000 adaptive sustainability assessment approach for agricultural products

Martin Wildenberg*, Tanja Altaparmakova, Kewin Comploi, Dominik Frieling, Lydia Matiasch, Helmut Burtscher

Umweltforschungsinstitut GLOBAL 2000 / Friends of the Earth Austria, Neustiftgasse 36, 1070 Vienna, Austria, * Corresponding author. E-mail: *martin.wildenberg@global2000.at*

Eco-labelling has been suggested as an approach to manage towards a more sustainable world by several authors and organisations (e.g. de Snoo 2006, Bruce & Laroiya 2006, Rigby et al 2001, UNDP (CSD) 1996). Although approaches to "measure" sustainability have often been criticised we follow the opinion of Gomez-Limon & Sanchez-Fernandez (2010) that the "*design and use of such indicators can be extremely useful in that they force those involved in the discussion of sustainability to identify the key aspects of sustainable agriculture and to assign weights to them.*" In such a context the often very general and theoretic discussions about sustainability are confronted with real world practices and problems and are requested to come up with workable solutions and improvements.
This poster presents the GLOBAL 2000 adaptive sustainability assessment approach (ASAP), with which the environmental performance of agricultural products is measured from field to shelf. Also first product specific results and lessons learned during the implementation phase will be presented.
The aim of the approach is to arrive at a comprehensive understanding of the environmental impacts of a certain agricultural product and the connected life cycle. Furthermore, it strives to set incentives for farmers to adopt a more sustainable production mode and to help consumers make deliberative consumption choices, by informing them about the environmental impacts of products.
Five field-level based indicators (N-balance, P-balance, humus-balance, pesticide use and energy intensity) and five based on "material input per service unit" (MIPS) indicators (carbon-footprint, biotic and abiotic material input, water input and area used) are used. We calculate the indicators based on date provided by each producer and company along the production chain. The approach uses a stepwise process that explicitly involves stakeholders in the refinement and adaptation of monitoring and benchmarking and serves as a discussion and knowledge transfer arena.
Our method is applied in the context of the REWE International AG, GLOBAL 2000 and Caritas Sustainability Program for Fruits Vegetables and Eggs and is used for the REWE label Pro Planet (www.pro-planet.at) in Austria. This program provided the necessary frame and infrastructure to further develop the methodology and tools in an applied context.

References

Bruce C. and A. Laroiya, 2007. The Production of Eco-Labels. Environmental and resource Dynamics, 36: 275-293.
de Snoo, G. R., 2006. Benchmarking the Environmental Performances of Farms. The International Journal of Life Cycle Assessment, 11:22-25.
Gomez-Limon, J.A., G. Sanchez-Fernandez, Empirical evaluation of agricultural sustainability using composite indicators, Ecological Economics, 69: 1062-1075.
Rigby D., P. Woodhouse, T. Young, M. Burton, 2001. Constructing a farm level indicator of sustainable agricultural practice. Ecological Economicsm 39: 463-478.
UNDP (CSD), 1996, Eco-labeling United Nations Commission on Sustainable Development. Available at: http://www.un.org/documents/ecosoc/cn17/1996/background/ecn171996-bp8.htm

84. Product carbon footprint for a line of production of cookies

C. Naranjo[1,*], O. López[2], C. Montoya[2], J. Leal[1]

[1]Gaia Environmental Services, Medellín Colombia, [2]NOEL Cookie Company, Medellín Colombia, [*] Corresponding author. E-mail: *cnaranjo@gaiasa.com*

A line of cookies, from a Latin-American company, has been analysed in their life cycle, taking the impact on GHG emissions. A total of 16 references were analysed. The methodology used for the study was the PAS2050: 2008. The study includes the GHG emissions from materials extraction, transportation, manufacturing, packaging, use and disposal of packaging.
The study uses the Umberto for carbon footprint software ® v 1.2, with ecoinvent 2.2 datasets. Primary data from process are included, taking energy consumptions for each stage, process and mass efficiency in lamination process and baking process.
The cookies included in the study are healthy line with less fat and less sugar content. This cookies uses raw materials like wheat flour, bran, oat flour, corn flour, palm oil, sugar and water. The baking uses natural gas. The packaging of the product uses PET/BOPP/Met, PET/BOPP, Folding boxboard and Corrugated cardboard. In the study the distribution is in truck and distances are calculated with the sales registers for 2011. See Fig. 1 with the system for assessment.
For a cookie raw materials account for 55-68% of the carbon footprint associated mainly with wheat flour, followed by product packaging that represents between 17 and 30%, the process is in between 9 and 17%. The carbon footprint of each cookie depends of its formulation (Fig. 2).
In the analysis, it appears that within the same line of cookies can have different levels of baking and variables as the percentage of waste in the lamination and baking and drying curve for the product, become representative in the product's carbon footprint, due to increased raw material requirements.
Having the overall product carbon footprint, it was multiplied by the total production, and it allows calculate the Product Carbon Footprint contribution for the line of cookies (Fig 3). The Honey reference has 41% of the total carbon footprint of the entire line; it is because it has the major sales.
The wheat is the most important raw material for the cookie production and for the product carbon footprint too. The importance of this will be associated to the risk of one company against climate change.
Change traditional wheat for organic one, was evaluated, but the organic production is small and it cannot produce the enough raw material needed for the production of the company. Otherwise, the quality requirements of the organic wheat do not meet demands of the product.
For the products uses different packaging with aluminium and it increases the carbon footprint. The folding box board and the corrugated cardboard are very important in the emissions for the product.

References

Carbon Trust, DEFRA, BSI, 2008. PAS2050:2008. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. BSI: London, 43p.
ISO. ISO 14040:2006. Environmental management -- Life cycle assessment -- Principles and framework
ISO. ISO 14044:2006. Environmental management -- Life cycle assessment -- Requirements and guidelines
Bimpre, M. 2006. Life Cycle Assessment of the production of homemade and industrial bread in Sweden. KTH. 25 pages.

Table 1. Carbon footprint contribution per life cycle stage (%).

Life Cycle Stage	Raw Material	Processing	Packaging	Distribution	Disposal
Sesame SKU1	63.7	17.0	16.5	2.4	0.3
Sesame SKU2	61.2	16.2	19.2	3.0	0.3
Oatmeal Chocolate SKU1	65.0	10.0	22.1	2.6	0.4
Oatmeal Chocolate SKU2	62.2	10.1	25.2	2.1	0.3
Oatmeal and Red Berries SKU1	58.5	11.5	26.5	3.0	0.4
Oatmeal and Red Berries SKU2	56.5	11.0	29.2	2.9	0.4
Oatmeal Granola SKU1	59.9	12.4	24.4	2.9	0.4
Oatmeal Granola SKU2	54.9	11.2	30.5	3.0	0.4
Chocolate Bs. 6x2	63.3	8.6	25.1	2.6	0.4
Fruits and Cereals	67.2	11.1	18.7	2.9	0.1
Grain Fusion SKU1	63.2	15.4	18.2	2.8	0.3
Grain Fusion SKU2	66.5	10.6	19.3	3.1	0.6
Honey SKU1	67.0	11.4	18.5	2.7	0.3
Honey SKU2	67.6	11.5	17.6	2.7	0.5
Vanilla	60.2	8.9	26.7	3.7	0.5
Yogurt Strawberry Bs x 6	55.1	14.3	27.0	3.2	0.5

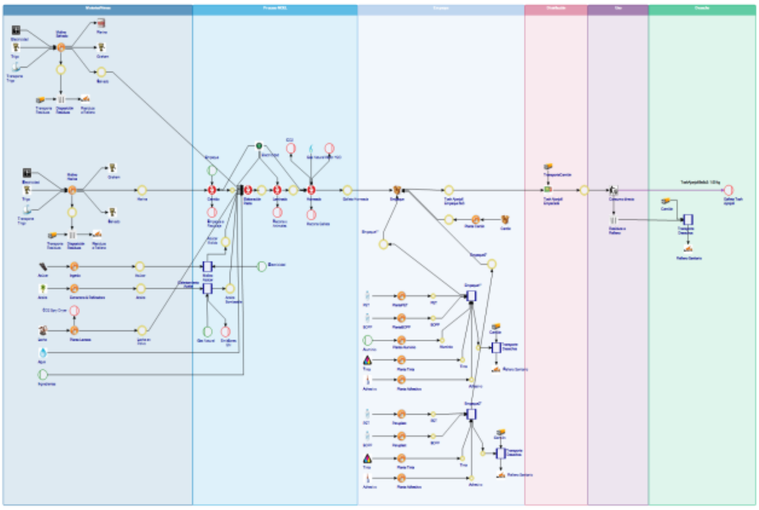


Figure 1. System diagram for cookie production

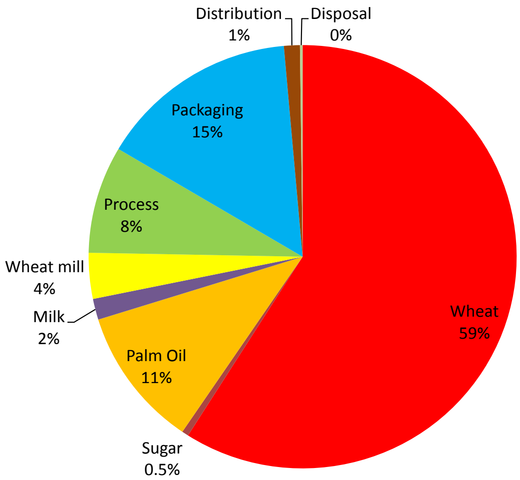


Figure 2. Example for raw material contribution in the product carbon footprint

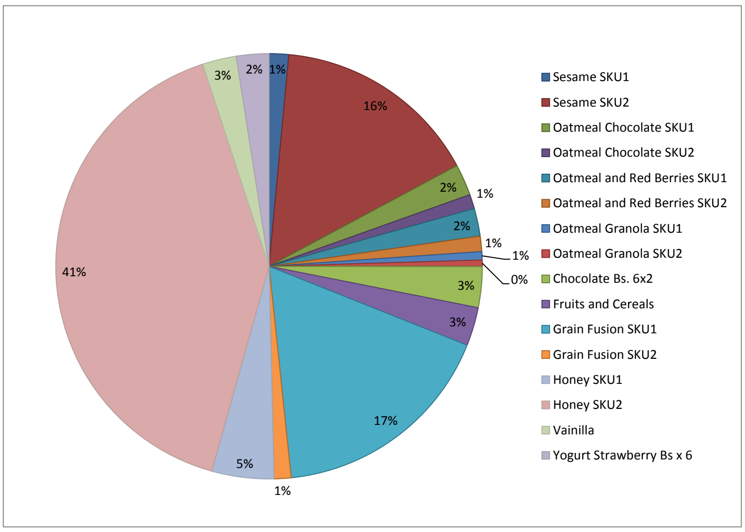


Figure 3. Product carbon footprint contribution by reference.

85. LCA in organic and conventional product comparison: a review

Matthias S. Meier[1,*], Christian Schader[1], Franziska Stoessel[2]

1 FiBL - Research Institute of Organic Agriculture, Ackerstrasse, CH-5070 Frick, Switzerland, [2] ETH Zurich, Institute of Environmental Engineering, Schafmattstrasse 6, CH-8093 Zurich, Switzerland, [*] Corresponding author. E-mail: *matthias.meier@fibl.org*

Life cycle assessment (LCA) is an important assessment tool to evaluate the environmental impacts of agricultural products. One focus of assessments within agriculture has been the comparison of organic and conventional production systems to evaluate whether organic or conventional agricultural products are more environmental friendly. In different studies it has been shown that organic farming has benefits for the environment when focusing on the farming practice (e.g. Mondelaers et al. 2009). However, when using LCA to calculate impacts on a per unit product basis organic products show higher impacts for certain categories mostly due to lower yields. If they are calculated per area*time unit basis the impact is lower due to lower inputs of agricultural means. Though, there are studies showing a better performance of organic products for both functional units (e.g. the global warming potential of milk in Cederberg and Mattsson (2000) or Hörtenhuber et al. (2010)). Basically, there are two reasons for contradicting results of LCA studies comparing organic versus conventional products: Firstly, real differences in performance of organic and conventional products (from farm to farm and supply chain to supply chain) secondly, methodological artifacts within LCA. To elucidate these presumed reasons we reviewed LCA studies on organically and conventionally grown products of different product groups (fruits and vegetables, cereals, dairies and meat), systematically analysing the parameters listed in Table 1.

We identified shortcomings on different levels that impair the comparison of LCA studies between organic and conventional products. The most stringent is that system-specific characteristics of organic agriculture are not fully reflected on the inventory level, which can lead to bias in certain impact categories (e.g.: climate change, eco-/human toxicity, photo-oxidant formation, acidification and eutrophication). This is either due to a lack of data or due to insufficient data quality. For example calculations of nitrous oxide emissions do not consider the different transformation processes of organic fertilisers (which act mainly via the soil N pool) but treat them as mineral fertilisers; heavy metal contents of manure from organic farms are based on measurements of manure from conventional agriculture; or carbon sequestration usually is not included or the interdependence of the C- and N-fluxes in soils is not reflected. Further, currently used allocation rules within the livestock sector (milk and meat) seem to bias LCA results from organic and conventional production systems (Flysjö et al. 2012). Since milk and meat production are interlinked and changes in the one production will affect environmental impacts in the other system expansion should be used instead of allocation. On the LCIA-level, environmental impact categories such as soil quality and functional biodiversity, which are important impact categories for the analysis of agricultural systems, are not considered when comparing organic and conventional agriculture products. We conclude based on present LCAs that an inter-comparison of products from organic and conventional agricultural products is difficult and improvements can be reached on the inventory and impact assessment level. With such adaptions LCA can become even more important to evaluate the environmental impacts of agricultural products.

References

Cederberg, C., Mattsson, B., 2000. Life cycle assessment of milk production - a comparison of conventional and organic farming. J. Clean. Prod. 8, 49-60.

Flysjö, Anna , Christel Cederberg, Maria Henriksson, and Stewart Ledgard. 2012. The interaction between milk and beef production and emissions from land use change e critical considerations in life cycle assessment and carbon footprint studies of milk. J. Clea. Prod. 28, 134-142.

Hörtenhuber, S., Lindenthal, T., Amon, B., Markut, T., Kirner, L., Zollitsch, W., 2010. Greenhouse gas emissions from selected Austrian dairy production systems - model calculations considering the effects of land use change. Renewable Agric. Food Syst. 25, 316-329.

Mondelaers, K., Aertsens, J., Van Huylenbroeck, G., 2009. A meta-analysis of the differences in environmental impacts between organic and conventional farming. Br. Food J. 111, 1098-1119.

Table 1. Parameters reviewed in the analysis of LCA studies.

Parameter
Attributional or consequential LCA
Goal of the study
Temporal and geographical system boundaries chosen
Functional unit(s) used
Assumptions made regarding agricultural production (including yields)
Inventory data basis used
Site-specific emission and characterisation factors used
Allocation rules applied
Impact categories assessed
LCIA-methods used
Sensitivity analyses to choices of methods conducted
Uncertainty analyses of results conducted

86. Environmental impacts and resource use in feed production for Atlantic salmon aquaculture

Erik Skontorp Hognes[1,*], Friederike Ziegler[2], Veronica Sund[2]

[1] SINTEF Fisheries and aquaculture, Norway, [2] SIK- the Swedish Institute for Food and Biotechnology, Sweden, [*] Corresponding author. E-mail: *erik.hognes@sintef.no*

As a part of the development of salmon feeds with reduced content of fish meal and fish oil the occupation of agricultural land; marine primary-production-required; GHG emissions and cumulative energy demand was assessed for a set of different feed diets for Atlantic salmon. The assessments were performed with LCA methodology and covered the salmon production system from growing and catch of feed ingredients till the salmon was ready for slaughter. The results were compared with Swedish chicken and pig production. Data on the feed compositions was delivered by the three of the world's largest salmon feed producers.
LCAs of Norwegian seafood have mainly focused on potential climate impact and cumulative energy demand (Winthur et al., 2009). This project expanded the scope of environmental impacts by addressing the reliance on agricultural land and marine resources (Pelletier and Tydemers, 2007), the latter by including the marine primary production required (PPR) to sustain the marine ingredients (Pauly and Christensen, 1995). The PPR was calculated by combining the trophic level of each fish species with catch location and average primary production per square meter ocean surface for that location. This method has important uncertainties from deciding the tropic level of the species and the average primary production per surface area for each fishing location.
A salmon that is fed the average Norwegian feed diet, in 2010, has a carbon footprint of 2.6 kg CO_2 equivalents; it occupies 3.3 m^2 agricultural land and requires 115 m^2 of sea primary-production area. Studying both potential climate impact and primary production required made it possible to study trade-offs between them, but also discover where there are no trade-offs. E.g. reducing the content of American marine ingredients would increase both the carbon footprint and the PPR, since the American species, used in these feeds, have a low trophic level and are sourced by very energy efficient fisheries.
Increasing or decreasing the use of marine ingredients may alter the carbon footprints by ± 7%. One reason for this is that the marine ingredients are replaced with soy protein concentrate that is attributed with a high carbon footprint since it contributes to deforestation. Deforestation has previously rarely been included in the GHG assessment of salmon feed production.
The comparison with pig and chicken concluded that salmon has the lowest climate impact and occupies the least agricultural land. Even an almost "vegetarian" salmon occupy less agricultural land than chicken. Pig had the highest carbon footprint and the highest occupation of agricultural land.
Important parts of the current and future feeds are derived from by-products from fisheries (pelagic and whitefish species) and from poultry by-products, The use of by-products highlighted the importance of the allocation strategy, here mass allocation was used and thus the by-products contributed significantly to the carbon footprint and this highlights the importance of evaluating how future standards for GHG assessments should treat allocation requirements. This LCA study was part of a bigger project by Nofima Marin on the resource utilisation and eco-efficiency of Norwegian salmon farming in 2010 (Ytrestøyl et al., 2011).

References

Pauly D, Christensen V. Primary production required to sustain global fisheries. Nature. 1995;374: 255-257. [With erratum from 376: 279]

Pelletier N, Tyedmers P. Feeding farmed salmon: Is organic better? Aquaculture. 2007;272(1-4):399-416.

Winther U, Ziegler F, Skontorp Hognes E, Emanuelsson A, Sund V, Ellingsen H. Project report: Carbon footprint and energy use of Norwegian seafood products www.sintef.no/Fiskeri-og-Havbruk-AS/Prosjekter/2011/Miljoregnskap-og-klimasporing-av-sjomat/. Trondheim, Norway: SINTEF Fisheries and aquaculture; 2009.

Ytrestøyl T, Aas TS, Berge GM, Hatlen B, Sørensen M, Ruyter B, et al. Resource utilisation and eco-efficiency of Norwegian salmon farming in 2010 Norway: Nofima; 2011.

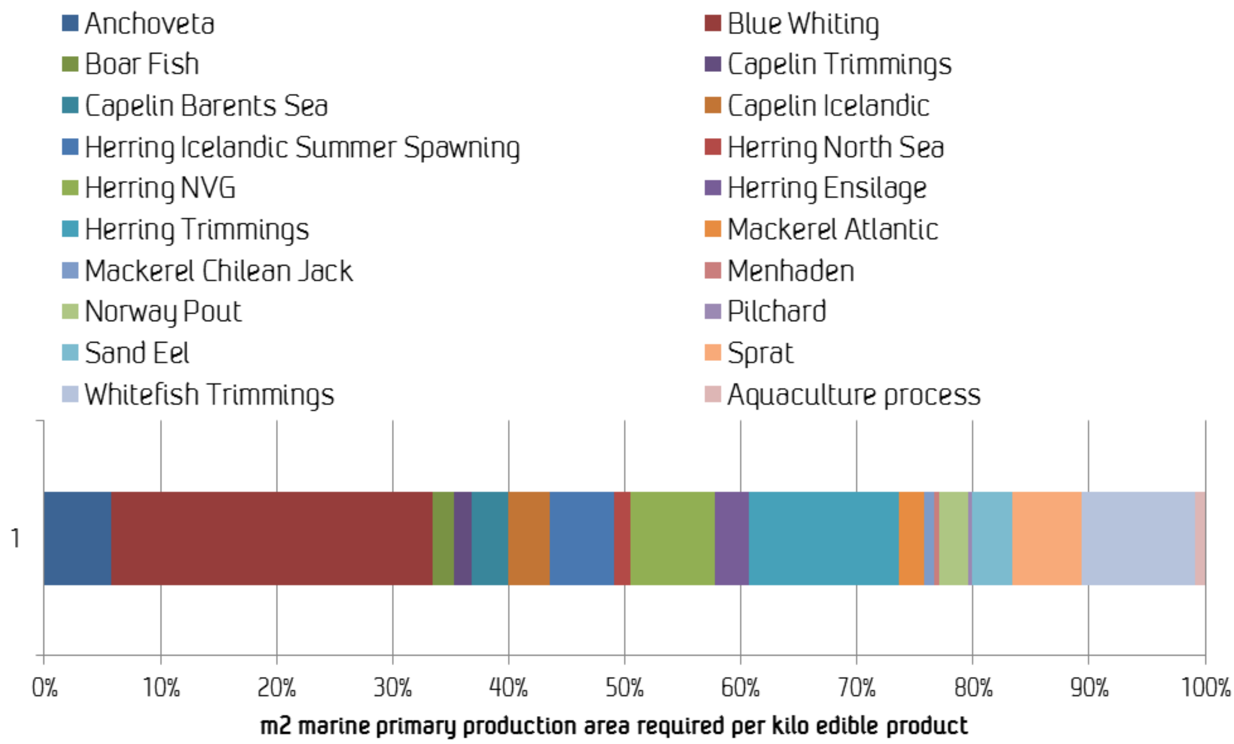


Figure 1. Contribution to marine primary-production-required per kg edible salmon from 2010 diet

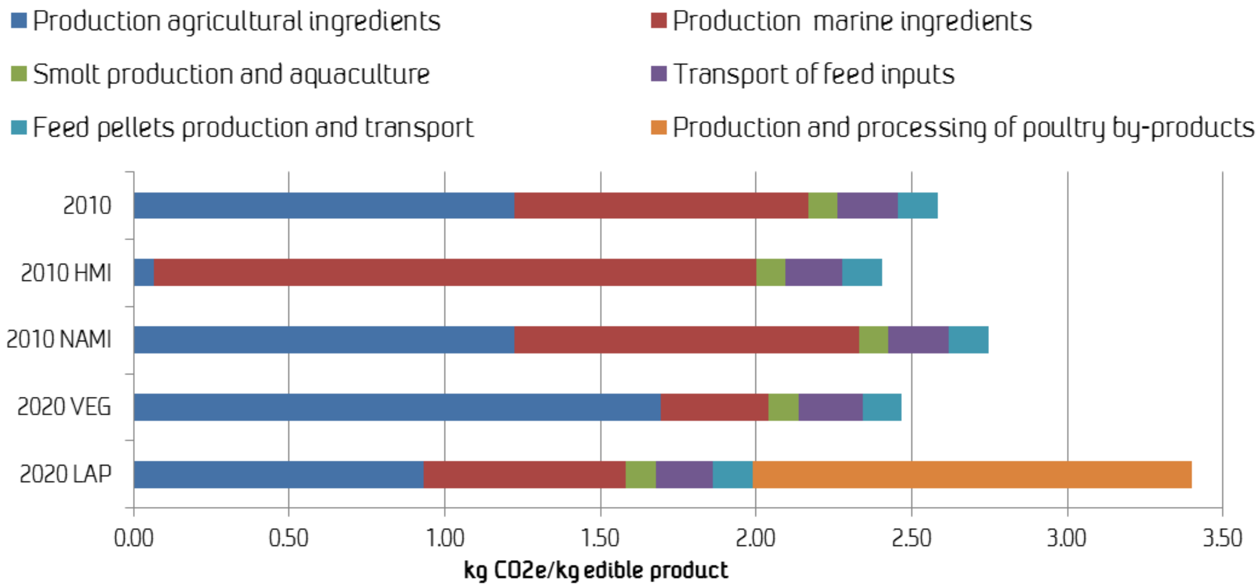


Figure 2. Total carbon footprint per kg edible product for each diet. 2010=Average Norwegian diet in 2010; HMI=High level of Marine Ingredients; NAMI=No American Marine Ingredients, VEG=Low content of marine ingredients; LAP=Use of poultry by-products.

87. EPD of extra-virgin olive oil: an Italian experience

Giulio Mario Cappelletti[1], Giuseppe Martino Nicoletti[1], Alessandro Leone[2], Giuseppe Lopriore[3], Carlo Russo[*]

[1]Dipartimento di Scienze Economico-Aziendali, Giuridiche, Merceologiche e Geografiche (SEAGMeG) – Università degli Studi di Foggia - Via Caggese, 1 – 71121 Foggia, Italy, [2]Dipartimento di Dipartimento di Scienze Agro-Ambientali, Chimica e Difesa Vegetale (DiSACD) – Università degli Studi di Foggia - Via Napoli, 25 – 71122 Foggia, Italy, [3]Dipartimento di Scienze delle Produzioni e dell'Innovazione nei Sistemi Agro-alimentari Mediterranei (PRIME) – Università degli Studi di Foggia - Via Napoli, 25 – 71122 Foggia, Italy, [*]Corresponding author. E-mail: *c.russo@unifg.it*

The Environmental Product Declaration (EPD) is a document based on the ISO 14025 standards; it contains information about environmental performance related to a product. The quantification of environmental performance is calculated using Life Cycle Assessment methodology, thus following the principle guides of ISO 14040 standards. Nowadays many EPDs have been issued by the International EPD System®, especially for the agro-food sector, due to the increased interest of companies and customers in this environmental communication tool. Recently, the EPD has also been made available for extra-virgin olive oil (PCR 2010).
To obtain the EPD, LCA methodology was applied to the life cycle of a pitted extra-virgin olive oil packed in a 1-litre glass bottle. The functional unit (FU) is 1 litre of extra-virgin olive oil, while the system boundaries include: a) the olive production (agricultural phase), b) the transportation of the olives from the olive grove to the oil mill, c) the process used for producing pitted extra-virgin olive oil (industrial phase), d) the packaging process involving putting the oil into 1-litre glass bottles, e) transportation to distribution, f) the "end of life" of the product and packaging disposal. An allocation procedure was only carried out for the phases of olive oil extraction and packaging (CORE); this distinguished between: olive oil, pomace and pit. Allocation was calculated considering two factors: mass and economic value of the co-product.
Data referring to the production of the raw material are all primary data, exception for the information about fuels, lubricants, and machines, for which the ecoinvent v.2 database was used. As for transportation, distances were estimated by referring to average values, while emissions were calculated by using the PE – International database (updated July 2010). Data relating to olive oil production are all primary data directly collected in the oil mill. As far as electricity production is concerned, the Italian power grid mix derives from the literature (AEEG and GSE, 2011). Data referring to packaging production comes from the Plastic Europe and PE-International database. As for the "use phase", the percentages of recovery and landfill disposal of glass, as well as the data regarding cardboard and plastic, derive from reports published by the National Packaging Consortium. Collected data were elaborated using GaBi 4.4 software, with reference to the functional unit represented by 1 litre of pitted extra-virgin olive oil (weighing 0.92 kg) packed in a 1-L glass bottle. To obtaining one litre of extra-virgin olive oil, 5.29 kg of olives, harvested on a surface of 18.9 m^2, are needed. In accordance with the indications of PCR clause 10, an environmental impact assessment was carried out distinguishing the use of resources, potential environmental impact and other indicators. The agricultural phase would benefit from more non-renewable material and energy resources, while the extraction process requires more renewable material resources, water and electricity. Finally, the packaging phase would need more renewable energy resources. The feedstock energy is only linked to the PVC production: it is almost 0.003 kg of crude oil and 0.002 kg of natural gas per functional unit.
As for the assessment of the potential environmental impacts, the Global Warming Potential (GWP) is more affected by the agricultural phase, followed by the packaging phase. This impact derives from the fuel consumption (for the soil cultivation) and from the production process of the primary packaging (glass bottle and PVC capsule). The value of CO_2 equiv. per FU is over 1 kg, and over 50% of this derives from the olive-production while 30% derives from the packaging: The same trend is detected for the categories of Acidification, Photochemical Ozone Creation and Abiotic Depletion, while, as far as the Ozone Layer Depletion is concerned, the packaging phase is the most pollutant. As for Eutrophication, the most pollutant phase is the olive oil extraction, due, in particular, to the higher pollutant charge of the wastewater (washing water, vegetation water and water added in the extraction process). Analyzing the impact calculated as kg of phosphate equiv., almost 90% of the total value per FU (about 16 g) is linked to the extraction phase (wastewater).
As for the impact categories referring to toxicity, except for Terrestial Ecotoxicity, the packaging phase is again the most pollutant. Here too, the explanation is linked to the production of the glass container, which accounts for over 80% of the total impact.

The analysis of the environmental performance includes the assessment of the impact categories of Land Use by using the Eco-indicator-99 (Land Use and Land Conversion) and IMPACT 2002+ (Land Occupation) evaluation methods. For all these categories, over 90% of the total impact derives from the agricultural phase. Then other indicators are analysed; the “Material subject for recycling or other use” category includes pruning residues (agricultural phase), leaves and little branches (extraction phase) and recycled packaging (glass, plastic and cardboard). The production of “Hazardous and environmentally active waste” is nil, as is the production of toxic emissions. With regard to other types of waste, the wastewater deriving from the extraction phase and the waste disposed in landfill (not recycled) must be indicated. Finally, pruning residues and by-products of the olive oil extraction (leaves and little branches, pomace and pit) are included among the waste that could be recovered as renewable energy.

The LCA study enables us to detect some improvements which could be adopted for the agricultural phase (as reducing to the minimum the employment of non-renewable energy resources and water, preserve the fertility of the soil or recovering the energy value of pruning residues, by the controlled burning carried out in a boiler for the thermal energy production) and for the processes of olive oil extraction and packaging, (as improving the management of waste and wastewater during the extraction phase or studying other types of container). While in order to reduce the environmental impact of the end of life, efforts must be made to improve the communication with consumers, by stating more information on the label, with the aim to guide consumers towards sustainable methods of packaging disposal.

The critical analysis of PCR developed by the International EDP® System aims to elaborate some consideration about the possibility of Italian firms adopting these rules or submitting to some modifications after the expiry date (31 Dec 2013), on the basis of elements which are more representative of the Italian olive oil industries. From this critical analysis we can assert that PCR could be adopted by the Italian olive oil firms without any particular problems. As for the environmental performance developed in paragraph 10 of PCR, some suggestions might be made in order to explain the impact of toxicity and land use better. The analysis of toxicity would be more complete if the two toxicity categories developed by Ralph K. Rosenbaum et al. in the UseTox method are considered (Ralph et al., 2008): Human Toxicity (expressed as no. of cases x kg) and Freshwater Ecotoxicity (expressed as kg DCB-equiv.). These suggestions could be relevant in consideration of the need to assess the toxicity for the processes in which there is an agricultural phase involving a high impact in terms of toxicity, deriving from the use of pesticides.

In the same way, as for the land use categories, indicators could be better shown, as Baitz (2002) suggests, by distinguishing: Erosion Resistance; Mechanical Filtration; Physicochemical Filtration; Groundwater Replenishment; Biotic Production. Finally, some considerations regarding the CO_2 balance; to achieve a correct balance, the CO_2 quantity immobilised into soil from olive trees and herbaceous biomass (winter and spring weeds, grass cover etc) should be indicated. Frequently, this information is not indicated, or only partially, due to the lack of experimental data able to evaluate it in an appropriate way, even if some studies (Sofo et al., 2005) determined for Mediterranean areas the year-1 of CO_2 fixed in an olive grove.

References (selected)

Alberti F., Neri P., Cecchini F. 2005. LCA applied to the Olive Oil Chain – standard study (in Italian). Thesis of Master in Environmental Development and Management of the Agro-Environmental Systems. University of Bologna, ENEA-PROT-P135-100.

Baitz, M. 2002: Die Bedeutung der funktionsbasierten Charakterisierung von Flächen-Inanspruchnahmen in industriellen Prozesskettenanalysen. Ein Beitrag zur ganzheitlichen Bilanzierung. Dissertation. Shaker (Berichte aus der Umwelttechnik), Aachen

Carli G. 2002. The Environmental Impact of the Olive Oil Production: application of the Life Cycle Assessment (LCA) methodology (in Italian).

International EPD Consortium (IEC). 2009. General Programme Instructions for Environmental Product Declaration EPD, Version 1.0, of 29/02/2009.

PCR 2010:07; CPC 21537 – PCR for virgin olive oil and its fractions; ver. 1.0, 2010-04-27.

88. Communication strategies for product sustainability messaging aimed at end consumers

Thomas Redd[1,*], Doug Harrison[2], Laura Riley[2], Greg Thoma[3]

[1] The Sustainability Consortium, jointly administered by Arizona State University and University of Arkansas, USA, [2] Harrison Group Inc., [3] University of Arkansas, USA, [*] Corresponding author. E-mail: *tredd@asu.edu*

The proliferation of communication guidelines for environmental claims and conflicting eco-labelling schemes has caused industry organisations to acknowledge the need to provide consumers with a more harmonised set of sustainability labels and claims in the marketplace (Ryan Partnership Chicago/Mambo Sprouts Marketing, 2011). Government and industry organisations also widely recognise the need for more scientific research to help us understand how to effectively communicate sustainability information to end-consumers (Ernst & Young, 2009; Galatola, 2011; Ryan Partnership Chicago/Mambo Sprouts Marketing, 2011). Initial reports conclude that consumers prefer to be presented with grades or scores of a product's sustainability performance, rather than the absolute values, ratios, and physical units found in most LCA studies (Ernst & Young, 2009). Consumers' intent to use product level sustainability disclosures is also highly dependent on the message being presented at the point of sale and verified by an independent organisation (Ernst & Young, 2009; Ryan Partnership Chicago/Mambo Sprouts Marketing, 2011).
Building on these industry reports, The Sustainability Consortium launched an applied research project to answer some of the key questions posed by leading retailers and brand manufactures in the Consumer Goods industry. This particular empirical study seeks to understand, from a consumer's perspective, how the design and format of LCA-generated scores presented with products at the point of sale affects the message's understand-ability, believability, comparability, and usability. Researchers also attempt to measure consumer preferences for 13 sustainability attributes (Fig. 1) and 22 messaging formats (Fig. 2) on product level disclosures. To examine variance in consumer preferences across product categories, the attribute and format variables are tested in combination with five different product category variables such as cereal, paper towels, and laundry detergent. Variance in consumer preferences across geo-political scales is also examined by sampling consumers from four different sovereign nations, including France.
Qualitative and quantitative results from ten focus group sessions and three online surveys suggest there are ample opportunities for establishing scientifically-grounded best practices for communicating LCA-generated information. From a global perspective, the data indicates 1) the presence of a scoring scale and attributes with a positive tone are critical for establishing understand-ability among consumers 2) consumers are most likely to prefer a messaging format that's grounded in the percentages scoring technique 3) scores accompanied by stoplight colour coding that identifies three thresholds of relative industry performance substantially enhance usability. Further research is ongoing and scheduled for completion in May, 2012.

References

Ernst & Young, 2009. Pistes pour un etiquetage environnemental lisible et efficace – Resultat d'enquete consommatuer.
Galatola, M., 2011. The role of the ILCD Handbook as methodological basis for robust life cycle based environmental policies, SETAC 2011. Milan
Ryan Partnership Chicago/Mambo Sprouts Marketing, 2011. One Green Score for One Earth Consumer Sustainability Survey.

Figure 1. A sample of the sustainability attributes

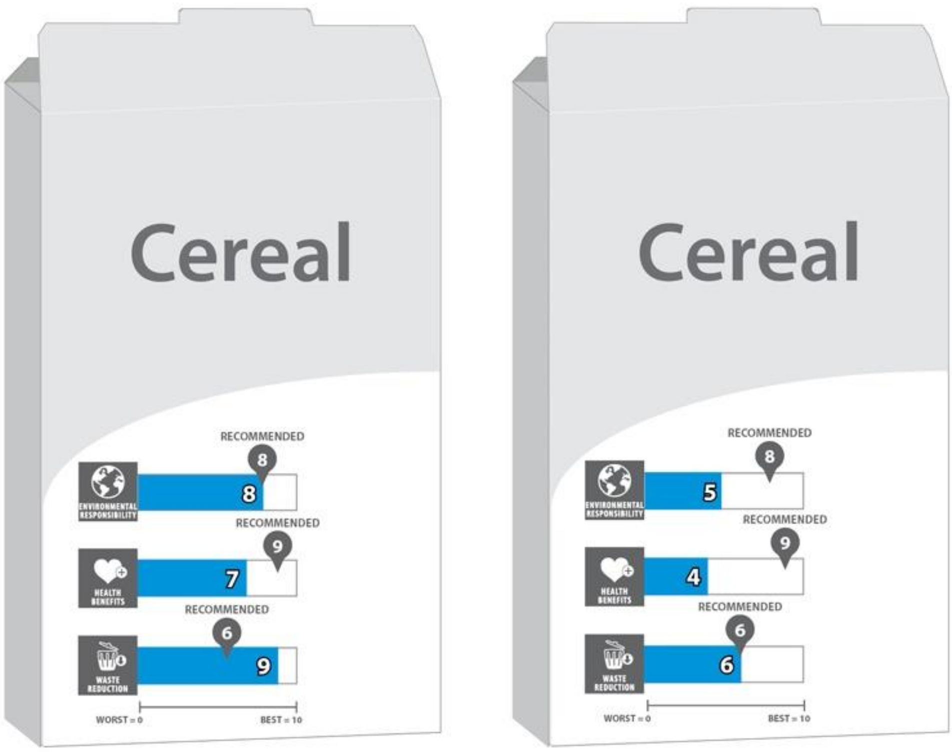


Figure 2. An example of one of the messaging formats

89. The good egg in the basket: improvements in egg production within the Pro Planet eco-labelling scheme

Kewin Comploi*, Martin Wildenberg, Tanja Altaparmakova, Dominik Frieling, Lydia Matiasch, Helmut Burtscher

Umweltforschungsinstitut GLOBAL 2000 / Friends of the Earth Austria, Neustiftgasse 36, 1070 Vienna, Austria, * Corresponding author. E-mail: *kewin.comploi@global2000.at*

Eco-labelling has often been proposed as a valuable instrument for the implementation of higher environmental and social standards. For instance, Gereffi et al. (in Honey 2002, p.51) suggest that "*While certification will never replace the state, it is quickly becoming a powerful tool for promoting worker [...] rights and protecting the environment in an era of free trade*".

The poster shows actual quantifiable improvements in egg production using the GLOBAL 2000 adaptive sustainability assessment approach. The results are based on five indicators (carbon-footprint, biotic and a-biotic material input, water input and area used) calculated for the most relevant stages of the product life cycle. For example, it can be illustrated that the mere replacement of Brazilian soy with soy produced in the EU more than halves greenhouse gas emissions of the whole egg production process due to different land use patterns (cf. Hörtenhuber and Zollitsch, 2010; Nemecek and Kägl, 2007). This coincides with favourable conditions for rising European soy production: first, the latest price increases on the market; second, the spreading of the corn woodworm in some European countries and the need for crop rotation to avoid extensive use of nicotiniod-based insecticides.

Both the method used and the results refer to the eco-label Pro Planet (www.pro-planet.at) as part of the REWE International AG, GLOBAL 2000 and Caritas Sustainability Program for Fruits, Vegetables and Eggs.

References

Honey, M., 2002. Ecotourism and certification: setting standards in practice; Island Press

Hörtenhuber, S. and Zollitsch, W., 2010. Greenhouse gas emissions of regionally produced alternative feedstuffs rich in protein for Austrian dairy production. In: Building sustainable rural futures, IFSA, Vienna, pp. 1349-1356.

Nemecek, T. and Kägl, T., 2007. Life Cycle Inventories of Swiss and European Agricultural Production Systems. Final report evoinvent V2.0 No 15a. Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for Sustainable Life Cycle Inventories, Zurich ans Dübendorf, Switzerland.

90. Carbon labels worldwide- a review of approaches and indices

Michael M. Blanke*, Florian Schaefer

INRES – University of Bonn, Auf dem Hüegel 6, D-53121 Bonn, Germany, * Corresponding author. E-mail: *mmblanke@uni-bonn.de*

This contribution does not provide a simple summary of carbon labels worldwide, but a critical review of existing international labelling policies and their background and ways of communicating their LCA foods results to the consumer. Advantages and disadvantages from both the manufacturer and consumer angle are presented.

Carbon Footprint is a tool to allocate products or services a numeric value for their specific impact on climate change based on Life Cycle Assessment (LCA). Accounting greenhouse gas emissions over the whole life cycle of products or services allows allocating each product a unique Carbon Footprint. This is a first steps to communicate the sustainability performance of products or services to the consumer.

A thorough market analysis conducted in 2011 resulted in the categorisation of seven schemes, into which the carbon labelling and consumer communication of carbon footprint LCA Foods results can be categorised:

(1) Presentation or label of a single static numeric value on the product in the shop; examples are activities by the Carbon Trust, UK and the single carbon footprint value as operated by Tesco's supermarkets in the UK (but not in Tesco's stores abroad). However, this labelling policy is restricted to ca. 20 food and non-food products (orange juice, skimmed milk, Walker's crisps, sliced bread, and light bulbs as the non-food product) out of their ca. 70,000 in-store products viz 0.03% of products on offer. A similar policy is operated by KEITI with their "CO_2 low label" or "COOL" in South Korea. The consumer, however, may find it difficult to judge and memorise numeric values, particularly if expressed on different units (e.g. packet size, litre, 100 g). Similar schemes operate in Thailand and Japan.

(2) Carbon reduction labels (Climatop) offer an answer or alternative to the static CO_2 value. They indicate the activity in the field of carbon footprint, without giving a single static CO2 value. Climatop in Switzerland is presented as an example and labels for green and white asparagus are discussed. Climatop labels have a life-span of two years, which encourages the manufacturer to further improve the carbon footprint.

(3) The colour schemes offer another alternative, where consumers do not need to memorise or interpret a carbon label. The French Casino supermarkets react to the French carbon footprint laws, "Grenelle 2" and Grenelle 2" by operating a green- yellow colour code called 'Indice Carbone', where the product is visually ranked within the best and worst product in the market.

(4) Air freight labels on the produce, operated by Tesco and Marks and Spencer in the UK (and KaDeWe in Germany and CoOp , Switzerland) indicate the food items with the largest carbon footprint, are easy to handle by the supermarkets and understand by the consumer; they also give regional produce a chance, if within that season.

(5) Carbon Zero initiatives like the CarboNZero in New Zealand and CarbonFree in the USA may cause ambiguity and the consumer think the production of this particular food is carbon neutral. These schemes, however, are based on a thorough carbon footprint or complete LCA and subsequent purchase of golden carbon certificates.

(6) QR codes on the products are a fairly new alternative since 2011, where the new generation of mobile phones access the internet in the shop and relevant information is available e.g. for a carbon footprint, which varies seasonally or the LCA is more complicated than a single value.

(7) Sustainability reports are the last option, favoured e.g. by companies in Germany, to report their carbon footprint both in the printed and online versions of their sustainability report, which have to follow the standards and guidelines for sustainability reports, e.g. by dnv.

Table 1. Overview of carbon footprint /water footprint / LCA Foods results to the consumer.

Types of communicating carbon footprint /LCA food results to the consumer	Examples of schemes or organisations	Examples in the food sector
(1) carbon value on produce[a]	Tesco “COOL” by KEITI, S. Korea	Milk, orange juice, bread, sugar, potato crisps
(2) Carbon reduction label	Climatop, Switzerland	sugar
(3) Colour schemes (green to yellow)	Casino France	All products on offer.
(4) Air freight labels on produce	Marks & Spencer (UK), Tesco (UK), KadeWe (D), Coop (CH)	Asparagus, pittaya, pineapple, lemon grass
(5) Carbon Zero or Carbon-free initiatives	CarboNZero, New Zealand CarbonFree, USA	
(6) QR codes on produce or package	Announced for the future	
(7) Sustainability reports	e.g. Unilever, Barilla etc.	All products

91. Food waste amounts and avoidability in Switzerland

Claudio Beretta[1,*], Franziska Stoessel[1], Stefanie Hellweg[1], Urs Baier[2]

[1] ETH Zurich, Institute of Environmental Engineering, Schafmattstrasse 6, CH-8093 Zurich, Switzerland [2] ZHAW Wädenswil, Life Sciences und Facility Management, Einsiedlerstrasse 29, CH-8820 Wädenswil, Switzerland, [*] Corresponding author. E-mail: *cberetta@ethz.ch*

A key element in making our food system more efficient and sustainable is the reduction of food losses across the entire food chain (Quested and Johnson, 2009). Nevertheless, in many LCA analyses food losses are missing. The first step in developing efficient measures is the analysis of the present situation and the identification of hotspots. This paper aims to quantify the food losses in Switzerland at the various stages of the food chain (agricultural production, postharvest handling and trade, processing, food service industry, retail, and households), to identify hotspots and analyse the reasons for the losses. Twenty-two food categories are modelled separately in a mass and energy flow analysis, based on data from 31 companies within the Swiss food chain, as well as from public institutions, associations, and from literature.

The energy balance shows that 48% of the total produced calories (edible crop yields at harvest time and animal products, including slaughter waste) is lost across the whole food chain. Half of these losses would be avoidable. The allocation of the avoidable food losses to the various stages of the food chain identifies agricultural production, the processing industry and households as playing a key role. Households waste 45% of the edible calories lost over the entire food chain. However, there are various uncertainties in quantifying food losses. A major uncertainty lies in the quantification of losses in agricultural production, which are mainly caused by quality sorting and omission of harvest due to high fluctuations in demand and inappropriate organisation. Further research to quantify losses and to develop strategies for optimisation is especially important in this field.

A broader scenario focuses on the potential increase in food availability by replacing animal products relying on feed grown on arable land by vegetarian products. In Switzerland, livestock relies up to one third on feed that is grown on arable land. If one third of the animal products were substituted by vegetarian products, 45% more calories would be available for consumption. If, additionally, all the edible parts of the food produced for Swiss consumption were eaten by humans, 50% more calories would be available (Fig. 1).

References

Quested, T., Johnson, H., 2009. Household Food and Drink Waste in the UK: A report containing quantification of the amount and types of household food and drink waste in the UK. Report prepared by WRAP (Waste and Resources Action Programme). Banbury.

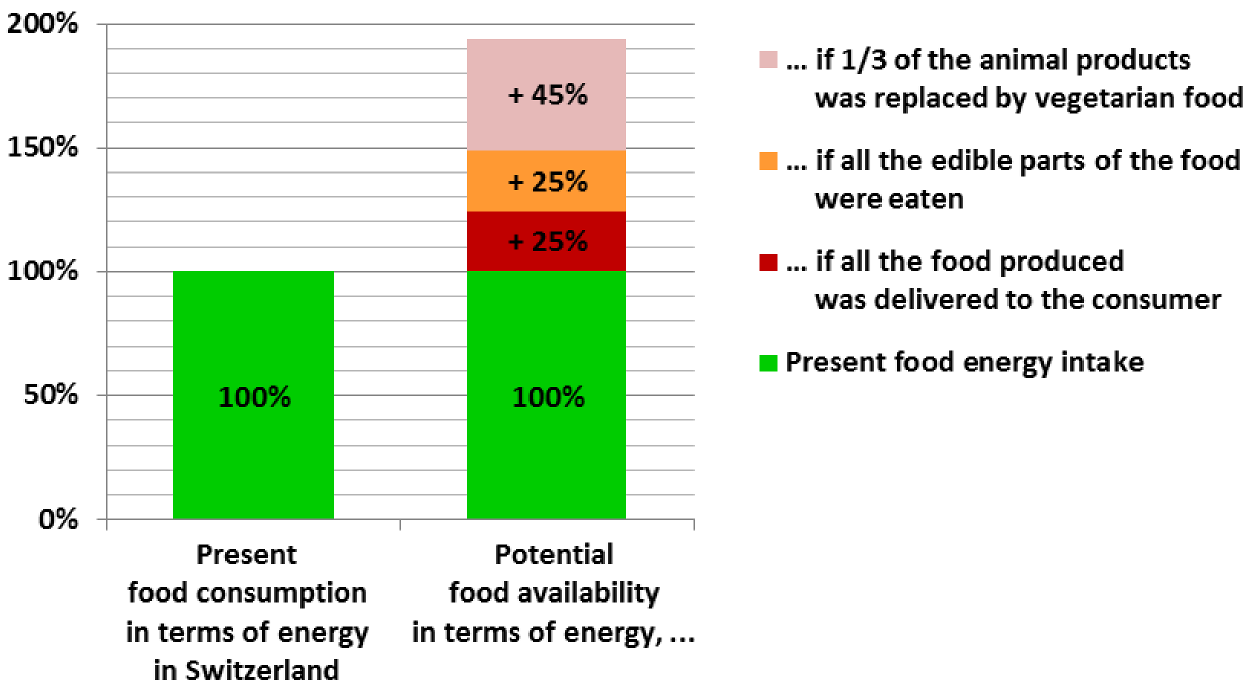


Figure 1. Total potential of avoiding food losses in Switzerland: In a theoretical scenario of perfect distribution, optimal methods of cooking and preparation, and the replacement of one third of the actually consumed animal products by vegetarian food, 195% of the presently consumed food calories would become available for consumption.

92. Life cycle assessment of alginate production

Juliette Langlois[1,2,*], Jean-François Sassi[3], Jean-Philippe Delegenes[2], Jean-Philippe Steyer[2], Arnaud Helias[1,2]

[1] Montpellier SupAgro, F-34060 Montpellier cedex 2, France, [2] INRA, UR50 LBE, F- 11500 Narbonne, France, [3] CEVA, F-22610 Pleubian, France, [*] Corresponding author. E-mail: *langlois@supagro.inra.fr*

Algal polysaccharides, also called phycocolloids, are the main commercial seaweed extracts: their production for sale reached 86,100 t over the world in 2009, considering agar, alginates and carrageenans productions (Bixler and Porse 2010). They are mainly used in the agri-food sector, as texturing agents, stabilisers, gel formers of film forming agents. Many other industrial applications exist, as microbiological and electrophoresis media for agar, use in textile printing and paper coating for alginates and in toothpaste, cosmetics and pharmaceutics for carrageenans (Bixler and Porse 2010). To our knowledge, their environmental assessment using Life Cycle Assessment (LCA) has not been performed yet, despite this large use at industrial scale.

We performed the LCA of phycocolloid production including seaweed cultivation. The Recipe method (Goedkoop et al. 2009) was used, with a hierarchist perspective using the ecoinvent v2.2 database and the SimaPro 7.3 software to carry out the impact assessment. The functional unit was to produce 1 kg of hydrocolloid. The study is a prospective for European countries, considering pilot and semi-industrial data from the North-Eastern Atlantic zone (data from Aleor, French seaweed producer). Bibliographic data were also used for electricity consumption (Mafart 1997). We modeled the production of food-grade phycocolloid, because this use is the main market for phycocolloids (Bixler and Porse 2010). Brown seaweeds are considered due to their high growth rate potential. In the present study, seaweed was *Saccharina latissima*, commonly found in this area, and reaching high alginate content, with high levels of guluronic to mannuronic acid ratio. It was cultivated on long-lines in coastal waters, after plantlet production in nursery, as described in Langlois et al. (2012). Seaweeds were treated straight after harvest, with common techniques of sodium alginate production (McHugh 2003). Seaweed were first washed, crushed and treated with alcohol. After acid lixiviation and dewatering, an alkaline extraction was carried out to solubilise alginates, followed by dewatering using filter press. An acid precipitation with blending was then operated, followed by a last dewatering and the addition of sodium carbonate before drying.

Contribution analysis results highlight the importance of the sodium alginate production itself (see Fig. 1). On average on every impact categories, the seaweed production accounts for less than 1% of the total impacts, even allowing bioremediation to marine eutrophication thanks to the nutrient uptake offshore. Electricity is the main contributor to environmental impacts for 12 over 18 impact categories analysed, reaching 39% of the total impacts on average. It is followed by the use of chemical (mainly because of hydrochloric acid), accounting for 26% of the total impacts on average. Heat and cooling requirements, wastewater and waste treatments, and the use of freshwater and mineral filter aid have only secondary impacts compared to them. This work underlines the key elements to improve for an ecodesign of phycocolloids production.

References

Bixler, H.J., Porse, H., 2010. A Decade of Change in the Seaweed Hydrocolloids Industry. J Appl Phycol 23 (3), 321-335.

Goedkoop, M., Heijungs, R., Huijbregts, M. et al., 2009. ReCiPe 2008, a Life Cycle Impact Assessment Method. Ministerie VROM, Den Haag, pp. 125

Langlois, J., Sassi, J.-F., Jard, G., et al., 2012. Life Cycle Assessment of Biomethane from Offhsore-cultivated Seaweed. Accepted in Biofuels Bioprod Bioref.

Mafart, P., 1997. Génie Industriel Alimentaire, Tome 1: Les Procédés Physiques De Conservation, vol.1. Technique Et Documentation. France: Lavoisier.

McHugh, D., 2003. A Guide to the Seaweed Industry. FAO Fisheries Technical Paper, 441. Rome. pp.118. Available online: http://www.fao.org/docrep/006/y4765e/y4765e00.htm

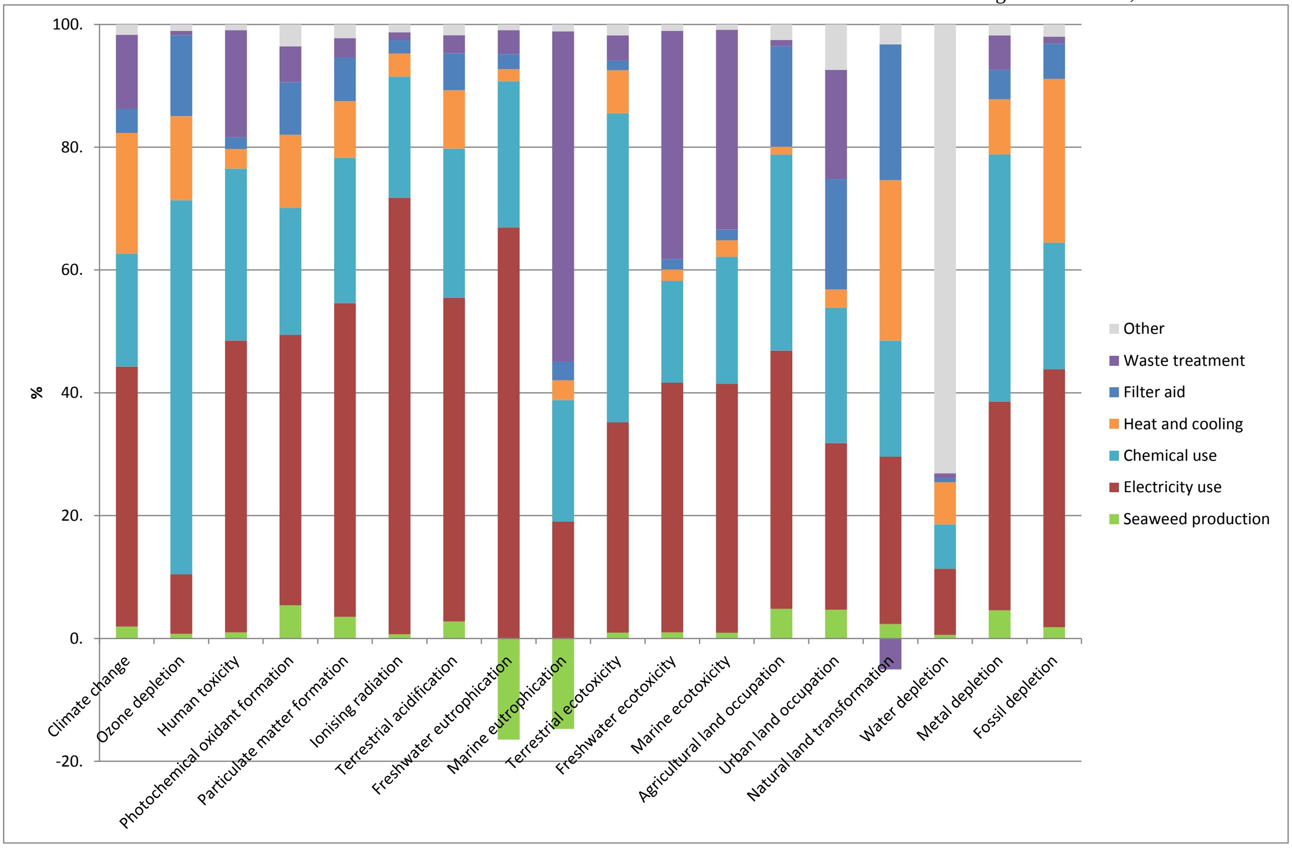


Figure 1. Contribution analysis for environmental impacts of sodium alginate production from offshore cultivated seaweed in European countries

93. Salinalgue project: designing a sustainable production system of biofuel and by-products from microalgae

Mélissa Cornélus[1,*], Pierre Collet[1], Stéphane LePochat[3], Roberto Bellino[3], Jean-Philippe Steyer[1], Arnaud Hélias[1,2]

[1] INRA, UR50, Laboratoire de Biotechnologie et de l'Environnement, Avenue des Etangs, 11100 Narbonne, France, [2] Montpellier SupAgro, 2 Place Viala, 34060 Montpellier Cedex 1, France, [3]EVEA, 56 Bd de la Fraternité, 44100 Nantes, France, [*] Corresponding author. E-mail: *melissa.cornelus@supagro.inra.fr*

Future developments for the microalgae sector are looking promising with high production yields and no direct competition with arable lands (Chisti, 2007). However, there remain some major bottlenecks related to the process chain where energy and water issues challenge the industrialisation of microalgae-based processes (Lardon et al., 2009).

In addressing these challenges, Life Cycle Assessment (LCA) has become an essential eco-design tool in the R&D step of algorefinery projects. As a consortium of 13 institutional and private partners, the Salinalgue project deals with the current environmental challenges underlined in several LCA studies on algal systems. This system features a culture set in an old farming salty land in the aim of reconversion with an extremophile native species. This ensures there will be no contamination and no competition against arable lands. Moreover thanks to the proximity with an industrial area, microalgae may be supplied with industrial CO_2 from exhaust gas. LCA aims at paving the way towards building more sustainable systems. Figure 1 gives an overview of the process chain: from the culture of biomass, its transformation by downstream processes to the use of oilcake and its conversion into energy and digestates which can later be recycled as inputs of the system. Below are the 4 key points addressed by the LCA study on the Salinalgue project:

(1) As there are no industrial facilities yet (even at pilot-scale) (Brentner et al., 2011), most of data is scaled up according to partners' knowledge and literature review. LCA helps to foresee the environmental impacts of emerging technologies. Several process technologies are considered at each step: such as green extraction by new technologies.

(2) Taking up the biorefinery concept, LCA helps partners by pointing out the hotspots, comparing options and informing eco-choices. This approach is reiterated at each stage of the system. For example, several CO_2 supply chains for the algae culture have been compared.

(3) The whole life cycle is analysed: from the production of algal nutrition to the recycling of nutrients contained in the oilcake (with the liquid fraction being recirculated towards cultivation ponds) (Collet et al., 2011). Thereby, LCA provides a holistic point of view. This is useful in such a collaborative project where each partner is mostly focused on its respective operation stage.

(4) Salinalgue is a multi-output system. In such a complex system, it is critical to deal with all the by-products in the LCA study, which raises some methodological issues such as allocation or substitution.

Salinalgue is a two-stage project, where an industrial scaling-up of the system will follow the experimental R&D stage. A step by step approach where the most preferable technology is selected will provide a roadmap towards the most sustainable process chain to yield biofuel and bioproducts from microalgae.

References

Brentner, L. B., Eckelman, M. J. et Zimmerman, J. B., 2011. Combinatorial Life Cycle Assessment to Inform Process Design of Industrial Production of Algal Biodiesel. Environ. Sci. Technol. 45 (16) pp.7060-67.

Chisti, Y., 2007. Biodiesel from microalgae. Biotechnology Advances 25 (3) pp.294-306.

Collet, P., Hélias, A., Lardon, L., Ras, M., Goy, R.-A. et Steyer, J.-P., 2011. Life-cycle assessment of microalgae culture coupled to biogas production. Bioresource Technology 102 (1) pp.207-14.

Lardon, L., Hélias, A., Sialve, B., Steyer, J.-P. et Bernard, O., 2009. Life-Cycle Assessment of Biodiesel Production from Microalgae. Environ. Sci. Technol. 43 (17) pp.6475-81.

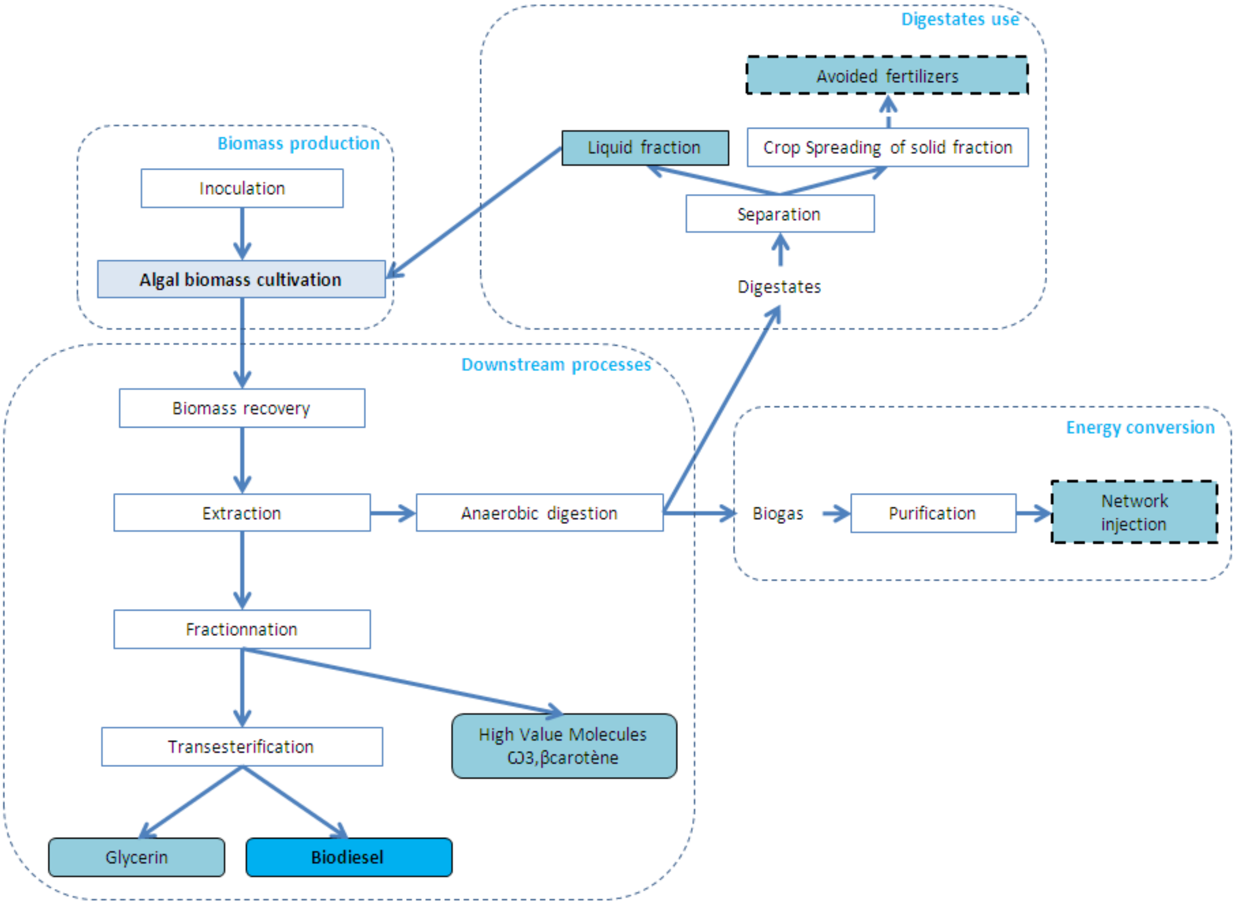


Figure 1. Process chain of the Salinalgue project overview

94. Using life cycle analysis to compare the environmental performance of organic and conventional apple orchards

Aude Alaphilippe[1,*], Sylvaine Simon[1], Laurent Brun[1], Frank Hayer[2], Gérard Gaillard[2]

[1] INRA, UERI Domaine de Gotheron, 26320 Saint Marcel les Valence, France, [2] Agroscope Reckenholz Tanikon ART, CH-8046 Zurich, Switzerland, [*] Corresponding author. E-mail: *aude.alaphilippe@avignon.inra.fr*

Although the conventional farming system for apple production remains the common practice worldwide, the organic farming system is developing. Although organic farming is generally claimed to be environmentally friendly, very few global assessments of the environmental impacts of organic orchard systems are presently available. In this work, we analyse with life cycle analysis (LCA) weak and strong points of the environmental performance of two organic and one conventional apple orchards, using a pluri-annual dataset from experimental orchard systems located in the middle Rhone valley in France. The analysis was performed with the SALCA method (SALCA-Crop, V3.1 adapted for pome fruit) and included relevant impact categories using characterisation models mainly from the methods EDIP97 and CML01.
Seven impact categories including ecotoxicity and human toxicity, as well as energy consumption and other environmental impact categories were calculated and are here presented. From this first insight, the organic systems were globally less impacting than the conventional system when considering the functional unit (FU) per hectare (ha*year), but among the highest impact –except for toxicity- when considering the (kg*year) FU because of low yields. High-yield conventional systems globally presented the opposite trend. The basic substitution of conventional by organic inputs or mechanical work was not sufficient to radically improve the overall environmental performance of the orchard system. This work also highlighted the importance of the cultivar in the orchard design towards more environmentally friendly apple production systems; a disease-susceptible cultivar such as Golden Delicious was more impacting than a low-susceptibility cultivar such as Melrose under the same cropping system. Only one-year results are here presented, to be further validated after several years of full production.
This study is the first apple LCA based on a multi-year system experiment, which provided all information requested to compute a LCA in a liable and high quality dataset, as it was collected on purpose in orchards under the same influence of the site or field context and management. The interest of such system experiments to create references for agricultural productions newly assessed with LCA methodology is discussed. Moreover the potential contribution of LCA to the design of innovative and less input dependant production systems is analysed: the calculation of the overall environmental impact potential (no focus on one aspect) of different farming systems, followed by an analysis of their weak and strong points, permits to propose some improvement of the farming systems.

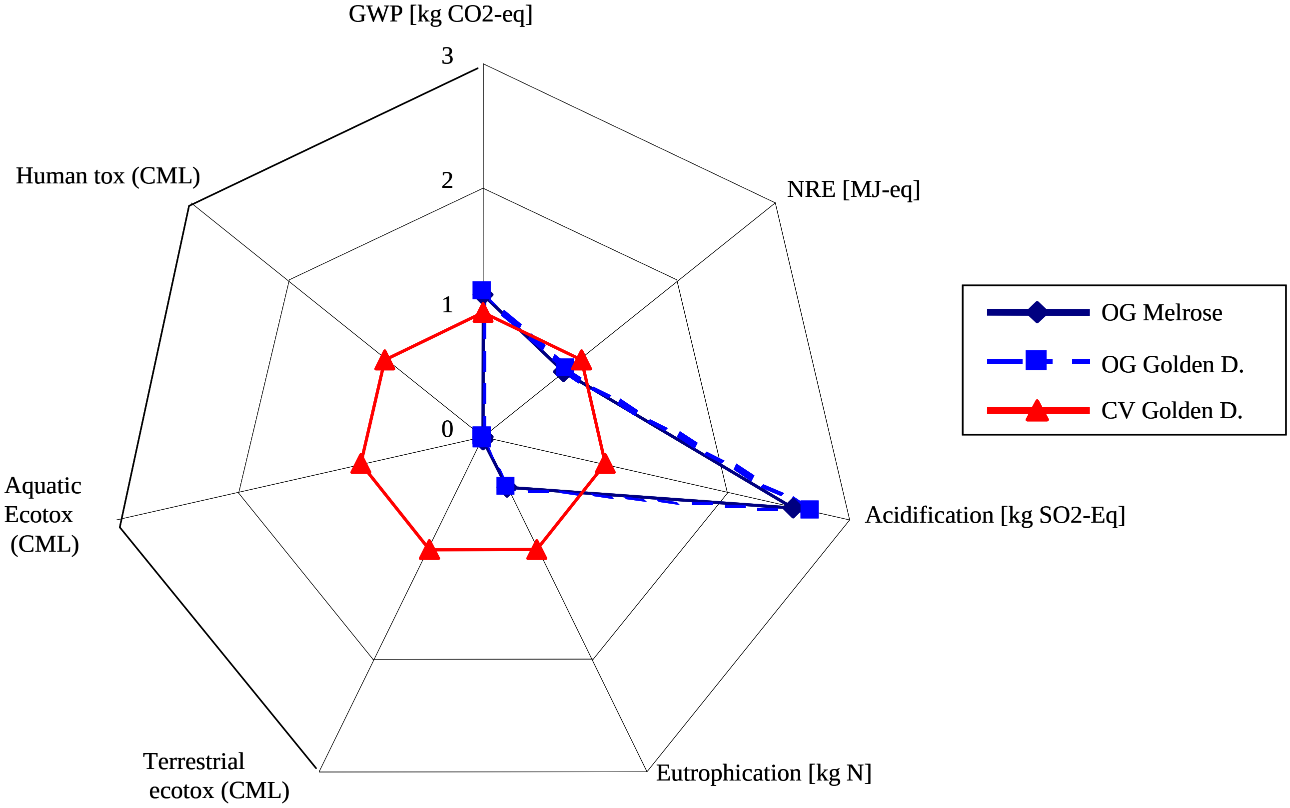


Figure 1. Seven impact categories (one-year data) of the two organic farming systems (OG Golden Delicious and OG Melrose) relatively to the conventional (CV) Golden Delicious farming system which was set up to the value 1 for each calculated impact category (functional unit: per ha*year).

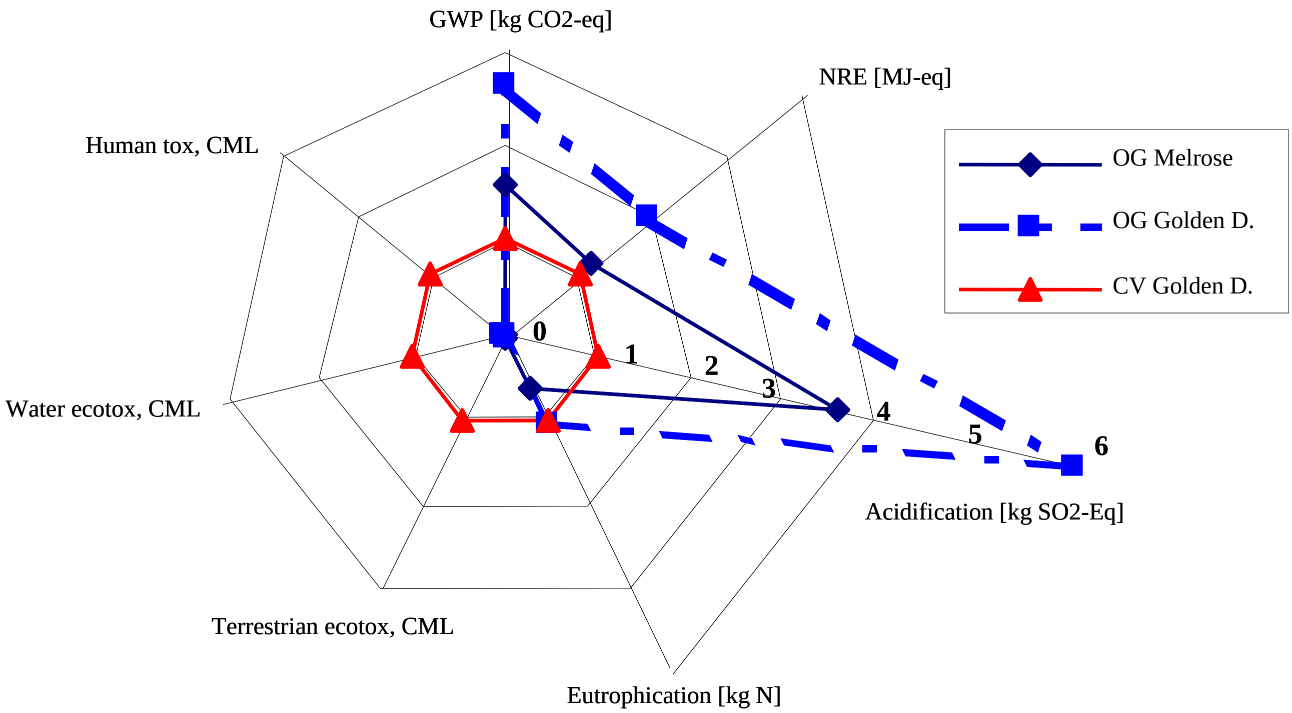


Figure 2. Seven impact categories (one-year data) of the two organic farming systems (OG Golden Delicious and OG Melrose) relatively to the conventional (CV) Golden Delicious farming system which was set up to the value 1 for each calculated impact category (functional unit: per kg*year).

95. Assessing environmental sustainability of apple ancient varieties in Northern Italy

Alessandro K. Cerutti[1,*], Daniela Galizia[1], Sander Bruun[2], Dario Donno[1], Gabriele L. Beccaro[1], Giancarlo Bounous[1]

[1] Department of Arboriculture, University of Torino, Via Leonardo da Vinci, 44 – 10095 Grugliasco (TO), Italy, [2] Department of Agriculture and Ecology, University of Copenhagen, Thorvaldsensvej 40, 1871 Frederiksberg, Copenhagen, Denmark, [*] Corresponding author. E-mail: *alessandro.cerutti@unito.it*

Until the middle of the last century, hundreds of different cultivars of apples were grown in Italy, as in many other fruit producing countries. However in the Sixties, with increasing orchard commercialisation, the use of commercial varieties became more common. Ancient cultivars were gradually replaced by commercial varieties and Italian fruit-growing practices underwent significant changes. Indeed more than 70% of the orchards in the Piedmont Region (Northern Italy) only have one cultivar, the commercial variety Golden Delicious. In this region alone, ancient apple germplasm actually consists of about 350 cultivars, 130 of which were recently noted for their qualitative, morphologic and agronomic traits. Ancient varieties are characterised by very unconventional quality traits, such as alternative fruit shapes, skin colour, nutritional values and organoleptic traits (such as crispness, juiciness and flavour). Because of this, they constitute a well defined niche market.

The commercial appeal of such products (and the following marketing) is based both on uncommon quality traits and claimed smaller environmental impacts due to the original vocation of the agronomic properties of the land. Nevertheless specific environmental assessment of the varieties have not been conducted yet.

Therefore, the aim of this research is to conduct a life cycle assessment (LCA) that compares the production of a representative ancient apple cultivar with Golden Delicious production, in order to evaluate significant differences in the environmental impacts. In particular, the objectives of this research are (I) to qualify and quantify the main environmental aspects of ancient apple varieties in Piedmont in order to establish parameters for the sustainability of that product; (II) to evaluate any significant differences with the environmental impact of the Golden Delicious supply chain and (III) to highlight where such differences are located along the supply chain.

The assessment covers the whole supply chain, including agricultural production and its inputs: processing, cooling, storage and transportation up to the consumer's phase. Storage and consumption within the consumer's house have not been included because they are considered to be the same regardless the supply chain. The functional unit was 1 kg of apples delivered to the consumer. The study was performed in accordance with the guidelines and requirements of the ISO 14040 standard series and with the cradle-to-gate approach as the basis for the Life Cycle Inventory (LCI) of the study. Data regarding agricultural inputs, consumption and agrotechnique have been obtained directly from the growers, who filled in a questionnaire for the season 2010-2011, and by consultation of Italian production protocols. The environmental aspects of the production of fertilisers and pesticides have been included within the boundaries. Data regarding resource use and supply chain properties have been obtained from retailers through interviews and field surveys.

96. Life cycle GHG and energy balance of organic apples: a case study in Italy

Maurizio Cellura[1,*], Mario Fontana[1], Sonia Longo[1], Marina Mistretta[2], Flavio Paoletti[3]

[1] Dipartimento dell'Energia, Università degli Studi di Palermo, Viale delle Scienze, 90128 Palermo, Italy, [2] Dipartimento Patrimonio Architettonico e Urbanistico (PAU), Università degli Studi Mediterranea di Reggio Calabria, Salita Melissari, 89124 Reggio Calabria, Italy, [3] Istituto Nazionale di Ricerca per gli Alimenti e la Nutrizione (INRAN), via Ardeatina N.546, 00178 Rome, Italy, [*] Corresponding author. E-mail: *mcellura@dream.unipa.it*

The main household environmental impacts are concentrated in food, transport and building sectors. The food sector is responsible for 20-30% of the various environmental impacts due to the final consumptions, and in the case of eutrophication for even more than 50% (Tukker at al., 2006).
Every stage of the production and consumption chain of food, from growing crops, to transportation and storage, manufacturing, distribution, purchasing and consumption, and treatment of waste, has environmental effects. Consumers choices can significantly influence the environmental impacts of production, retail and distribution phases of food (EEA, 2005). In particular, they can choose to consume more organic food, which represents a key factor in the food productive sector, due to the added value of its products, to the socio-economic benefits for the producers and the positive effects on the environment and human health.
The present study is part of a research developed within the project "BIOQUALIA – Nutritional and organoleptic quality and environmental impact of organic productions", and aims to evaluate the energy and environmental impacts of 1 kg of organic apples cultivated in the north of Italy.
The analysis was based on the Life Cycle Assessment (LCA) methodology as regulated by the international standards of series ISO 14040 (UNI EN ISO 14040, 2006a; UNI EN ISO 14044, 2006b).
In detail, the authors identified the supply chain flow charts, the relevant mass and energy flows and the key environmental issues for the assessed product, following the approach "from farm to fork". Particular attention was paid on key issues, such as primary energy consumption, water exploitation and fertilisers use in agricultural activities.
The application of LCA allowed assessing the incidence of each life cycle step of apples on the total impacts and identifying "hot spots" of the examined supply chain, by the identification of phases and processes that are responsible of the largest impacts.
In detail, the results showed an average primary energy consumption of about 7 MJ/kg and a global warming potential of about 0.5 kg CO_{2eq}/kg. A relevant incidence on the total impacts (about 70% of primary energy consumption and global warming potential) was related to the transport of apples to final users, hypothesising that the product is distributed on local (10% of the product), national (49%) and international markets (50%). The use of insecticides and the consumption of diesel for agricultural machines were found to be also significant in the overall energy and environmental impacts of apples. Finally the authors carried out a comparison between the outcomes of the presented study and the eco-profile of non organic apple production.

References

European Environment Agency, 2005. Household consumption and the environment. EEA report No 11/2005.

Tukker, A., Huppes, G., Guinée, J., Heijungs, R., de Koning, A., van Oers, L., Suh, S., Geerken, T., van Holderbeke, M., Jansen, B., Nielsen, P., May 2006. Environmental Impact of Products (EIPRO). Analysis of the life cycle environmental impacts related to the final consumption of the EU-25. Technical Report EUR 22284 EN. European Commission.

UNI EN ISO 14040, 2006a. Environmental management – Life cycle assessment – Principles and framework.

UNI EN ISO 14044, 2006b. Environmental management – Life cycle assessment – Requirements and guidelines.

97. Life cycle assessment combined with eMergy for the evaluation of an organic apple production system

Francesca Rossetti[1,*], Benedetto Rugani[2], Elena Neri[1], Nicoletta Patrizi[1], Simone Bastianoni[1]

[1] Ecodynamics Group, Dipartimento di Chimica, Università degli Studi di Siena, via A. Moro, 2 - 53100 Siena, Italy, [2] Public Research Centre Henri Tudor (CRPHT) - Resource Centre for Environmental Technologies (CRTE), 66, rue de Luxembourg - P.B. 144, L-4002, Esch-sur-Alzette, Luxembourg, [*] Corresponding author. E-mail: *francesca.rossetti@unisi.it*

Traditionally organic farms aim to limit as much as possible external inputs to concentrate the production towards more sustainable systems. Organic systems aim to favouring the preservation of ecosystems and the conservation of the landscape and of local complexity. But the management of a farm by means of organic practices does not always assure its sustainability. In this work we combined Life Cycle Assessment (LCA) and eMergy to study the sustainability and environmental performance of an organic farm producing fresh apples and apple-juice. The parallel application of these two methodologies allows obtaining more significative and comprehensive results and it emphasises the necessity of a link between ecological and economic analyses in agricultural systems (Odum, 1996). The farm examined is situated in the south of Tuscany (Italy), with an annual yield of 34 ton/ha. The farm is managed in conformity to organic farming rules, paying particular attention to the preservation of biodiversity and to the respect of natural resources. The use of drinking water is almost completely substituted by the capture of rainwater and by the renewable use of well water. The farm is equipped with a photovoltaic system that satisfies the own electricity demand. For the analysis, the production process was divided in three phases: in Phase 1 apples are cultivated and collected; in Phase 2 apples are washed and selected, a part is destined to the fresh market and a part is transformed in apples-juice (Phase 3). LCA analysis was performed using SimaPro 7.3 (PRè Consultants); for the characterisation we have selected impact categories from CML 2 Baseline Method 2000 (Guinèe et al., 2001): Acidification (AC), Eutrophication (EU), Global Warming Potential (GWP100) and Photochemical Oxidation (PO). Despite what is usually observed in literature for conventional farms (Milà-i-Canals et al., 2006), LCA results showed that Phase 1 is the less critical of the system (AC 7%, GWP 7%, EU 17% and PO 6%). These results are justified by the fact that the farm, according to regulations, reduces at minimum the use of fertilisers and pesticides, decreasing the generation of an important relevant share of many impact categories considered in LCA for this process. Phase 2 and 3 represent the most detrimental (Phase 2: AC 24%, GWP 30%, EU 21% and PO 17%, Phase 3: AC 68%, GWP 64%, EU 62% and PO 76%). In Phase 2 machineries, transport and fuels represent the major impacts, while in Phase 3 the most critical input corresponds to the glass packaging phase. Emergy results highlight a quasi-self-sufficiency of the considered system. In fact the imported flows (F) are very low and it is remarked also by the EIR indicator. The% renewability (%R) is nearly 80%. Transports have a considerable weight in the amount of impacts due to the purchase of apples from other regions that the farm sells as fresh market. The transport of fruits through long distances has usually a relevant impact, and the farm should limit or totally eliminate this unsustainable scenario. Finally, glass represents another major input of the farm and probably the use of a lighter type of glass would decrease impacts related to the bottling phase. Results obtained from the two methodologies provide a wide range of useful information to better identify environmental hotspots of production systems and also to communicate to producers the opportunities to improve their sustainability.

References

Guinèe, J. B., Gorreèe, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. A., de Bruijn, H., van Duin, R., Huijbregts, M. A., 2001. Handbook on Life Cycle Assessment. Operational Guide to the ISO standards, Kluwer Academic Publisher: Dordrecht, The Netherlands

Milà i Canals, L., Burnip, G.M., Cowell, S.J., 2006. Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): Case study in New Zealand. Agriculture Ecosystems & Environment 114, 226-238.

Odum, H.T. 1996. Environmental Accounting: Emergy and Environmental Policy Making. John Wiley and Sons, New York. 370 pp.

98. Preliminary research on the analysis of life cycle assessment in the production of rapeseed and biodiesel in Poland

Magdalena Borzecka-Walker*, Antoni Faber, Rafał Pudełko, Katarzyna Mizak

[1] Institute of Soil Science and Plant Cultivation-State Research Institute, * Corresponding author. E-mail: *mwalker@iung.pulawy.pl*

The aim of the project is to gather necessary information to perform LCA analysis on rapeseed production and its conversion to biodiesel, the execution of these studies according to ISO standards adopted by LCA software as well as a professional presentation of the survey results at different levels of administration and biodiesel production plants.

Poland, like every other country in the EU is committed to reducing greenhouse gas emissions. The main road leading to this is to be the increased use of RES. In this situation, particularly important research, studies, analysis and technical-organisational measures are aimed at securing the implementation of its obligations for Directive 2009/28/EC of 23 April 2009. From the moment when the Directive comes into force in order to promote RES, bioethanol or biodesiel must ensure the reduction of greenhouse gas emissions by 35% and from 2017 by 50%, moving up to 60% in 2018 compared to conventional fuels. Reducing emissions is to be determined by life cycle analysis (LCA).

The developed database contains the information on energy costs over the cycle of cultivation, harvesting and storage and processing of biomass under Polish conditions. Survey (200) was conducted on farms producing rapeseeds for biofuel production.

The average GHG emission was 24.5 g eq. CO_2 MJ^{-1}. The LCA analysis for the production and processing of biomass to biodiesel were performed with SimaPro. The greenhouse gas emissions were incorporated into an assessment of environmental impacts such as acidification, eutrophication, loading with organic compounds, inorganic, toxicity to humans and ecosystems, and carcinogenic impact. The results from the research will be adjusted to a scale of administrative regions in accordance with the requirements from the European Commission, where member states are obliged to present the results of LCA for liquid biofuels for administrative units NUTS-2.

References

Directive 2009/28/EC. The European Parliament and of the Council on the promotion of the use of energy from renewable sources amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.

99. The construction of a database for the evaluation of greenhouse gas emissions from cultivation of crops for biofuels in Poland

Magdalena Borzecka-Walker*, Katarzyna Mizak, Antoni Faber, Alina Syp, Rafal Pudelko

Institute of Soil Science and Plant Cultivation-State Research Institute, * Corresponding author. E-mail: *mwalker@iung.pulawy.pl*

Poland, like every other country in the EU is committed to reducing greenhouse gas emissions. The main road leading to this is to be increased use of Renewable Energy Sources (RES). In this situation, particularly important research, studies, analysis and technical-organisational measures aimed at securing the implementation of its obligations Directive 2009/28/EC of 23 April 2009. From the moment when the Directive comes into force in order to promote RES, bioethanol or biodesiel must ensure the reduction of greenhouse gas emissions by 35% and from 2017 by 50%, moving up to 60% in 2018 compared to conventional fuels. Reducing emissions is to be determined by life cycle analysis (LCA).

The study included farms producing raw materials that can be used to produce liquid biofuels. The farms (1500) producing winter wheat, maize and oilseed rape were selected at random and surveyed thoroughly. The sample size was set at 3% the number of farms producing or likely to produce raw materials for biofuels production purposes. These farms produced raw materials in different soils and different weather conditions between 2005 and 2010 with the exception of extreme conditions (especially farms located in flooded areas in 2010).

References

Directive 2009/28/EC. The European Parliament and of The Council on the promotion of the use of energy from renewable sources amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC

100. Environmental implications of using biomass versus fossil fuels for energy production: the case of willow, an energy crop

Thu Lan T. Nguyen*, John E. Hermansen

Dept. of Agroecology, Aarhus University, Blichers Allé 20, Postbox 50, DK-8830 Tjele, Denmark, * Corresponding author. E-mail: *ThuLan.ThiNguyen@agrsci.dk*

Fossil fuel use for energy production is facing serious problems related to resource depletion and environmental degradation, notably climate change. Biomass fuels e.g. wood waste, crop residues, energy crops, in contrast, are considered renewable and carbon neutral. Unlike fossil fuels that take millions of years to be available as an energy source, biomass can be regenerated relatively quickly through photosynthesis. Biomass burning releases CO_2 back to the atmosphere but this biogenic CO_2 is not counted as contributing to global warming. Apart from wood waste and crop residues, energy crops e.g. willow and miscanthus have recently received large attention as a potential source of renewable energy. Whereas biomass is "carbon neutral" when burned, the inputs used to produce it may be a source of CO_2 and other GHG emissions.
Our research questions were: (1) What if the upstream impacts of energy production from biomass fuels, i.e. those connected with biomass cultivation and distribution, are included?, (2) In addition to global warming and non-renewable energy, what about other impact categories like acidification, eutrophication, ecotoxicity, human toxicity, etc., and 3) How to account for trade-offs among different impact categories? A thorough and comprehensive analysis is thus necessary to assess the sustainability of substituting biomass for fossil fuels in energy production. The task is not only to include more impact categories than global warming and non-renewable energy but also to perform the analysis at a more aggregated level, i.e., translating impacts in different mid-point categories into a single unit so that they can be weighted and added together to give a single score value.
In this paper, we present as a case study the results of an LCA study on electricity generation from willow produced on arable land, in comparison with fossil fuels. Inventory data for the entire process from willow cultivation to energy production were from Nielsen and Illerup (2003) and Uellendahl et al. (2008). Midpoint impact categories considered were global warming, non-renewable energy, acidification, eutrophication, respiratory inorganics, human toxicity, ecotoxicity, photochemical ozone and nature occupation. All midpoint impacts were then translated into a single monetary unit. The LCIA method used was Stepwise 2006 (Weidema, 2009). For a verification of the monetarisation scheme used in the Stepwise2006 method versus that used in previous studies, a sensitivity analysis was also performed.
The midpoint impact assessment shows that substitution of willow for fossil fuels would bring both environmental benefits and costs. The substitution for coal offered environmental benefits in all impact categories considered except for nature occupation and eutrophication. The substitution for natural gas reduced impacts on human toxicity, ecotoxicity, global warming and non-renewable energy but increased nature occupation, eutrophication, respiratory inorganics, acidification and photochemical ozone. The results at the aggregated level show that energy production from willow scores better than from coal (0.11 *vs.* 0.12 EUR/kWh) but worse than from natural gas (0.11 *vs.* 0.06 EUR/kWh). Much of this inferior performance is accounted for by the impact on nature occupation of biomass fuel crops; nature occupation is by far the main contributor with a share of approx. 80% of the aggregated single score. Nature occupation thus remains a major environmental hotspot for bioenergy development, stressing the importance of seeking improvements in relation to this indicator in order for biomass fuels like energy crops to be a viable fuel source.

References

Nielsen, M., Illerup, J.B., 2003. Emissionsfaktorer og emissionsopgørelse for decentral kraftvarme, Eltra PSO projekt 3141, Kortlægning af emissionsfaktorer fra decentral kraftvarme, Delrapport 6, Faglig rapport fra DMU, nr. 442.
Uellendahl, H., Wang, G., Moller, H.B. et al., 2008. Energy balance and cost-benefit analysis of biogas production from perennial energy crops pre-treated by wet oxidation. Water Sci Technol. 58, 1841-1847
Weidema, B.P., 2009. Using the budget constraint to monetarise impact assessment results. Ecological Economics 68, 1591-1598.

101. Influence of allocation methods in the quantification of the environmental impacts of compost application

Julia Martínez-Blanco[1,*], Joan Rieradevall[1,2], Pere Muñoz[3], Assumpció Antón[3,4]

[1] Institute of Environmental Science and Technology (ICTA), Universitat Autònoma de Barcelona (UAB), 08193 Bellaterra, Barcelona, Spain, [2] Chemical Engineering Department, Universitat Autònoma de Barcelona (UAB), 08193 Bellaterra, Barcelona, Spain, [3] Institute of Research and Technology in Agrifood Sector (IRTA), Environmental Horticulture, 08348 Cabrils, Barcelona, Spain, [4] Departament d'Enginyeria Química, Universitat Rovira i Virgili (URV), 43007 Tarragona, Spain, [*] Corresponding author. E-mail: *julia.martinez@uab.cat*

Fertilisation has been reported as one of the key factors for the environmental performance of horticulture products, which are often highly nutrient-demanding, particularly for systems of low technology (Antón et al., 2005; Romero-Gámez et al., 2011). An special case, among fertilisers, is compost. Our previous studies have indicated that compost fertilisers have higher environmental impacts than mineral ones in some impact categories when yields are used as reference flow (Martínez-Blanco et al. 2011).
Compost is also known to be a slow-release fertiliser, usually applied for periods of 1-2 years, considering the nutritional necessities of the annual or biannual crop cycle, in order to avoid individual environmental burdens of transport and machinery for each crop. Therefore, compost is applied at the beginning of the rotation, but is supplying nutrients to all the considered crops.
Therefore, when the burdens of compost production and application have to be distributed among the crops in the cycle, a multifunctional problem arises. According to ISO 14044, wherever possible, allocation should be avoided by dividing the unit process or expanding the product system. Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products underlying physical relationships, as a first option, or other relationships (e.g. economic value).
Accordingly, the main goal of our study is the comparison of the environmental performance of four crops in a rotation, which are applying compost and mineral fertilisers, when several methods for compost burdens distribution are used.
For the study, four experimental Mediterranean crops (chard, tomato, cauliflower and onion) in a rotation are compared and the reference flow hectare is selected. Three main stages are included within the boundaries of the system: fertiliser production, transport and cultivation. Most of this data were obtained experimentally in the fields and the composting plants, by the authors or from previous research of the group (Martínez-Blanco et al., 2011). When local information was not available, bibliographical sources and the database v2.0 were used. The environmental assessment is following the obligatory classification and characterisation phases defined by the ISO 14044 and four mid-point impact categories are considered (acidification, eutrophication, abiotic depletion and global warming).
Four alternative approaches for compost burdens distribution are compared here: (1) system expansion, i.e. the impacts of mineral fertilisers avoided due to compost use are subtracted from the total burdens; and three allocation alternatives (2) according to compost delayed mineralisation; (3) according to nitrogen crop requirements; and (4) according to economic allocation using cost-benefit values.
Compost burdens distributions are different for the four approaches and thus impact differences with the mineral fertiliser traditional option, for each one of the crops in the rotation, are detected.

References

Antón, A., Montero, J.I., Muñoz, P., Castells, F., 2005. LCA and tomato production in Mediterranean greenhouses. Int. J. Agric. Resour. Gov. Ecol. 4(2), 102–112.

Romero-Gamez, M., Anton, A., Soriano, T., Suarez-Rey, E.M., Castilla, N., 2011. Environmental impact of screenhouse and open-field cultivation using a life cycle analysisi. The case study of green bean production. J Cleaner Prod. On-line, doi:10.1016/j.jclepro.

Martínez-Blanco, J., Muñoz, P., Antón, A., Rieradevall, R., 2011. Assessment of tomato Meditarrean production in open-filed and standard multi-tunnel greenhouse, with compost or mineral fertilizers, from an agricultural and environmental standpoint. J Cleaner Prod. 19, 985-997.

102. How to overcome time variation in LCA

Saioa Ramos*, Guillermo Pardo, Jaime Zufia

Food Research Division, AZTI-Tecnalia, * Corresponding author. E-mail: *sramos@azti.es*

When performing an environmental impact analysis of food products and processes, LCA appears as a suitable tool. However, there are a number of uncertainties that need to be faced. Current studies regarding life cycle analysis assessment of food products have reported high variability on the results depending mainly on the quality, reliability and significance of the inventory data of the process or product. Functional unit and allocation or selection of system boundaries are other factors that influence directly on results.

All those mentioned factors have been deeply studied; nevertheless, time dependant variations are, still today, a poorly studied variable (Reap et al., 2008). It has been shown that regarding to primary production, and more specifically to the extraction of wild resources, this time scale could lead to an important variability on the obtained results (Ramos et al., 2011). This is due to unpredictable external factors that affect the performance of the activity such as environmental conditions or whether episodes. Regarding food chains there are several sub-sectors which are susceptible to show variations on the impact characterisation depending on the selected period of time.

On the topic of fisheries, recent study has shown great differences on the Global Warming Potential per ton of landed fish when performing a Timeline LCA of the purse seine Basque fleet (Ramos et al., 2011). Furthermore, significant variation in agricultural yields has been also suggested. Thus, in recent study on winemaking by Vazquez-Rowe et al., 2012, significant variability of about 20% is also described for the eutrophication potential in a range of 4 years.

To overcome timeline matters, an approach with Basque trawling fleet have done using Data Envelopment Analysis (DEA) combined with LCA (Iribarren et al., 2010). DEA was implemented to identify possible variations between different years for each impact category along the selected period of time. This methodology analyses differences in the efficiency of every single ship of each year and compare different impact categories between the years and between each ship.

Results reported variations up to 25% in all the environmental impact categories between studied years. However, when comparing operational efficiencies between fishing vessels on each year, variations up to 10% have reported. Therefore, DEA+LCA analysis suggest that for the Basque trawling fleet there is no considerable potential to reduce the environmental impact due to the fact that almost all the ships showed similar efficiencies.

On the whole, there is a need to evaluate a wide range of years when performing a primary production LCA. Moreover, it is suggested that combining LCA with DEA could lead to an eco-efficiency benchmarking analysis, in order to support decision-making.

References

Iribarren D, Vázquez-Rowe I, Moreira MT, Feijoo G. Further potentials in the joint implementation of Life Cycle Assessment and Data Envelopment Analysis. Sci Total Environ 2010;408(22):5265–72.

Ramos, S., Vázquez-Rowe, I., Artetxe, I., Moreira, M.T., Feijoo, G., Zufia, J.. 2011a. Environmental assessment of Atlantic mackerel (*Scomber scombrus*) in the Basque Country. Increasing the time line delimitation in fishery LCA studies. Int J of Life Cycle Assess 16 (7), 599-610.

Reap, J., Roman, F., Duncan, , Bras, B. 2008. A survey of unresolved problmens in life cycle assessment – Part 1: goal and scope and inventory analysis. Int J Life CyCle Assess 13:290-300.

Vazquez-Rowe., I, Villanueva-Rey, P. Moreira, Feijoo, G. 2012. Environmental analysis of Ribeiro wine from a timeline perspective: Harvest year matters when reporting environmental impacts. Journal of Environmental Management 98:73-9.

103. The role of spatial modelling using GIS in the development of life cycle inventory for Australian agriculture

Jonathan Hercule[1], Sandra Eady[2,*], Tim Grant[3], Russell Lyons[4]

[1] AgroParisTech, Paris, France, [2] CSIRO, Locked Bag 1, Armidale NSW 2350, Australia, [3] Lifecycle Strategies, Swanson Street, Melbourne VIC 3000, Australia, [4] CSIRO, 306 Carmody Road, St Lucia QLD 4067, Australia, [*] Corresponding author. E-mail: *sandra.eady@csiro.au*

In Life Cycle Assessment, spatial data has been used in a range of applications: to enable site or region specific impact assessment, to track the flow of pollutants through the environment, for biodiversity assessments of land use, and for developing new impact categories such as desertification. However, there has been limited application of spatial data to generate underlying life cycle inventory. This paper examines how spatial data can assist in building a national inventory for agriculture, where a consistent approach across the nation is required, and detailed data collection at all sites or regions is not feasible.

One of the first tasks in establishing national life cycle inventory for agriculture is to define the major production systems that represent the country's production. In some instances, this can be approached using GIS layers to describe land systems within which the production of an agricultural commodity is likely to be relatively consistent. This approach has been applied to Australian wheat and wool production systems, with combinations of GIS layers for soil type, rainfall and land use being used to identify relatively homogeneous regions for production. The goal of national inventory is to present data with a structure that allows both regional differences in production systems to be identified as well as inventory in a format that is appropriate for the next user in the supply chain. GIS layers can be used to provide data to make this inventory spatially explicit.

Emissions to the environment are often dependant on the geo-location of the agricultural production system; this includes emissions such as pesticides, nutrient discharge to waterways and indirect N_2O emissions from fertiliser and animal waste, where regional differences in rainfall, temperatures and soils can be major determinants of flows to the environment. By geo-locating agricultural production these flows can be estimated in a consistent manner across the nation by using appropriate GIS layers.

Inputs from the techno-sphere for a number of important agricultural operations are influenced by factors related to the geo-location of the production system. As part of the AusAgLCI project we have been investigating means of using GIS data layers to standardise and simplify the choice of inventory for agricultural production, so that important factors determining variation in inputs are accounted for without the need for detailed individual research by the LCA practitioner. As long as the geo-location of the production system is known, standard data can be accessed to give appropriate inventory for that region. These include inputs such as pumping energy required for irrigation (largely determined by the height water needs to be pumped) and fuel inputs for cultivation (largely determined by clay content of the soil). With GIS data now at the scale of $5km^2$ or less, it becomes feasible to use this resource to accurately represent the local conditions for agricultural production.

There are a number of ways in which GIS data can be used to enhance the development of LCI, in a manner that assists with consistency for national databases, allows a level of automated updating, and improves the accuracy of data inputs for production systems. The challenge is in turning these opportunities into reality, with the provision of easy to use interfaces between GIS data and LCI, a step that is only just commencing for enabling the use of GIS data.

104. Using spatial data to define industry sub-sectors for Australian wheat

Gonzalo Mata[1,*], Sandra Eady[2], Tim Grant[3]

[1] CSIRO Animal Food and Health Sciences, Private Bag 5, Wembley WA 6913, Australia, [2] CSIRO Animal Food and Health Sciences, Locked Bag 1, Armidale NSW 2350, Australia, [3] Life Cycle Strategies Pty Ltd, 125 Swanston Street, Melbourne Vic 3000, Australia, [*] Corresponding author. E-mail: *Gonzalo.Mata@csiro.au*

A Life Cycle Inventory (LCI) for Australian agriculture is currently in development. Within each industry (e.g. crops, livestock, horticulture) sub-sectors need to be identified so that the LCI can be structured in such a way users can identify the most appropriate on-farm production process for their supply chain. For instance, major users of Australian wheat purchase grain based on grade (grain hardness and protein content) rather than the region or production system employed to produce the wheat. However, primary sources of information to describe production data don’t necessarily relate well to grade classifications, as grain quality expectations can be affected due to environmental interactions or manipulated through blending of various sources of grain to meet market specifications post-farm gate.

Therefore, in developing a national inventory for wheat it is necessary to define these production systems as regional sub-sectors of the industry, decide how many sub-sectors are needed to represent important differences in environmental impacts, and from these systems construct inventory processes that have utility to the downstream users of Australian wheat.

This paper will explore the GIS methodology needed to define regional sub-sectors for the Australian wheat industry, using a combination of industry expertise and spatial data on land use, soil types, rainfall and other available statistical boundaries and biophysical parameters to capture regional differences which translate into differentiated production systems with differentiated environmental impacts.

105. Improving pesticide accounting in agricultural life cycle assessment: a review of existing LCA practice and available LCA and Ecological Risk Assessment models

Mitchell Burns[1,*], Philippe Roux[2], Carole Sinfort[3], Claudine Basset-Mens[1], Eric Malezieux[1]

[1] CIRAD, Research unit: Hortsys. TA B-103/PS4 Boulevard de la Lironde 34398 Montpellier Cedex 5, France, [2] IRSTEA, Research unit: ITAP, 361 rue JF Breton, 34196 Montpellier, France, [3] SUPAGRO, Research unit: ITAP, 2 place Pierre Viala 34060 Montpellier Cedex 2, France, [*] Corresponding author. E-mail: *mitchell.burns@cirad.fr*

Pesticides require special attention in agricultural Life Cycle Assessment (LCA) because: i) relative to other chemical LCAs pesticides are intentionally applied to the biosphere; ii) they are designed to effect a target group of organisms; and iii) for comparative purposes, they are a characteristic that distinguishes organic and conventional cropping systems (Hauschild 2000). In a complementary paper, van Zelm *et al*. (submitted) characterise an overlap or gap that may exist between LCI (inventory) and LCIA (impact assessment) phases for the toxicity assessment of pesticide emissions in agroecosystems. Specifically, conceptual models or available experimental data used to assess pesticide fate should be compatible with the respective LCI and LCIA phases when considering temporal and spatial scale. This was identified to be an outcome of limited guidance being available for combining location-specific LCI and globally estimated LCIA outputs. However, as LCA has been moving toward more locally explicit impact assessments, similar to that of Ecological Risk Assessment (ERA) (USEPA 1998), accurate characterisation of more complex interactions become increasingly important and the models used in LCI and LCIA phases should be adapted.

An important conceptual difference between LCA and ERA is that they respectively estimate “potential” global and “true” local impacts. It is common in LCA that generic emission factors are used to estimate the extent of chemical distribution between the air, water and soil phases of the environment, with limited accounting of fate beyond the agricultural parcel gate. In contrast, ERA accounts for more complex fate phenomena to define the distribution and emission of pesticides beyond the farm gate, as accurate accounts of these processes are important for risk management. The development of more complex emission models (e.g. PestLCI; Birkved and Hauschild 2006) has seen LCA move toward this level of complexity. However, an inventory of pesticide application and fate management techniques (e.g. buffer zones, etc) according to crop type is needed to improve accurate estimations of chemical distribution and fate for such modelling efforts. This paper presents a review of how LCAs have typically accounted for pesticide fate and distribution in the field together with the main available fate models benchmarked against environmental fate (including transport, transfer and degradation) characterisation approaches typically used in ERA, and proposes some methods to improve this area of research.

References

Birkved, M., Hauschild, M.Z., 2006. PestLCI - A model for estimating field emissions of pesticides in agricultural LCA. Ecol. Model. 198, 433-451.

Hauschild, M. Z. 2000. Estimating pesticide emissions for LCA of agricultural products, in Weidema, B.P., Meeusen, M.J.G (Eds.), Agricultural data for life cycle assessments, vol. 2. LCANet Food, The Hague, pp. 64-79.

United States Environmental Protection Agency (USEPA) 1998. Guidelines for Ecological Risk Assessment, USEPA, Washington.

van Zelm, R., Larrey-Lassalle, P., Roux, P., (submitted). Bridging the gap between LCI and LCIA for toxicological assessments of pesticides used in crop production: application to banana growing, in: 8th International Conference on LCA in the Agri-Food Sector, Saint Malo, France, 2-4 October 2012.

106. Comparison of assessment methods for the environmental impacts of pesticide production

Kiyotada Hayashi[1,*], Koichi Shobatake[2], Naoki Makino[2]

[1] National Agriculture and Food Research Organization, Japan, [2] TCO2 Co. Ltd., Japan, [*] Corresponding author. E-mail: *hayashi@affrc.go.jp*

With regard to the environmental impact of agricultural production systems, the proportion of pesticide production is smaller than that of, for example, chemical fertiliser production (Hayashi, 2011). Although life cycle inventory (LCI) analysis of pesticides has been limited, there have been recent development in estimation techniques for the environmental impacts of pesticide production (Wernet et al., 2008) and for the LCIs of pesticides (Sutter, 2010). However, the differences between the various assessment methods are not well understood, and the practical implications for LCI data construction are not known. Therefore, this study compares different assessment methods for the environmental impacts of pesticide production.

The assessment methods considered in this study include (1) assessment based on ecoinvent 2.2 (hereafter, ecoinvent), (2) estimation using the Finechem tool (hereafter, Finechem), and (3) estimation using emission factors derived from input-output tables for Japan (hereafter, IO). We conducted two comparisons, namely, one between ecoinvent and Finechem, and the other between ecoinvent and IO. Twenty active ingredients were analysed in the first comparison, and 52 pesticide products (13 active ingredients) were assessed in the latter. Global warming (IPCC 2007 GWP 100a) was used as the impact category. S-PLUS (TIBCO Spotfire S+® 8.1J for Windows) was employed for statistical analyses such as regression analysis.

The result of the first comparison showed that it is difficult to find a correlation between the results obtained from ecoinvent and from Finechem (Fig. 1). In general, the variability (standard deviation) in the case of Finechem was larger than that in the case of ecoinvent. The result of the second comparison demonstrated that the determination coefficients of the regression analyses were sufficiently large, and that the regression coefficients were significant at the 1% or 5% levels. The estimated values based on IO tended to be 5 times or 10 times larger than those based on ecoinvent (Fig. 2, Table 1).

These results indicate that under the assumption that estimated values based on ecoinvent are reliable, further study is necessary for developing reliable estimation methods. In addition, although the values obtained from ecoinvent can be predicted from the values obtained from IO, adjustments may be necessary because of the tendency to overestimate in the latter. The dependence of the results on the selection of the assessment method is expected to be resolved by further development of life cycle inventories.

References

Hayashi, K., 2011. Assessing management influence on environmental impacts under uncertainty: a case study of paddy rice production in Japan, in: Finkbeiner, M. (Ed.), Towards Life Cycle Sustainability Management. Springer, pp. 331-340.

Sutter, J., 2010. Life cycle inventories of pesticides, Swiss Centre for Life Cycle Inventories, St. Gallen.

Wernet, G., Hellweg, S., Fischer, U., Papadokonstantakis, S., Hungerbühler, K. 2008. Molecular-structure-based models of chemical inventories using neural networks. Environ. Sci. Technol. 42, pp. 6717-6722.

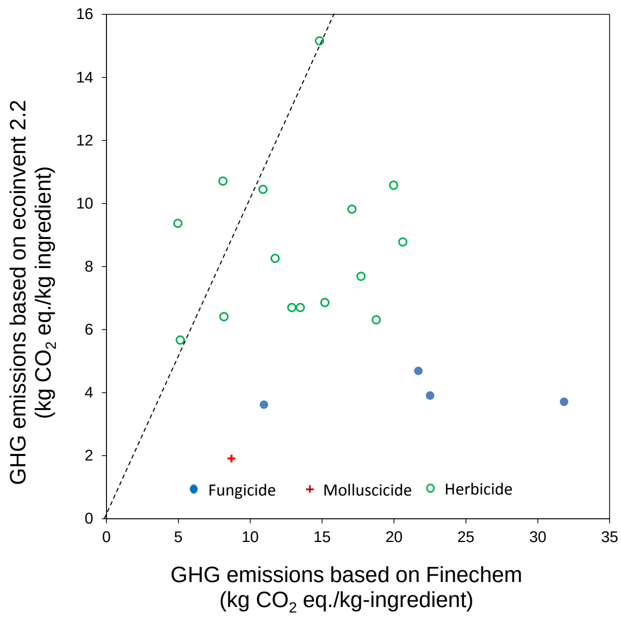


Figure 1. The relationship between greenhouse gas (GHG) emissions based on Finechem and those based on ecoinvent 2.2.

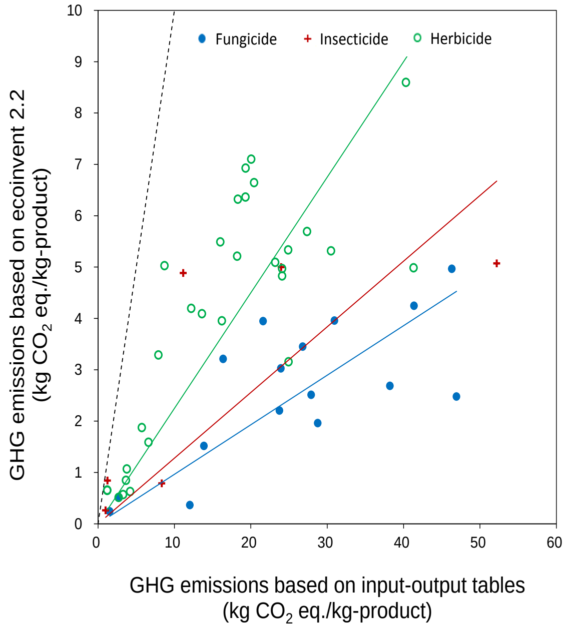


Figure 2. The relationship between greenhouse gas (GHG) emissions based on input-output tables and those based on ecoinvent 2.2.

Table 1. The results of regression analysis without intercepts

Type of pesticide	Adjusted R-square	Regression coefficient
Fungicides	0.881	0.097**
Insecticides	0.702	0.128*
Herbicides	0.880	0.225**

* Significant at 5% level.
** Significant at 1% level.

107. Implementing decision making in irrigation management based on productive and environmental indicators

Maria José Amores[1,*], Francesc Ferrer[2], Orene Cabot[2], Assumpció Anton[1,3], Albert Duaigües[2], Francesc Castells[1], Eskinder Demisse Gemechu[1]

[1] Universitat Rovira i Virgili, Tarragona, Spain, [2] LabFerrer. Centre Assessoria Dr Ferrer SL, Cervera (Lleida), Spain, [3] Institut de Recerca i Tecnologia Agroalimentaria, Enginyeria i Biosistemes. Cabrils (Barcelona), Spain, [*] Corresponding author. E-mail: *mariajose.amores@urv.cat*

Irrigated agriculture faces the need to improve water management practices at a farm level from a productive, social and environmental point of view. To evaluate the performance from a productive and water use efficiency point of view, some indices have been traditionally used (Hoffman et al., 2007; Fessehaziona et al., 2011 and Stirzaker, 2011). They could be classified in three categories: first, Water Productivity or Irrigation requirements based on water consumption, yield and evapotranspiration (ET) estimations; second, Leaching fraction or drainage requirements for soil and water salinity control; and third, Irrigation system performance. To be useful to the irrigator, these indexes should be calculated for each Farm Management Unit (FMU), which may correspond to a particular field where water consumption, potential water crop requirements (ETc), yield, manpower, machinery and other parameters could be assigned. At the end of the season, these indicators could help assessing its strategy, by benchmarking the FMUs performance, think about changes or improvements that may have a beneficial impact on the crop and water use performance of the farm. To be able to steer irrigation management along the growing season and make tactical decisions, reliable information should be obtained at different scales (FMUs, farm and watershed). Assessing practices to reduce the environmental impact of irrigation (mainly, avoiding consumptions beyond real necessities and reducing the impact of leaching and erosion) will have to be done at a FMU scale and be integrated into the manager’s dashboard. Therefore, the quantification of water use sustainability indicators should be based on a solid and simple conceptual model, so it can be integrated into the farmer’s decision making processes.
This work presents an attempt to implement an inventory procedure at a Management Unit level with the aim of calculating production, water use and environmental indicators to quantify and assess the impact of irrigation, and to integrate it into the managers’ dashboard to make strategical decisions. We believe that the results from this project could help orienting the application of the much more complex Life Cycle Assessment (LCA) to quantify and assess the environmental impact of irrigation considering a system beyond the limits of the farm. As a practical trial of assessment of sustanaibility and decision making a case study was carried out in the Ebro Valley near Lleida (NE Spain) where three irrigated farms where chosen during the 2011 growing season. The crops were vineyard, nectarine and corn for silage. In this first year, the goal of the project was to construct a web-based program to calculate the basic production and water use efficiency indicators, based on real data and a solid conceptual model. The idea behind was to validate the results for the studied FMUs, to test if this procedure can be used at a larger farm scale (with many FMUs) and to assess the eventual insertion into the software of environmental sustainability indicators.
By and large, there are several criteria and methods to assess environmental sustainability at farm level, so it provides some increasing variability and uncertainty. Hence, it is needed assessing that from statistical point of view or from large regions. However, that issue lead us to discuss about underestimating or overvaluing the sustainability, highlight the importance of geographical reference units and quantify the uncertainty that these decisions could have to choose the correct criteria for decision makers.

References

Fessehaziona, M.K., R.J. Stirzaker, J.G. Annandale, C.S. Everson.2011. Improving nitrogen and irrigation water use efficiency through adaptive management: A case study using annual ryegrass. Agriculture, ecosystems, environment 141 (2011): 350-358

Hoffman, G.J., R.G. Evans, M.E. Jensen, D.L. Martin, R.L. Elliott (Eds.). 2007. Design and operation of farm irrigation systems. 2nd Edition. ABABE, MI, USA.

Stirzaker, R.J. 2011. Strategy, tactics and heuristics for managing solutes in horticultural crops. Proc. 6th IS on Irrigation of Horticultural crops. Ed.: S. Ortega-Farias and G. Selles. Acta Hort. 889, ISHS 2011

Table 1 Results for the studied Farm Management Units (FMUs) in 2011.

FMU	Area (ha)	Yield (kg ha^{-1})	Water consumption[1] (m^3 ha^{-1})	Energy consumption[2] (kWh)	Acc[3]. ETo (mm)	Acc[3]. Precip. (mm)	WUE[4] (kg m^{-3})	EUE[5] (kg kWh^{-1})
Vineyard	4.2	11,935	2,040	2,859	838	93	4.36	4.17
Nectarine	12.5	44,780	5,424	3,300	897	272	5.86	13.57
Corn	70.0	14,286	6,750	3,000	595	81	1.96	4.76

[1]The irrigation season for each crop was as follows: Vineyard (28 March to 26 Sept 2011; 26 weeks), Nectarine (7 March to 31 Oct 2011; 34 weeks) and Corn (23 May to 12 Sept, 2011; 16 weeks). Irrigation uniformity and application efficiency of the irrigation system was estimated from default values, with sprinklers in the corn and drippers in the other two crops.
[2] Energy consumption (kWh) was provided by the farms' manager.
[3]Weekly weather data (Precipitation and Reference ET) was gathered from nearby automatic monitoring weather stations considering the irrigation season for each crop. Crop coefficients (kc = ETc/ETo) were obtained from local extension agents.
[4]Water Use Efficiency (WUE), considering applied water + effective precipitation.
[5]Energy Use Efficiency (EUE), considering the energy used to pump the water.

Acknowledgements: This work was partially financed by the VALOR 2010 funding from the Generalitat de Catalunya (Spain). We also would like to thank the valuable advice from Prof. Josep M. Villar (University of Lleida). We also would like to thank the interest and the help from the farm managers.

108. LCAs for a large repertoire of Finnish outdoor plant products

Merja Saarinen*, Yrjö Virtanen, Helena Hyvärinen

MTT Agrifood Research Finland, * Corresponding author. E-mail: *merja.saarinen@mtt.fi*

The paper presents results of LCA for a large repertoire of outdoor crops produced in Finland in two resource use categories (land and energy) and three impact categories (climate change, eutrophication and acidification). Assessed crops were cereals, potato, rapeseed, pea, broad bean, carrot, beetroot, yellow turnip, parsnip, Chinese cabbage, onion, strawberry and blackcurrant.

The functional unit of LCAs is the kilo of a product at the farm gate (without packaging). System boundaries include production of agricultural inputs and energy in the upstream phases, and use of agricultural inputs and energy, and mechanical work in the production phase on farms. However, potential packaging and storage on farms, and transportation of agricultural inputs to the farm were excluded from the product systems. Emissions from organic soils were excluded. Data on agricultural input manufacture were obtained from industry, which produces most agricultural inputs used in Finland. Energy consumption was based on the Finnish average grid values. Data on the use of inputs for crop production were based on the national agricultural database, comprising data on the cultivation instances of various crops, i.e. primary data. Data on mechanical work were based on physical models. Data represent average Finnish production. Emissions and impacts from domestic animal production (including manure-based emission from animal shelter and storage) were not allocated to the manure used as fertiliser for plant production. There were no other significant allocation issues.

For the climate impact calculation, estimation of N_2O, CO_2 and indirect N_2O emissions from the field were based on the IPCC method and data (IPCC, 2006). Data on NH_3 emissions from the application of fertilisers were estimated based on models from the EEA (European Environmental Agency, 2006). For the assessment of eutrophication, site-specific nitrogen and phosphorus leaching models and site-dependent factors were applied.

The climate impact and acidification of rapeseed was by far the highest, and the lowest was for root vegetables and potato (Table 1). The eutrophication potential of broad bean was highest, followed by Chinese cabbage, and the lowest eutrophication potential was for carrot, followed by oat and barley. Production of rapeseed consumed most energy, and root vegetables the least. Source of energy varied considerably among products.

It is concluded that the priority order of products varies according to impact category, as indicated in Table 1. CO_2 and N_2O are the main emissions that impact climate. Their share differs for different plants. Figure 1 illustrates the significance of different emissions for climate impact.

The study was part of the ConsEnv-project. The results have been used in the LCA for lunch portions (Saarinen et al., 2012) together with LCA results for animal-based products (Usva et al., 2012).

References

European Environment Agency (EEA) 2006. EMEP/CORINAIR Emission Inventory Guidebook - 2006. Web publication. Available at: http://www.eea.europa.eu/publications/EMEPCORINAIR4

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4. Agriculture, Forestry and Other Land Use. IPCC-NGGIP Publications

Saarinen, M., Kurppa, S., Virtanen, Y., Usva, K., Mäkelä, J. and A. Nissinen, 2011. LCA approach to the impact of homemade, ready-to-eat and school lunches on climate and eutrophication. J Cleaner Production. 28, 177-186.

Saarinen, M., Kurppa, S., Virtanen, Y., Usva, K., Mäkelä, J. and A. Nissinen, 2012b. LCA approach to the impact of homemade, ready-to-eat and school lunches on climate and eutrophication. J Cleaner Production 28, 177-186.

Usva, K., Nousiainen, J., Hyvärinen, H. and Y. Virtanen 2012. LCAs of average animal products pork, beef, milk and eggs in Finland. Abstract for LCA Food 2012 Conference.

Table 1. Results of LCAs for plant products produced in Finland.

Product	Climate impact, kg CO_2 eq/kg	Eutrophication, g PO_4 eq/kg	Acidification, g AE/kg	Land use, ha/1000kg	Total energy use, kWh/kg
Barley	0.60	1.27	0.67	0.28	0.557
Rye	0.86	2.00	0.80	0.34	0.656
Wheat	0.68	1.44	0.62	0.26	0.554
Oat	0.62	1.26	0.89	0.29	0.588
Potato	0.08	0.25	0.08	0.04	0.054
Rapeseed	1.48	3.39	1.42	0.71	0.949
Pea	0.41	1.55	0.57	0.41	0.516
Broad bean	0.43	4.56	0.58	0.37	0.641
Carrot	0.06	0.11	0.05	0.02	0.024
Beetroot	0.08	0.21	0.07	0.04	0.027
Yellow tur-	0.08	0.20	0.08	0.03	0.039
nip	0.21	0.37	0.16	0.05	0.050
Parsnip	0.25	0.41	0.21	0.05	0.376
Chinese	0.13	0.31	0.11	0.05	0.053
cabbage	0.47	2.59	0.37	0.30	0.703
Onion	0.45	2.94	0.36	0.33	0.743
Strawberry					
Blackcurrant					

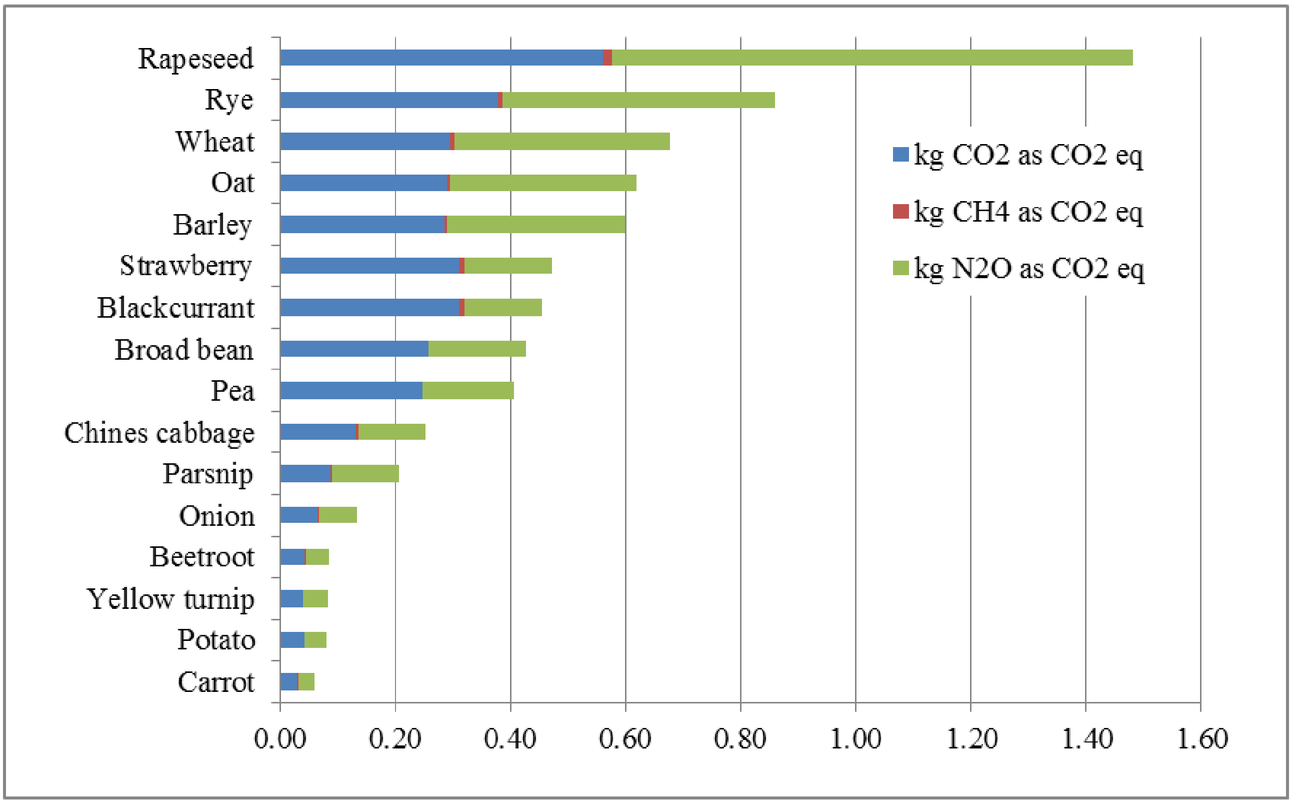


Figure 1. Climate impact per kg product divided by main emissions.

109. Life cycle assessment of long-lived perennial cropping systems: almond and pistachio production in California

Elias Marvinney[1,*], Alissa Kendall, Sonja Brodt, Theodore Dejong

[1]University of California, Davis, California, USA, [*]Corresponding author. E-mail: *emarvinney@ucdavis.edu*

This study characterises typical almond and pistachio orchard production systems in the U.S. state of California. These industrialised agro-ecosystems are of great economic and environmental importance, encompassing more than 361,300 ha of the state's agricultural land and yielding more than 85% of global almond exports and 50% of global pistachio exports. In 2009, 240,000 tonnes of almonds and 82,000 tonnes of pistachio were imported by the European Union (USDA 2009). Commercial nut orchards in California's Central Valley demand significant agrochemical inputs, irrigation, and fuel for mechanised field operations throughout their lifespan - up to 25 years for almond (Duncan et al 2011) and 80 years for pistachio (Beede et al 2008). Due to high-intensity inputs and long lifespans of these systems, the California nut industry is responsible for significant emissions of greenhouse gases (GHGs) and other atmospheric pollutants.

Orchards have the potential to sequester carbon in soils and/or biomass (Kroodsma and Field 2006). In California, much of this biomass is used to produce electricity at regional electricity generation plants, widely distributed in California (Wallace 2007). The potential for sequestration versus emissions offset through use of waste biomass as an energy feedstock is dependent on management characteristics, orchard lifespan, and other factors. Here we present a comparative assessment of the net GHG footprint of California almond and pistachio production, accounting for material and energy inputs of production up to farm gate, excluding processing and distribution as well as annual variation in operations and inputs as trees mature.

The study also explores the potential for carbon sequestration and GHG offset under several possible scenarios for both pistachio and almond. Data were gathered from cost-return studies, farmer surveys, and published literature. Transportation characteristics, in-field emissions from fuel combustion, and soil ecological processes were independently modelled. Our analysis provides information for growers on where to focus GHG reduction efforts and offers insights regarding the trade-offs between energy and material inputs and field emissions under different management scenarios, as well as providing an assessment of the typical greenhouse gas and energy footprint of an economically important export commodity.

References

Beede R. H., Kallsen C. E., Holtz B. A., Klonsky K. M., Moura R. L. D., 2008. Sample costs to establish an orchard and produce pistachios: San Joaquin Valley South – Low volume Irrigation. University of California Cooperative Extension. Davis CA.

Duncan, R. A., Verdegaal, P. S., Holtz, B. A., Doll, D. A., Klonsky, K. A., Moura, R. L. D., 2011. Sample costs to establish an orchard and produce almonds: San Joaquin Valley North – Microsprinkler Irrigation. University of California Cooperative Extension. Davis CA.

USDA Office of Global Analysis, 2009. Tree Nuts: World Markets and Trade. United States Department of Agriculture, Foreign Agriculture Service. Washington, D.C.

Wallace H. N., 2007. Feasibility of a California Energy Feedstock Supply Cooperative. California Center for Cooperative Development. Davis, CA.

110. Land use key parameters to be addressed in life cycle assessment study of soybean grains

Anna Lúcia Mourad[1,*], Lília Sichmann Heiffig-del Aguila[2]

[1] CETEA – Packaging Technology Center / ITAL – Institute of Food Technology, Brazil, [2] Brazilian Agricultural Research Corporation – EMBRAPA Temperate Agriculture, Brazil, [*] Corresponding author. E-mail: *anna@ital.sp.gov.br*

Brazil is considered as one of the largest exporters of agricultural products in the world. The growth of Brazilian agriculture in a continuous and solid way is extremely important to improve the quality of life of millions of Brazilians. A great part of this growth has come from the soy complex (grain, meal and oil) whose exports have more than quadrupled over the last 10 years, reaching the value of US$ 23.8 billion in 2011. In 2011, Brazil produced 74.3 million tons of soy, being ranked as the second largest world producer of soy with 26% of the world crop, estimated at 263.7 million tons. The cultivation of soy occupies the largest area (35.7%) among the products of the annual and perennial crops of the country. Soy is planted-practically all over the country with the Center-west (49%) and South (34%) being two of largest areas. The recent expansion of the crop has taken place in areas of degraded pasturelands. Due to the importance of this crop to the country, the objective of this work is to select important parameters relative to the land use which can be considered in a life cycle assessment study of soy grains. The first selected parameter is the occupation of agricultural lands for this crop. The country has an area of 8.5 million of km^2 of which 37.3% is used for general agricultural and pasture purposes and 25.6% for cultivation of food products such as meat and vegetables. As the parameters for land use have not been established yet for LCA purposes in the country, 2.18 million of km^2 was considered as the reference area for normalisation of land for food production. The average land occupation for the soy crop 1.12 m^2yr per ton produced in 2010. Besides the territorial occupation itself the authors suggest that the total amount of fertilisers in relation to the nitrogen, phosphorus and potassium macronutrients as well as the total amount of pesticides (only actives) used per hectare could be indicative of the human interference on the land. These indicators are independent of the climate, temperature, relief, type of the soil or other factor that minimises the anthropogenic interference due to the capability of nature recovering. They are also independent of time, a key parameter in agricultural impacts. The impact of land use could be evaluated by soil organic matter content as this measure is considered as the one of the best stand-alone indicator of life support functions of land. Soil organic matter, consisting mostly of C, is the largest terrestrial pool in the C biogeochemical cycle. Soil organic matter, although occupying only 5% of the total soil volume, has an important influence in soil physical, chemical and biological properties, directly influencing the productivity of soybean. Management systems capable of maintaining and even increasing the soil organic carbon may stocks contribute to maintaining the productive capacity of soils and to mitigate the problem of increasing atmospheric CO2.

References

Milà i Canals L., Romanya J., Cowell S.J., 2007. Method for assessing impacts on life support functions (LSF) related to the use of 'fertile land' in Life Cycle Assessment (LCA). J. Clean. Prod., 15, 1524-1440.

Guinée JB et al., 2001. Life Cycle Assessment – An operational Guide to the ISO Standards, Leiden.

Instituto Brasileiro de Geografia e Estatística - IBGE, 2012. SIDRA- Banco de Dados Agregados. Produção agrícola municipal.

Associação Brasileira das Indústrias de Óleos Vegetais, 2012. Dados estatísticos do complexo soja.

Brady NC, 1989. Natureza e propriedade dos solos. São Paulo, McGraw-Hill, 878p.

Campos BC et al., 2011. Long-term C-CO_2 emissions and carbon crop residue mineralization in an oxisol under different tillage and crop rotation systems. Rev. Bras. Ciênc. Solo, 35, 819-832.

111. Comparison of the sustainability of different potato production systems: use of AgBalance methodology to identify strengths and weaknesses of organic, conventional and genetically modified disease-resistant potato cultivation

Martijn Gipmans[1,*], Jan Schöneboom[2], Brigitte Achatz[2], Thorsten Storck[3], Markus Prüfe[4], Manuel Knauer[4], Peter Saling[2]

[1] BASF Plant Science Company GmbH, c/o metanomics GmbH, Tegeler Weg 33, 10589, Berlin, Germany, [2] BASF SE, Sustainability Center, GU/NE – Z7, 67056 Ludwigshafen, Germany, [3] BASF Plant Science Company GmbH, GB/P, Agrarzentrum Li444, 67117 Limburgerhof, Germany, [4] BASF Plant Science Company GmbH, GBB/E, Agrarzentrum Li444, 67117 Limburgerhof, Germany, [*] Corresponding author. E-mail: *martijn.gipmans@basf.com*

AgBalance is a new LCA-based methodology to assess sustainability in the agricultural value chain in a comprehensive and holistic way. AgBalance is based on BASF's Eco-Efficiency and SEEBALANCE Analysis, to which, in consultation with international stakeholders and experts, new indicators specific for agriculture were added. AgBalance integrates over 200 data points in up to 70 indicators covering ecological, economical and social aspects of agriculture and received a validation of coherence and functionality from Det Norske Veritas (DNV), TÜV Süd as well as the National Sanitation Foundation (NSF) in the US.
Herein we present a case study using the AgBalance methodology, that investigated the factors determining the sustainability of different potato production systems in Germany. In general terms, potatoes are cultivated either using organic standards as defined by the EC Regulation 834/2007 or using conventional cultivation practices.
Potato late blight is a fungal disease caused by *Phytophthora infestans*, which accounts for annual losses (costs of control and damage) estimated at more than € 1,000,000,000 in the EU alone. BASF has developed a potato with a full and durable resistance against late blight through introduction of resistance genes from a wild potato by genetic modification (GM) technology into a modern European potato variety
We compared the sustainability of organic, conventional and genetically modified (GM) disease-resistant potato cultivation using the AgBalance method. Agronomic input data is taken from KTBL publications, the state office for agriculture in Lower-Saxony, federal ministry agriculture statistics and further public databases such as EUROSTAT. This data is representative of the 2007/08 growing season. Average yields are assumed to be 45 t/ha for the conventional and GM-, and 25 t/ha for the organic cultivation system.
Under conditions of moderate Phytophthora infestation, the aggregated sustainability impact score of the GM potato is similar to conventional varieties. The organic cultivation system receives a somewhat worse impact score, mainly due to the lower yield and the use of copper hydroxide in relatively high amounts. Applying scenario analysis, the effect of increased levels of infestation pressure is shown to result in significant environmental and economic benefits of the GMO potato, as conventional and organic production systems are affected by increased application rates of fungicides and yield losses. At the level of individual impact categories, there are marked strengths and weaknesses in each of the three alternatives. The GMO and conventional production are associated with fewer burdens through land use, acidification potential and emissions to water. Organic potato on the other hand is associated with less pressure on biodiversity in agricultural areas, less energy and resource consumption and global warming potential. On the economic side, there are no major differences in macro-economic indicators such as farm profits and subsidies, but production costs for organic potatoes are much higher. The differences in the social stakeholder categories are rather small, which is partly related to trade-offs between indicators that relate to benefits (positive implication, e.g. wages) as well as to burdens (negative implication, e.g., working accidents).
Taken together, AgBalance has proven to be a useful methodology to look at the sustainability of potato cultivation in a holistic way. AgBalance identifies the strengths and weaknesses of the different potato production systems and can deliver guidance for sustainable development in potato cultivation.

References

KTBL: Betriebsplanung Landwirtschaft 2006/7, 20. Auflage 2006. ISBN: 3-939371-07-6
Saling P., Kicherer A., Dittrich-Kraemer B., Wittlinger R., Zombik W., Schmidt I., Schrott W., Schmidt S., 2002. Eco-efficiency Analysis by BASF: The Method. Int. J. Life Cyle Assess. 7, 203-218.

Kölsch, D., Saling P., Kicherer A., Grosse-Sommer A., Schmidt I, 2008. How to measure social impacts? A socio-eco-efficiency analysis by the SEEBALANCE® method. Int. J. Sustain. Dev. 11, 1 – 23.

Schöneboom J.C., Saling P., van Gelder R. 2012. AgBalance – Holistic Assessment of Sustainability in Agriculture. Submitted to Int. J. Life Cycle Assess.

112. Modelling Estonian field crop farm types for LCA

Sirli Pehme*, Eve Veromann

Estonian University of Life Sciences, Estonia, * Corresponding author. E-mail: *Sirli.Pehme@emu.ee*

Food and beverage final household consumption in EU accounts for 22–34% of total life-cycle impacts in all environmental impact categories (except eutrophication that accounts 60%) (Tukker et al. 2006). Furthermore, different agricultural production practices have different impacts on environment. Thus, assessment through life cycle is needed to choose the best practises and mitigate pressure on environment.

Although, field crops have significant share in Estonian agricultural production, no agricultural LCA studies have been done in Estonia so far. There are huge variability in resource use and environmental impact between farms because of the variation in actual practises (Halberg 1999). Most of the LCA-s are based on data from case studies (Pfefferli and Gaillard 2000). To ensure the representativeness and to aim more general validity, assessment should be based on a larger sample of farm data. There are huge differences between countries in agricultural practices, therefore country-specific inventory data about farm practises is needed. In addition, environmental assessment through the life cycle taking account local conditions is necessary.

The aim of this paper was to create representative farm type models of Estonian arable crop farms based on different characteristics – main product type, management type, animal density (in case of mixed farming) – for further life cycle assessments. Different data sources were used to create inventory data of Estonian field crop farm models: systematic statistics, farm accountancy data networks, experiments, surveys, recommendations and expert opinion. Data of land area, energy and fertiliser use, yields and other characteristics are presented for each farm type. Challenges on creating farm type models are discussed.

References

Halberg, N., 1999. Indicators of resource use and environmental impact for use in a decision aid for Danish livestock farmers. Agriculture, Ecosystems and Environment 76 17-30.

Pfefferli S., Gaillard, G., 2000. Development of a new management tool by combining LCA and FADN. In: Weidema, BP., Meeusen MJG., (eds.), Agricultural data for life cycle assessments, Vol 2., Agricultural Economics Research Institute, The Hague, The Netherlands, 137-144.

Tukker, A., Huppes, G., Guinee J., Heijungs, R., de Koning, A., van Oers, L., Suh, S., Geerken, T., van Holderbeke, M., Jansen, B., Nielsen, B., 2006. Environmental impacts of products (EIPRO). European Science and Technology Observatory and Institute for Prospective Technological Studies, Seville, Spain. Available from: http://ec.europa.eu/environment/ipp/pdf/eipro_report.pdf

113. Developing innovative technical agronomical practices in sunflower cultivation for industrial applications: experience from central Italy

Francesco Razza[1,*], Francesco Degli Innocenti[2], Michele Falce[2], Luciano Concezzi[3], Andrea Massoli[3]

[1] Novamont, Piazz.le Donegani 4, 05100 Terni – Italy, [2] Novamont, Via G. Fauser 8, 28100 Novara - Italy, [3] 3A - Parco Tecnologico Agroalimentare dell'Umbria, Frazione Pantalla, 06059 Todi (Terni) – Italy, [*] Corresponding author. E-mail: *francesco.razza@novamont.com*

At worldwide level biomass is mostly exploited for producing materials rather than for energy use. Within the PSR Umbria project (Piano per lo Sviluppo Rurale - Cooperazione per lo sviluppo dei nuovi prodotti, processi e tecnologie nei settori agricolo e alimentare e in quello forestale) a biodegradable hydraulic lubricant was developed starting from sunflower oil with high oleic acid content. Such a lubricant, functionally equivalent to the fossil-based one, is characterised by a lower persistence in the environment since it is biodegradable. The PSR project was focused also on the development of innovative (low impact) technical agronomical practices for sunflower cultivation. Basically two high oleic sunflower cultivars (i.e. VarA and VarB) have been selected and tested with three different agronomical practices: "normal input" (i.e. traditional cultivation with a normal input of fertilisers and pesticides), "low input" (i.e. fertilisers were reduced by 50%) and "zero input" (i.e. no fertilisers applied but green manure before sunflower sawing). The lower input practices have been tested because the inputs to the agricultural phase (e.g. fertilisers, pesticides etc.) are generally higher than the actual needs of the crop as confirmed by experimental data reported in Table 1, where it is interesting to notice that the crop yield for the "low input" and "zero input" is equivalent or even higher compared to the "normal input". In reference to the "zero input", it is important to point out that, according to previous filed trials performed elsewhere, it was observed that green manure practice in sunflower cultivation, provides the right amount of nutrients (e.g. nitrogen) without generating a depletion of them,in the soil, rather they increase. Further the crop yield remain constant and similar to that generally obtained in a "normal input" practice also in the long run but with the advantage that the green manure practice protects the soil from erosion as well.

These practices have been assessed by LCA analysis where the functional unit was defined as the production of 1 kg of oleic acid. Also the oil cake (the co-product of mechanical oil extraction) uses (i.e. animal feed or energy utilisation) have been included in the analysis. LCIA results show that the highest benefits were reached in the "zero input" where the oil cake is used for energy purposes.

Basically it has been demonstrated that the right match between the sunflower cultivar, the geographic area, the integrated production of sunflower oil, contribute to reduce loads to the environment making the whole agricultural system particularly efficient.

References

Raschka, A., Carus, M.,2012. Industrial material use of biomass. Basic data for Germany, Europe and the world. Nova-Institute GmbH Chemiepark Knapsack, Industriestrasse, Hurth (Germany).

Laureti D., Pieri S., 1999. Fertilizzazione Biologica del Girasole. L'informatore Agrario, supplemento al n.10, 20-23. Istituto Sperimentale per Colture Industriali di Bologna – Sezione di Osimo (Ancona)

Laureti D., Pieri S., 2001. Sovescio di leguminose nella fertilizzazione del girasole. L'informatore Agrario, n.30. Istituto Sperimentale per Colture Industriali di Bologna – Sezione operativa periferica di Osimo (Ancona)

Table 1. Experimental data (2011) – Silveri farm

Agronomic pathway	**Sunflower cultivar**	**Yield (t/ha)**	**Moisture (%)**	**Dry matter DM (kg/kg fresh seeds)**	**Oil (% on DM)**	**Oil extraction yield (%)**	**Extracted oil (kg/kg fresh seeds)**	**Oleic acid content (%)**	**Sunflower seeds (kg fresh matter per kg of oleic acid**
NORMAL INPUT	VarA	2.3	4.8	0.952	51.2	80	0.390	89.0	2.9
	VarB	2.5	5.1	0.949	42.8	80	0.325	88.5	3.5
LOW IN-PUT	VarA	3.0	5.1	0.949	48.5	80	0.368	89.8	3.0
	VarB	2.8	6.2	0.938	39.1	80	0.293	88.4	3.8
ZERO INPUT	VarA	2.7	6.8	0.932	48.5	80	0.361	89.9	3.1
	VarB	2.2	6.4	0.936	45.1	80	0.338	88.5	3.4

114. Effects of different crop management on durum wheat GHGs emission

Salem Alhajj ali[1,*], Giuseppe De Mastro[1], Luigi Tedone[1], Stefano Bona[2], Lucia Coletto[2]

[1] University of Bari - Dipt of agricultural and environmental science, Via Amendola, 165/A - 70126, Italy, [2] University of Padova – Dipt. of agronomy, Viale dell'Uuniversità 16, 35020, Legnaro, PD, Italy, [*] Corresponding author. E-mail: *salem_grb@yahoo.com*

Having all the impacts, associated with different management typology of the agro-ecosystem, assessed through the use of LCA methodology according to an implementation of the data collected directly from the field to minimise data uncertainties, could be a very sensitive model which can support the current political action plans for reduction of the impacts which lead to the environmental effects. Moreover, the use of this kind of model to identify more sustainable combination of agricultural practices could satisfy the agronomists, economists as well as environmentalists once the productivity of the system is being maintained with as low inputs and emissions as possible.

For that, an on farm field trial in Southern Italy (Basilicata Region) was conducted to evaluate alternative agricultural practices that could help farmers to reduce reliance on fossil fuel, lower the input costs and decrease the GHG emissions of durum wheat production systems through life cycle assessment (LCA) method. The focus of this study was specially oriented to the potential GHGs emitted (expressed as CO_2-eq) as a consequence of the different levels of soil tillage (intensive (IT), reduced (RT) and conservative (CT) combined with different nitrogen fertilisation rates (90, 60, 30, 0 kg N ha^{-1} as urea). A special attention was given to grain yield as this represents the main farmers' objective and was being further correlated to the emissions.

The LCA analysis considered the entire system of the field experiment in which each treatment was replicated three times and the farm gate was considered as the system boundary. Inventory data of all agricultural operations in the field including tillage, seeding, fertilisation, herbicide application and harvesting, have been collected. Then, all data were used to estimate and compare (Through SimaPro 7.3 using the IPCC 2007 GWP 100a method for comparison) the impacts of different wheat production systems. Furthermore, in spit of all the data collected directly from the field, nitrogen balance was also calculated as the differences between nitrogen inputs and outputs of total nitrogen in the soil, grain, straw and plant residues sampled.

This study showed that there was a higher proportion of energy consumption and GHG emissions attributed to N fertiliser and to the ploughing operations for the production of wheat. The GHG emissions of different wheat production systems showed statistically significant differences within the treatments (tillage and N fertiliser rate), but no significant differences found in the interaction between tillage and N fertiliser applications.

Going deep into the analysis, we found that the highest CO_2-eq emissions was reported in the intensive tillage (IT) with 90 kg N ha^{-1} mainly because of the high emissions associated with the fertiliser and fuel production. The overall emissions were lower in conservation tillage (CT) and in reduced tillage (RT) systems compared by intensive tillage (IT) system due to the diesel fuel consumption which resulted from the high number of field operations.

On the basis of this first year of activity, conservation tillage (CT), represent a more environmentally-friendly system of wheat cultivation, which could sequester more CO_2, and as a consequence gave lower emissions and impact on the environment than both other systems. In contrast, intensive tillage (IT) was the worst in all scenarios.

115. Environmental and economic life cycle assessment of organic and conventional olive systems

Ramez Mohamad[1,*], Annalisa De Boni[2], Gianluigi Cardone[1], Marie Reine Bteich[1], Michele Moretti[2], Rocco Roma[2], Vincenzo Verrastro[1]

[1] Mediterranean Agronomic Institute of Bari, Italy, [2] Dep. Agro-environmental and territorial Sciences, Bari University, Italy, [*] Corresponding author. E-mail: *ramez_19777@yahoo.com*

The environmental awareness of the agricultural sector has been increasing during the last decades. Covering the environmental impacts of any agricultural product or service became a fundamental trend towards the optimal selection between different alternatives to improve the production. Organic farming has been reported to be an innovative system that contributes to the reduction of environmental impacts of agricultural practices. This study will investigate this assumption through a comparison of the environmental impact and the economic performance between two production systems of olive cultivation in Apulia region-Italy.
Based on a survey for farms selection, two olive farms have been selected for a case study of organic and conventional management systems. The criteria of farms selection was based on the similarity of general characteristics (location, olive variety, trees age, irrigated, planting system) and the dissimilarity of agricultural practices management, particularly fertilisation, soil management, pest and weed control.
LCA based methodology, adopting the Eco-indicator 99 method, has been used for assessing the environmental impact. Data collection has been analysed by SimaPro software considering 1 hectare as a functional unit with a system boundary limited to olive production (cradle to farm gate).
The olive life cycle was assumed to extend over 50 years and was divided into three phases: juvenility phase (4 years), growing phase (13 years) and productive phase (33 years). The environmental impacts were roughly similar during the juvenility and growing phases due to the likeness of the conventional managements in both case studies. Environmental results below are associated to the productive phase when the organic farm was certified organic and the conventional farm changed into no-tillage conventional system.
Fertilisation and soil management activities resulted in a higher environmental impact in the organic system compared to the conventional one, in terms of both single impact category and damage categories (damage to the human health, ecosystem quality and resources). This is due to the emissions induced by the transportation and application of animal manure as well as the higher fuel consumption for managing the soil in the organic system compared to no-tillage conventional one. Nevertheless, the total environmental impact of agricultural practices was lower in the organic system compared to the conventional one, mainly the lower impact on the fossil fuel depletion as a result of the more recurrent weed and pest control activities in conventional system. In fact, the total environmental impact caused by pest control activity was higher in the conventional system even if carcinogenic effects, ecotoxicity and minerals depletion impact categories were higher in the organic system, as a result of copper products uses for pest control.
LCC methodology has been used for assessing the economic performance of both systems by calculating all costs and revenues over the life cycle. No large differences were registered between the farms in terms of costs and revenues in juvenility and growing phases, while bulk differences were recorded in the productive phase. The organic system resulted in higher total costs and lower yield compared to the conventional one. However, it showed higher revenue and consequently higher net income thanks to the higher selling price. Both systems had a positive Net Present Value (NPV), showing a positive investment. Furthermore, the Internal Rate of Return (IRR) resulted, in both farms, higher than the bank interest rate (1.25%). The organic system resulted to have a higher NPV and IRR than the conventional one. Therefore, according to the farming price system and based on the profitability and financial analyses, the organic system can be considered a more profitable investment system than the conventional one.

116. A life cycle assessment of rice cultivation

Torsten Rehl[1,*], Barbara Dannenmann[2], Sabine Deimling[1], Joachim Müller[2]

[1] PE INTERNATIONAL AG, Leinfelden-Echterdingen, Germany, [2] University of Hohenheim, Institute of Agricultural Engineering (440), Stuttgart, Germany, [*] Corresponding author. E-mail: *t.rehl@pe-international.com*

Rice cultivation represented about 22% of the world´s grain crops area in 2008 (FAO 2010). It is cultivated across a vast area spanning wide ranges of temperate, subtropical and tropical climates. Various climatic and socio-economic conditions at different locations effect the rice cultivation and the environmental impacts going along with it. Rice cultivation however is a major source of green house gasses, especially methane (CH_4) (e.g. Sass 2005; Yan 2005). However, the alternation in wetting and drying to mitigate methane emissions can increase nitrous oxide (N_2O) emissions through enhanced nitrification and de-nitrification (Akiyama 2005a & b). Complex mechanisms impact not only global warming potential (GWP) but also acidification (AP), eutrophication (EP) and photochemical ozone creation (POCP). The objective of this study was to assess the environmental performance of different rice cultivation systems in various countries and climates.

Eight cropping systems have been chosen for data collection on precipitation, soil type, irrigation system, fertilisation, field operations, and pesticide use: conventional, irrigated, lowland rice cultivation in China (CC); deepwater rice cropping system in India (DWI); irrigated, conventional, lowland rice cultivation in India with rice straw incorporation (II) and with rice straw burning (IB); irrigated, conventional, lowland rice cultivation in the Philippines with two cropping cycles (IP2) and with three cropping cycles per year (IP3); irrigated, organic rice cropping system in Japan (OUJ) and a rain-fed, conventional, upland rice cultivation system in Japan (UJ). Rice systems were compared on a hectare (ha yr-1) as well as a product (kg ha-1) basis. The life cycle assessment followed ISO 14040. The inventory quantities fossil/renewable primary energy demand (PED) and water use (WU) were analysed. GWP, AP, EP and POCP were computed according to the CML method (Guinée 2002).

Upland rice systems (UJ and to some degree OUJ) showed lowest impacts among the rice-cropping systems tested in WU, GWP, AP and POCP and one of the lowest in PED and EP. OJ performed well in PED, GWP, POCP, AP and WU. On the other hand, the highest environmental impacts was caused by the systems IP2 and IP3, due to high inputs of fertiliser, diesel etc. along with low yields. The main emissions were released in the field by decomposition of nitrogen into NH_3, NO_{3-} and carbon into CH_4. Fertiliser production and diesel combustion in tractors and diesel generators of irrigation pumps were identified as additional sources of emissions. A removal of straw for bio-energy or burning on the field reduces nitrogen (i.e., N_2O, NO_x, NO_3^-) and carbon related emissions (CH_4). At the same time potential accumulation rate of soil organic carbon decreases and consequently increase PED and GWP due to the additional demand for mineral fertiliser. The main challenges were identified as the water regime and organic inputs, as well as the type of soil, the climate, the field management and the production and intensity of fertiliser use.

References

Akiyama, H., Yagi, K., Yan, X., 2005. Summary of N_2O emissions from rice paddy fields: Summary of available data. Global Biogeochemical Cycle 19: 1005.

FAO (2010). FAOSTAT database, Food and Agriculture Organization (FAO). Rome

Guinée, J. B., 2002. Handbook on LCA: Operational Guide to the ISO Standards. Netherlands

Watanabe, A., Kimura, M., 1999. Influence of chemical properties of soils on methane emission from rice paddies. Commum Soil Sci Plant Anal. 30: 2449-2463.

Yan, X., Yagi, K., Akiyama, H., Akimoto, H., 2005. Statistical analysis of the major variables controlling methane emission from rice fields. Global Change Biology 11 (7): 1131-1141.

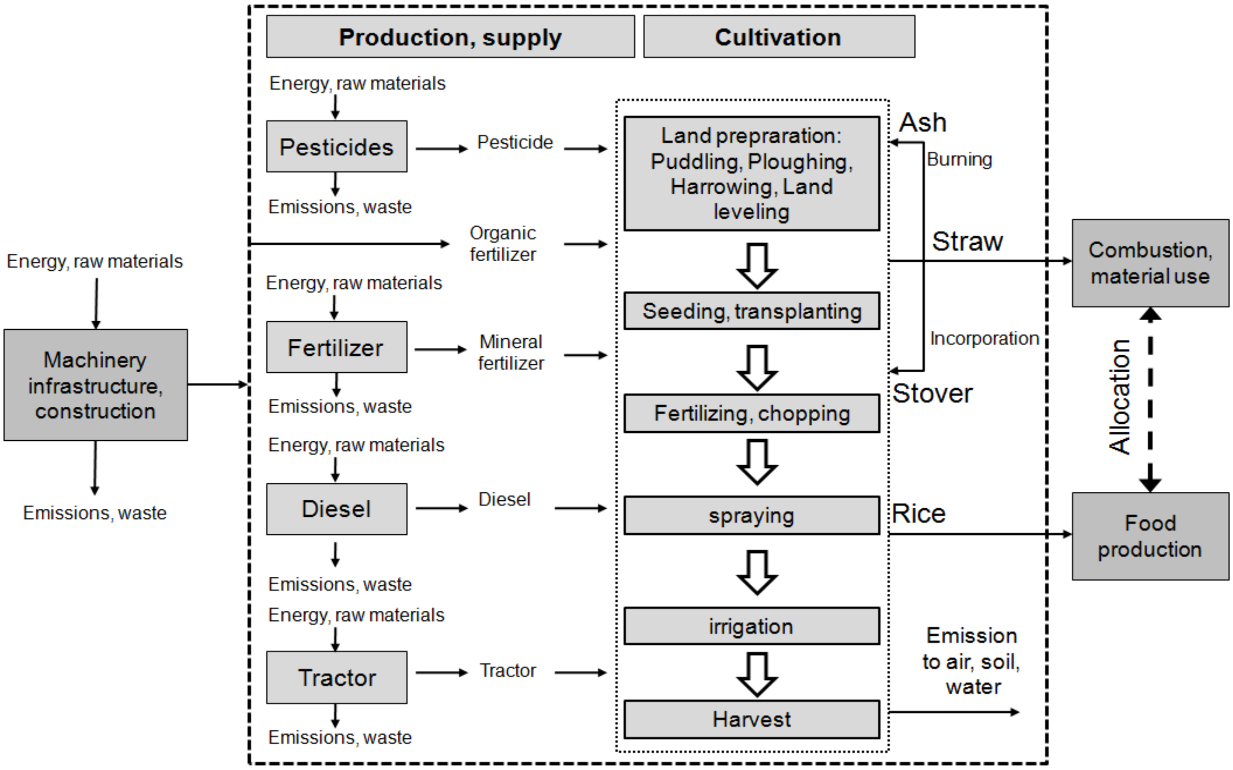


Figure 1. System boundaries of the LCA study applied for the various rice cropping systems.

117. Environmental assessment of rice production in Camargue, France

Ancia Drocourt[1], Yannick Mervant[1], Franc□ois Milhau[1], Mireille Chinal[2], Arnaud Hélias[1,3,*]
[1] Montpellier SupAgro, 2 Place Viala, 34060 Montpellier Cedex 2, France, [2] Sud Céréales, Mas des Saules RD 6113, 30300 Fourques, France, [3] INRA, UR50, LBE, Laboratoire de Biotechnologie et de l'Environnement, Avenue des Etangs, 11000 Narbonne, France, [*]Corresponding author. E-mail: *arnaud.helias@supagro.inra.fr*

In a context where environmental labelling tends to become certain, life cycle inventories of agricultural productions are key elements for the food product assessment. The herein study focuses on rice crops in the Camargue area (south of France), and the functional unit is "to produce one ton of rice".
The system is limited to the field operation, and does include transportation after harvest and recycling of agricultural machinery. The inventory is based on generic data available in the ecoinvent database. Cultural practices and crop yield from real data of Sud Céréales (an agricultural cooperative, main rice producer to Camargue) are used to describe the system. In this context, consistent with the regulation, crop residues are ploughed in. Models proposed in the literature are used to assess the emission from fertilisation, and methane release from rice fields is calculated using the IPCC model (IPCC 2006).
The life cycle impact assessment is carried out with the Impact 2002+ method (except for climate change where characterisation factors over 100 years are used). Results are put in front of American rice production available in the database. This comparison shows the environmental benefit of the Camargue rice (a decrease of 42% for greenhouse gas emissions) mainly due to more efficient submersion practices. Fertilisation (production and crop emission) is the most polluting step for all impacts; methane release is also a problem (Fig. 1). A sensitivity analysis underlines the lesser importance of agricultural machinery and the importance of the model chosen to assess the fertilisation. A decrease in yield would also cause greater damage to the environment; this is in accordance with a "mass based" functional unit.

References

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., (Eds). IGES, Japan.

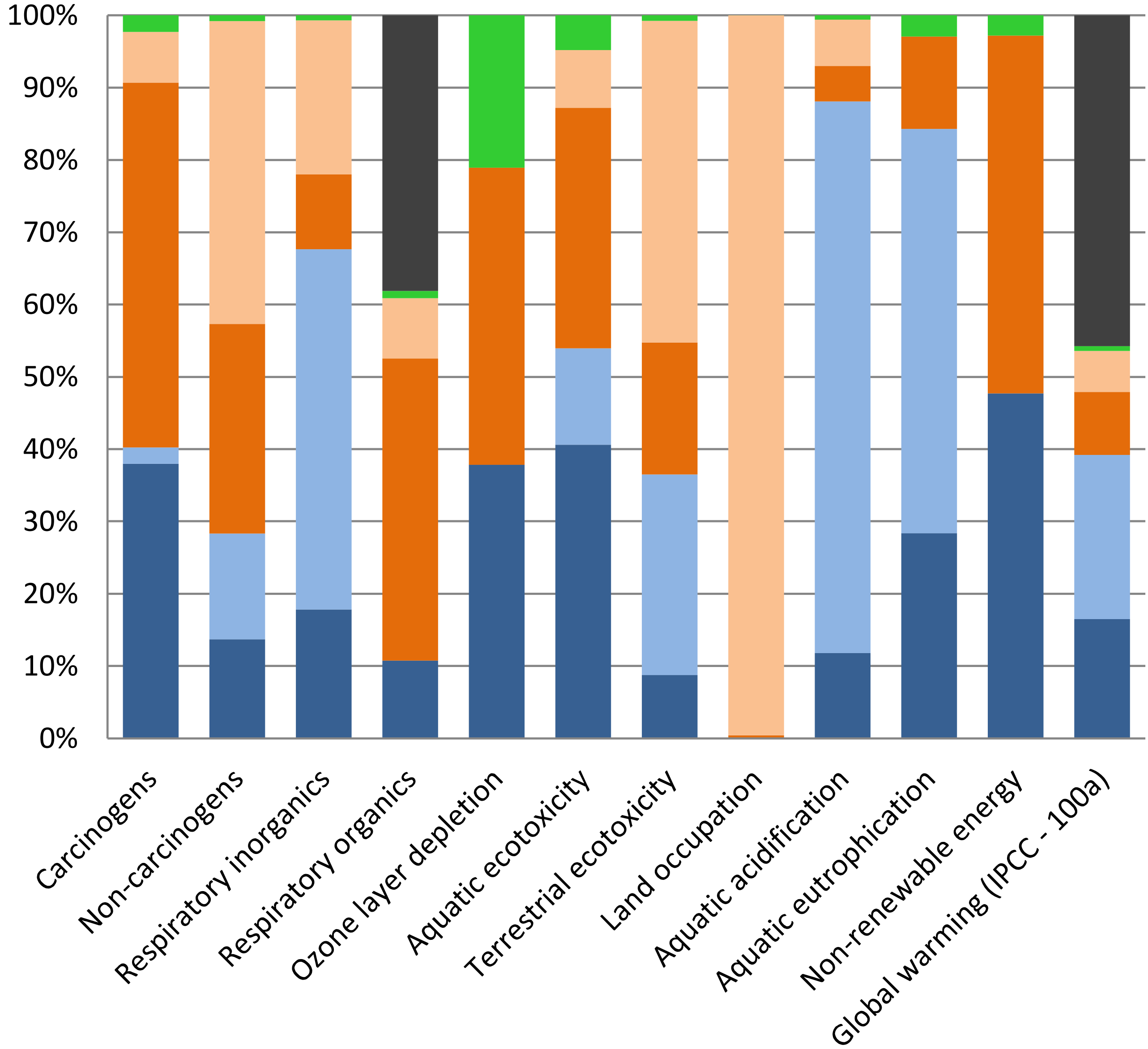


Figure 1. Process contribution for one tonne of rice in Camargue. Dark blue: fertiliser production, light blue: crop emission due to fertiliser, dark orange: machineries and fuel production, light orange: crop emissions due to agricultural acts, green: pest treatment consequences and grey: submersion practice.

118. Allocating environmental burdens from fertilisation to crops in a rotation

Alice Gueudet[1,*], Hervé Bossuat[1], Marilys Pradel[2], Afsaneh Lellahi[3], Aurélie Tailleur[3]

[1] ACTA, le réseau des instituts des filières animales et végétales, 149 rue de Bercy – Paris, F-75595, France, [2] Irstea, UR TSCF Technologies et systèmes d'information pour les agrosystèmes, Domaine des Palaquins – Montoldre, F-03150, France, [3] ARVALIS-Institut du végétal, Service Agronomie Economie Environnement, Boigneville, F-91720, France, [*] Corresponding author. E-mail: *alice.gueudet@acta.asso.fr*

Allocation is a complex issue for LCA practitioners. Due to many allocation methods, the choice between them can have a significant impact on the results. It is particularly relevant when applying LCA to assess fertilisation impacts as all nutrients brought by fertilisers are not all taken up by the crop in the system under study but remain available for future crops in rotation. Crop residues also produce nutrients which return to soil after harvest. Hence, these nutrients can be considered as co-products from the crop.

Fertilisation contributes highly to different potential impacts for crops: it induces in particular 30-60% of the primary energy consumption, 60-80% of the global warming potential, 90% of the eutrophication potential, 80-90% of the acidification potential (personal communication from ARVALIS and UNIP). Thus, allocation method used to allocate fertilisers to crops can be decisive for the LCA outcomes.

Allocating burdens from fertilisation is a subject of concern since the beginning of agricultural LCA and different reports dealt with this issue (Audsley et al. 2003; Gac et al., 2006). Recently, a working group (Agri Footprint Method, Blonk 2010) made recommendations to take into account mineral and organic nutrients from synthetic fertilisers, manure and crop residues. However, these references rarely lead to common accepted rules in term of methodology or their suggestions are not always easy to apply because of data availability.

In the framework of a project aiming at assessing environmental impacts of crop fertilisation by LCA, a bibliography review has been conducted to identify the different solutions involved in LCA literature regarding allocation rules for PK nutrients in synthetic fertilisers, N from manure and N from crop residues (Table 1).

Considering this large range of choices, allocation rules have been selected in order to be tested on study cases which will simulate levers to reduce impacts from fertilisation such as cover crops introduction and the use of different types of fertilisers (mineral and organic). These simulations will assess the rules feasibility and identify the most consistent. This work will result in recommendations about cropping plan allocation for mineral and organic N, P, K nutrients and for crop residues, regarding the aim of the study and the data availability.

References

Audsley et al. 2003, Harmonisation of environmental life cycle assessment for agriculture, final report DG VI Agriculture 2003, 101p.

Blonk et al. 2010, Methodology for assessing carbon footprints of horticultural products - A study of methodological issues and solutions for the development of the Dutch carbon footprint protocol for horticultural products Blonk Milieu Advies p.36-40

European Commission 2010, ILCD Handbook, General Guide for Life Cycle Assessment – detailed guidance, 394p.

Gac et al. 2006, GES’TIM Guide méthodologique pour l’estimation des impacts des activités agricoles sur l’effet de serre, Version 1.2, 156p.

William et al. 2006, Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Defra project report IS0205, 97p.

Table 1. Allocation rules proposed in the literature for PK nutrients in synthetic fertilisers, N from manure and N from crop residues

Reference	Allocating PK	Allocating N from manure	Allocating N from crop residues
ILCD Handbook 2011	Attributional or system extension recommended	Attributional or system extension recommended	Attributional or system extension recommended
Blonk et al. 2010	To each crop in rotation according to recommended quantities	Slowly degradable N fraction equally to the crops in rotation, quickly degradable N fraction to the current crop	According to surface areas
Gac et al. 2006	To each crop in rotation according to the requirements	According to degradation dynamic or equally between the crops in rotation (between two applications)	-
Williams et al. 2006	To the current crop except if it is an exigent crop (ex. Potatoes). In that case: surplus P to each crop according to requirements	To the current crop	-
Audsley et al. 2003	To each crop in rotation according to the recommended quantities	To each crop in rotation according to the recommended quantities	-

119. Designing sustainable crop rotations using life cycle assessment of crop sequences

Frank Hayer[1], Emmanuelle Bonnin[2], Benoit Carrouée[3], Gérard Gaillard[1], Thomas Nemecek[1,*], Anne Schneider[3], and Christophe Vivier[4]

[1]Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zurich, Switzerland, [2] Chambre d'Agriculture de la Nièvre, F-58028 Nevers, France, [3] Union Nationale des Plantes Riches en Protéines UNIP, F-75008 Paris, France, [4] Chambre d'Agriculture de l'Yonne, F-89015 Auxerre, France, [*] Corresponding author. E-mail: *thomas.nemecek@art.admin.ch*

The project CAS DAR PCB (Amélioration des performances économiques et environnementales de systèmes de culture avec Pois, Colza et Blé) aimed at analysing and optimising crop rotations in three French regions by the selection of crops in a crop rotation, defining their sequence and by different nitrogen fertilisation levels. An LCA study of a large number of crop rotations is very time consuming. Therefore the different sequences of previous crop-main crop were defined and analysed by LCA, considering the effects of a specific previous crop on cultivation, fertilisation, yield and emissions of the main crop. These crop sequences were subsequently combined to analyse 58 crop rotations. Two functional units were used in this analysis: hectare per year and € gross margin II. The study focused on the effect of legumes and reduced N fertilisation on environmental impacts.
Production data were collected by the Chambers of Agriculture for a typical cultivation in Burgundy, Beauce, and Moselle for the reference period 2002-2008. The yield data were taken from survey of field level by the Chambers of Agriculture. Background data describing infrastructure, inputs and processes stemmed from the ecoinvent database version 2.01 (ecoinvent Centre, 2007). The direct field emissions (NH_3, N_2O, P, NO^{3-}, heavy metals and pesticides) were estimated by models described in the SALCA method (Nemecek et al., 2010). The analysis included the production from cultivation up to the delivery at farm gate, as well as the environmental impacts linked with input factors and the direct field emissions. The selected rotations are given in Table 1. The gross margin II was calculated based on mean prices in the reference period.
To illustrate the results the global warming potential (GWP) is shown for selection crop rotations in Fig. 1. Following Nemecek et al. (2008) a difference of 4% between two crop rotations can be considered as significant. The same tendencies were found across all impact categories. The alternative crop rotations with pea (P1 to P3) consistently reduced the GWP as compared to the standard rotations without pea (S1 and S2), both per hectare and year and per € gross margin II. The effect was similar whether barley in the standard rotation was replaced by pea (P1) or pea was added (P2 and P3). The global warming potential per ha and year was reduced by around 10% and per € gross margin by around 12%. In P2 pea is inserted before the stubble wheat in rotation S2. This reduced the GWP by around 14% per ha and year and 19% per € gross margin II compared to S2. Looking at the second option, the reduced fertilisation, comparing the conventional and integrated rotations in Beauce shows that the impact per ha is reduced, whereas the impact per € gross margin II remains constant due to lower revenues. Combining introduction of pea with reduced N fertilisation (Beauce_Int P1-P3) seems to be the most effective way to reduce the GWP.
The analysis illustrates that peas allow to decrease impacts per ha and year on a rotational level and to increase the eco-efficiency (lower impacts per € gross margin II). This is caused by positive rotational effects (e.g. higher yields or a lower fertilisation in succeeding crops). Therefore this strategy is favourable compared to a reduced fertilisation in single crops. Combining both measures is the most effective strategy when looking at impacts, but on the other hand the gross margin II is reduced by around 40 € compared to the standard rotations.

References

ecoinvent Centre (2007): ecoinvent Data - The Life Cycle Inventory Data. Swiss Centre for Life Cycle Inventories, Dübendorf, ISBN 3-905594-38-2 available at http://www.ecoinvent.org.
Nemecek, T., Freiermuth Knuchel, R., Alig, M. & Gaillard, G. (2010): The advantages of generic LCA tools for agriculture: examples SALCAcrop and SALCAfarm. In: 7th Int. Conf. on LCA in the Agri-Food Sector, Notarnicola, B. (eds.). Bari, Italy. 433-438.
Nemecek, T., von Richthofen, J.-S., Dubois, G., Casta, P., Charles, R. & Pahl, H. (2008): Environmental impacts of introducing grain legumes into European crop rotations. European Journal of Agronomy, 28: 380-393.

Table 1. Crop rotations analysed in the regions Beauce, Burgundy and Moselle. S1 and S2 are the standard rotations without pea, P1 to P3 are the rotations with pea.

		Rotation				
		S1	S2	P1	P2	P3
Beauce Int/Con	1	Oilseed rape				
	2	Winter wheat				
	3	Malting barley	Winter wheat	Winter pea		Malting barley
	4	Oilseed rape	Malting barley	Oilseed rape	Winter wheat	Winter pea
	5	Winter wheat		Winter wheat	Malting barley	Oilseed rape
	6	Malting barley		Malting barley		Winter wheat
	7					Malting barley
Burgundy/Moselle	1	Oilseed rape				
	2	Winter wheat				
	3	Malting barley	Winter wheat	Spring pea		Malting barley
	4	Oilseed rape	Ma./Fe barley	Oilseed rape	Winter wheat	Spring pea
	5	Winter wheat		Winter wheat	Ma./Fe barley	Oilseed rape
	6			Ma./Fe barley		Winter wheat
	7					Ma./Fe barley

ma. = malting in Beauce and Bourgogne ; fe. = feed en Moselle

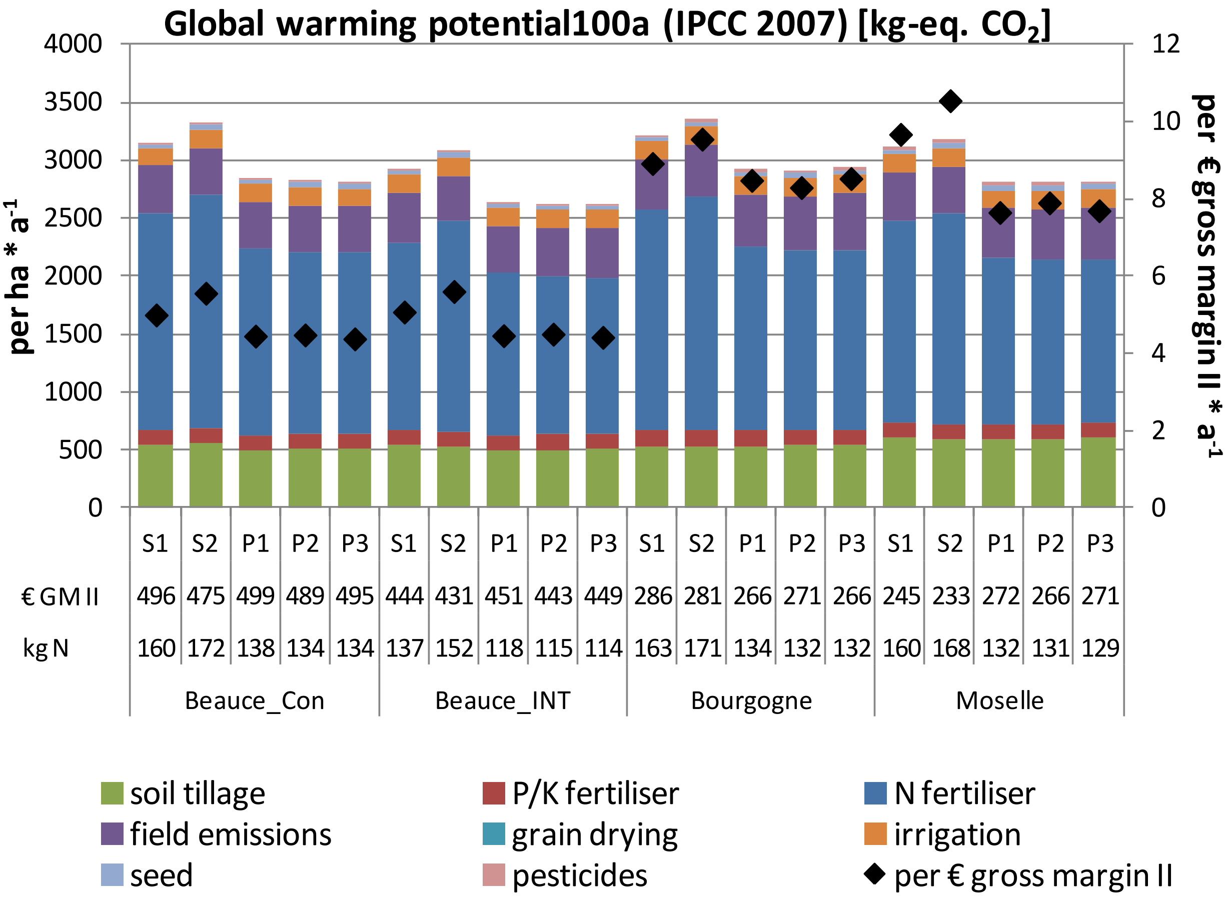


Figure 1. Global warming potential of crop rotations analysed in Beauce, Burgundy and Moselle in kg CO_2-equivalents per ha*a^{-1} (indicated by the columns) and per € gross margin II (diamond symbols). Beauce_CON = conventional production in Beauce, Beauce_INT = integrated production with reduced nitrogen fertilisation.

120. Life cycle assessment of various fertilisation systems in open field vegetable production, Flanders, Belgium

Reindert F. Heuts*, Jelle Van Loon, Eddie Schrevens

Katholieke Universiteit Leuven, Research group DQM, Div. M3-BIORES, Dep. of Biosystems, Celestijnenlaan 200E b2411, BE-3001 Heverlee, Belgium, * Corresponding author. E-mail: *Reindert.Heuts@biw.kuleuven.be*

In Flanders, many farmers and horticultural growers wonder how they can lower the residual nitrate on their parcels to satisfy the fertiliser regulations set up by European legislation. Currently this threshold is set at a residual soil nitrate value of 90 kg/ha; for many farmers a limit hard to comply. In open field vegetable production like leek and cauliflower an overuse of fertiliser is very common. In this sector, less then 40% of the cultivated parcels were found under the imposed limit (Vlaamse Milieumaatschappij (VMM) 2006). One of the reasons of this excess on fertiliser application is the need of farmers to procure a good yield with a high quality to be competitive on the domestic market. Lowering the nitrogen application rate however, with current recommended fertiliser schemes (i.e. KNS-system (Lorenz et al. 1985)), does not yet give sufficient guaranties to uphold these demanding standards.

In 2010 an experiment has been set up to monitor and evaluate the influence of different fertiliser application rates and strategies on the growth, yield and quality of a cauliflower crop, and with a special emphasis on the amount of nitrate leaching to soil and surface water. The experiment consists of 8 plots, treated with two different fertiliser doses and two application strategies, to create a two by two completely randomised factorial design, replicated in two blocks. Each plot was fitted with an impermeable foil to capture and sample the drainage. Destructive and non-destructive plant samples were taken to follow up growth evolution of the plant and at harvest measurements were done to evaluate the overall product quality.

The experiment was chosen in such a way that a comparison could be made between common cultivation practices and ‘improved’ management schemes developed with the intention of lowering the nitrate leaching. For the common practices a broadcast fertilisation with calcium ammonium nitrate is assumed and, based on the recommendation of the KNS-system, a split fertilisation of 150 kg N/ha at planting and 240 kg N/ha 7 weeks later is applied (*i.e.* the high dose), the mineral N present in the soil profile included. In attempt to achieve lower nitrate leaching, the improved strategies consist of: a) the same broadcast split fertiliser application but with a low dose of 50 kg N/ha at planting and 100 kg N/ha 7 weeks into the growing season, b) a weekly fertigation of ammonium nitrate with the high dose, and c) the same fertigation scheme with the low dose.

As expected, differences in yield have been found among the various tested fertilisation systems (*i.e.* broadcast high yielded 70,99 ton/ha, broadcast low 55.26 ton/ha, fertigation high 67.28 ton/ha and fertigation low 53.44 ton/ha). When aiming only at high productivity, one would not distinguish between the broadcast and fertigation application of fertiliser, but with current pressure on the environment more objectives come into consideration. For this reason a life cycle assessment (LCA) has been carried out (Guinee et al. 2002). Comparing each fertilisation system the LCA focuses on the differences in input loads and their corresponding environmental impacts in terms of resource depletion, global warming, toxicity, acidification and eutrophication with special attention to nitrate leaching. This way policy makers and farmers can consider to what extent the quantity or quality is influenced by a reduction in fertilisers and/or different treatment inputs regarding their impact on the environment. In this way, the LCA supports decision making regarding optimisation of the fertilisation system and adaptation to severe constraints about energy demand and emissions to air, water and soil.

In contrast with expectations, no large differences in impact have been found and even more surprising, the fertigation treatments did not score all too well. The nature of environmental impact with respect to the used fertilisation system, however, did change. This suggest that, even though a progressive split fertiliser application and fertigation are perceived as environmental conserving techniques, the problem is more complex and the larger context in which the farming activity takes place has to be considered in first instance before being able to make proper unambiguous fertiliser management recommendations.

References

Guinee, J. B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. A., de Bruijn, H., van Duin, R., Huijbregts, M. A. J., 2002.

Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background. Kluwer Academic Publishers, 692

Lorenz, H., Schlaghecken, J., Engl, G., 1985. Gezielte stickstoffversorgung - Das Kulturbegeleitende Nmin-Sollwerte System., 13, 646 - 648

Vlaamse Milieumaatschappij (VMM), 2006. MIRA-T 2006 Milieuindicatoren in zakformaat.

121. Comparison of CO_2 emissions in the wine production from traditional and organic farming techniques. A Spanish case study

A. Aranda*, S. Scarpellini, G. Ferreira, D. Zambrana

CIRCE - Centre of Research for Energy Resources and Consumption, University of Zaragoza, Zaragoza, Spain, * Corresponding author. E-mail: *alaranda@unizar.es*

Over the last several decades there is a growing demand for products made in a natural way with less intervention of chemicals. However, the concern arising from the generation of greenhouse gas (GHG) emissions from the different production systems entails an assessment of their energy consumption, as well as of their emissions.

Nowadays, the traditional crop has a number of technological elements added by the man which has been experiencing and acquiring more knowledge about the process over the past decades. In this sense, the traditional crop is defined as an industrialised method that currently is used by the largest systems of production of grape. However, the organic farming which differs from the traditional crop mainly because it avoids the use of agrochemicals such as fertilisers and non-organic plant protection products, is gaining a high interest in the last decade.

This paper shows the results obtained by the estimation of CO_2 equivalent emissions using SimaPro v.7.2 and ecoinvent 2.1 (PRé 2007) database, in order to assess the environmental impact of the traditional and ecological winegrowing. Although organic farming has as main objective the obtaining of maximum quality food while respecting the environment, preserving the fertility of the earth through the optimal use of resources and without the use of synthetic chemical products; the results show that this technique has associated higher amount of CO_2 equivalent emissions i.e. 22% more compared to the traditional crop.

122. Grape life cycle assessment: which methodological choices for a combined assessment with grape quality?

Christel Renaud-Gentié[1,*], Frédérique Jourjon[1], Marc Benoît[2]

[1]PRES LUNAM,UMT VINITERA, ESA, UR GRAPPE, 55 rue Rabelais, BP 30748, 49007 Angers Cedex 01 – France, [2]INRA, SAD, UR 055, Aster, 662 avenue Louis Buffet Mirecourt BP 35, 88501 Mirecourt, France, [*] Corresponding author. E-mail: *c.renaud@groupe-esa.com*

Social, market and institutional pressure is growing on wine producers for more environment friendly production strategies, while high quality wine are more than ever necessary in a very competitive globalised market. Identification of the best trade-offs between environmental performance of the vineyard management and quality of the wines would help producers in adapting their technical choices to these requirements. Consequently, we are building a combined method assessing grape production practices on qualitative and environmental aspects through the combination of LCA and quality evaluation of the grapes.
Within the LCA results published about wine production (Petti et al.,2010;Vázquez-Rowe et al.,2012), we have to improve the detailed analysis of the agricultural practices contributions in a wide diversity of technical management paths and taking into account the entire life of the vineyard.
Grape LCA is calculated in our project for existing contrasted vineyard management strategies in parallel of quality measurements on grapes. The aim of the project is to identify (i) if trade-offs are needed between quality and environment and (ii) for which specific part of production process (Renaud et al.,2010). This poster presents the methodological choices done for LCA application to viticulture in this purpose. It exposes examples of the choices made.
The main types of vineyard management strategies of Chenin Blanc Grape production for dry PDO wines in the Middle Loire Valley region, France were determined through typology methods applied on a detailed survey conducted on 77 diverse parcels. Five main types of vineyard management paths were identified resulting to the choice of 5 representative winegrowers plots used for collection of grapes and data for LCA.
The key methodological questions about LCA that were answered concern all the steps of LCA process: (i) Goal and scope definition: the most suitable functional unit for grape production is not only a grape quantity but also a production surface unit because of the importance of yield in wine quality elaboration (Renaud et al., 2010).
(ii) System boundaries: the entire life of the vineyard is taken in the system, including pre and post productive phases of the vineyard management, like vine planting, nursery, uplifting of the vineyard.
(iii) Data collection from the winegrowers and suppliers: all data have been detailed (machines, buildings, infrastructure, operations, fertilisers, pesticides, fuels, working time, transport) but some have in the future to be more lightly informed. A data collection tool was built for LCI implementation and direct emissions calculation.
(iv) Inventory: direct emissions calculation models suitable for viticulture have been identified and compared on literature basis for their choice (see an example in table 1) for pesticides, erosion, nitrogen, phosphorus, and heavy metals. They are applied for calculation.
(v) Impacts calculation: processes which need to be detailed and calculated specifically for viticulture are identified and relevant impacts categories for viticulture are chosen.
(vi) Interpretation: contributions analysis, sensitivity analysis and comparisons with literature give information on which processes need to be more detailed than others through the iterative process of LCA.
This poster is a contribution for LCA practitioners who want to deal with viticulture or more widely perennial plants. It proposes solutions about the main methodological choices to be made on the agricultural part of wine production. This study gives an assessment based on detailed data collected on real contrasted vineyard management systems in order to identify practices contributions for linking them to grapes quality results. It needs to be confirmed by further iterative calculations and two more vintages of observations.

References

Audsley E, Alber S, Clift R, Cowell S, Crettaz P, Gaillard G, et al. Harmonisation of environmental life cycle assessment for agriculture, Final Report Concerted Action AIR3-CT94-2028. In: Commission E, editor. Community Research and Technological Development Programme in the field of "Agriculture and Agro-Industry, including Fisheries" AIR 3: European Commission DG VI Agriculture; 2003. p. 129.
Birkved M, Hauschild MZ. PestLCI--A model for estimating field emissions of pesticides in agricultural LCA. Ecol. Modell. 2006;198(3-4):433-51.

Jolliet O, Crettaz P. Fate coefficients for the toxicity assessment of air pollutants. Int. Journal of Life Cycle Assessment. 1997;2(2):104-10.

Margni M, Rossier D, Crettaz P, Jolliet O. Life cycle impact assessment of pesticides on human health and ecosystems. Agriculture, Ecosystems & Environment. 2002;93(1-3):379-92.

Petti L, Ardente F, Bosco S, De Camillis C, Masotti P, Pattara C, et al. State of the art of Life cycle Assessment (LCA) in the wine industry. In: Notarnicola B, Settani E, Tassielli G, Giungato P, editors. LCAFood 2010. Bari (IT),: Universita degli studi di Bari; 2010. p. 493-8.

Renaud C, Benoît M, Jourjon F. Trade-offs between quality and environment in wine production: presentation of a research program for their combined assessment. In: Notarnicola B, Settani E, Tassielli G, Giungato P, eds. LCA Food 2010. Bari, (IT): 2010. p. 241-6

Thiollet M. Construction des indicateurs viti-environnementaux de la méthode INDIGO. Colmar: INRA; 2003.

Vázquez-Rowe I, Villanueva-Rey P, Moreira MT, Feijoo G. Environmental analysis of Ribeiro wine from a timeline perspective: Harvest year matters when reporting environmental impacts. Journal of Environmental Management. 2012;98(0):73-83.

Webb J, Hutchings N, Amon B. Agriculture other: Use of pesticides and limestone. EMEP/EEA emission inventory guidebook European Environmental Agency; 2009.

Table 1. Comparison of five pesticide emission models and indicators of use in LCA of wine-grape production.

	Data aspects considered											
Model	Volatilisation to air	Wind drift	Emissions to surface waters	Leaching to surface waters	Mode of application	Active substance properties	Agricultural practices	Crop species	Crop phenological stages	Application scale	Adapted to grapevine (Yes / No)	Quantitative (Yes / No)
(Audsley et al., 2003)	●		●	●		●				national / regional	no	yes
EMEP / CORINAIR (Webb et al., 2009)	●	●								national / regional	no	yes
PestLCI (Birkved et al., 2006)	●	●	●	●	●	●	●	●	●	plot	partly	yes
I-Phy (INDIGO) (Thiollet, 2003)	●	●	●	●	●	●	●	●	●	plot	yes	no
CST (Jolliet et al.,1997), (Margni et al., 2002)	●		●	●		●				national / regional/ plot	no	yes

Acknowledgements: A. Alaphilippe, E Bezuidenhoud, D. Boudiaf, G. Gaillard, P. Koch, and H. Van der Werf for their contribution to the work.

123. Ecodesign opportunities for a farmer's bread. Two case studies from north-western France

Michal Kulak*, Thomas Nemecek[1], Emmanuel Frossard[2], Gerard Gaillard[1]

[1] Agroscope Reckenholz-Tänikon Research Station ART, Zurich, Switzerland, [2] Institute of Agricultural Sciences, ETH Zurich, Switzerland, * Corresponding author. E-mail: *michal.kulak@art.admin.ch*

Bread is a staple food item for Europeans. Its history can be traced back to the beginning of our civilisation. However, due to industrialisation and urbanisation, its production processes have notably changed over time. Some consumers believe that organoleptic and nutritional qualities of the product were affected as a result of these changes. There are farmers in Brittany and Pays de la Loire regions, who reintroduce many of the forgotten bread production methods from pre-industrial era. They collect ancient varieties of cereals and cultivate them in an organic way. Some use horses for traction. The grain is milled on-farm and the bread is baked and sold locally. This way, a unique product is created. Sometimes though, this is done at the expense of the environment as traditional methods are not necessarily more ecoefficient. Van Holderbeke et al. (2004) compared environmental impacts from bread production in Belgium in the year 1800, 1900 and 2000. The life cycle carbon footprint results were 1.2 kg CO_2eq kg^{-1}, 1.1 kg CO_2eq kg^{-1} and 0.6 kg CO_2eq kg^{-1} respectively. The goal of this study was to measure environmental impacts of French farmer's bread and explore opportunities for changes in the design of production and distribution processes that would allow minimising environmental impacts while maintaining the positive attributes of this distinctive product to the consumer.

Data on farming practices, processing and distribution of bread were collected from two producers in north-western France. Recent version of Swiss Agricultural Life Cycle Assessment (Nemecek et al., 2008) tools and ecoinvent database were used to assess environmental impacts from the field to the consumer's table. The functional unit was 1 kg of bread delivered at home and ready for consumption. End-of life processes-human excretion and wastewater treatment were excluded from the analysis. Impact categories were selected to reflect a broad range of environmental effects, including global warming contribution, the use of natural resources and potential toxicity. Results of the studies were disseminated to the farmers. Semi-structured face-to-face interviews were conducted to choose promising ecodesign strategies- ones that would be effective in reducing environmental impacts and also accepted by the producer and his consumers.

Table 1 shows a comparison of LCA results expressed per 1 kg of bread. Factor 8 differences in total result exist between the two farms for some impact categories. This suggests significant differences in ecoefficiency may be achieved with different production methods. Fig. 1 shows the contribution of the particular production stages into the overall environmental impact in Case 1. Most environmental impacts come from the wheat cultivation, followed by distribution and baking. There are strategies that can improve the ecoefficiency and would be accepted by the producer. The first solution would be to expand the relative area with the cereals and use mechanical traction, instead of using the land to produce feedstuff for horses. At the same time, wheat variety currently cultivated by the farmer provides relatively low yields in the given soil conditions. It is expected, that choosing a variety better adapted to local conditions would improve the product environmental performance. It may also be possible to change the proportion of flour in the bread recipe. A higher proportion of crops that grow better than wheat, such as rye could be used. It may also be feasible to optimise baking and distribution processes. Fig. 2 shows results for the second producer. Changing the crop or variety can also be considered here. A large share of the impact comes from the baking process. This is mainly done in the oven at consumer's home. Baking the bread on-farm in a more efficient oven or forming a partnership with the baker could potentially add value to the sold product and at the same time reduce environmental impacts.

References

Nemecek, T., von Richthofen, J., S., Dubois, G., Casta, P., Charles, R, Pahl, H. 2008. Environmental impacts of introducing grain legumes into European crop rotations. Eur. J. Agron. 28, 380-393.

Van Holderbeke, M., Sanjuán, N., Geerken, T., De Vooght, D. 2004. The history of bread production. Using LCA in the past, in: Halberg, N. (Ed.), Life Cycle Assessment in the agri-food sector. Danish Institute of Agricultural Sciences, pp. 255-260

Table 1. Selected environmental impacts from 1 kg farmer's bread at the consumer's table of two case study farms in North-Western France.

Impact categories		Unit	Total impact (per 1 kg bread)
Non-renewable resource use - fossil and nuclear	Case 1	MJ eq	23.8
	Case 2	MJ eq	14.3
Global Warming Potential	Case 1	kg CO_2 eq	1.90
	Case 2	kg CO_2 eq	0.61
Eutrophication potential (terr., global)	Case 1	m^2	0.53
	Case 2	m^2	0.06
Aquatic ecotoxicity	Case 1	kg 1,4-DB eq	2.8×10^{-4}
	Case 2	kg 1,4-DB eq	0.2×10^{-4}
Land competition	Case 1	m^2	13.96
	Case 2	m^2	4.58
Total water use (blue water)	Case 1	dm^3	11.60
	Case 2	dm^3	5.84

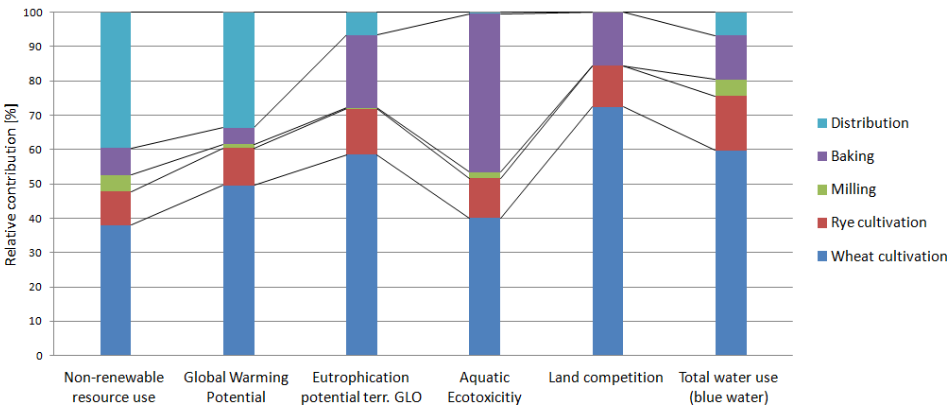


Figure 1. Case 1 - contribution of production stages into the environmental impacts of farmer's bread.

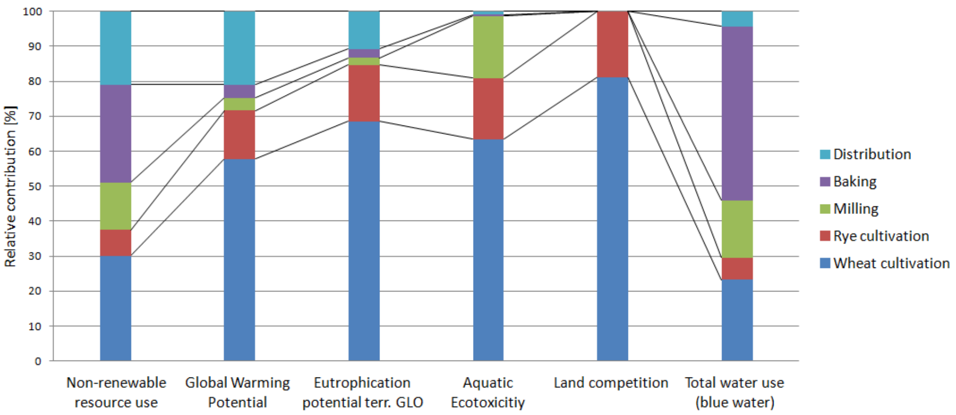


Figure 2. Case 2 - contribution of production stages into the environmental impacts of farmer's bread.

124. How to adapt generic LCI data to be reliant on the original recipe: case study of a biscuit

Roberto Bellino[1], Marie Vuaillat[1], Stéphane Le Pochat[1,*], Laure Vidal[2], Stéphanie Capdeville[2], Sylvie Martinel[3]

[1] EVEA, 56 bd. de la Fraternité, 44100 Nantes, France, [2] Eco-Concevoir, 23 Allée des Marronniers, 34130 Mauguio, France, [3] Nutrition & Santé, BP 106, 31250 Revel, France, [*] Corresponding author. E-mail: *s.lepochat@evea-conseil.com*

In 2011, EVEA and Eco-Concevoir carried out an LCA study of a biscuit for NUTRITION & SANTÉ, a leading company in Europe for health and organic agrifood products. It engaged in a global ecodesign process, carrying out several LCAs, of which one was the LCA of a biscuit.
Input data for LCIA were the different ingredients of the recipes and LCA practitioners had to deal with the lack of data or generic or inconsistent data from ecoinvent and LCA about specific ingredients or processes. To make LCIA results consistent as far as possible we had to adapt data from ecoinvent. For agricultural data, we developed a procedure to create the required data (Fig. 1, Table 1). Once the inventory was performed, we conducted the life cycle impact assessment of the biscuit. We compared results of the LCA of the biscuit with generic data and adapted data.
Adapting data for cultural processes in our case study increased the impacts by 10-75% on 7 impact categories (Fig. 2). If we assume an average uncertainty of 30% on each indicator, these increases are significant on 3 indicators: water consumption, eutrophication and acidification. These changes are mainly due to adaptation of yield which is lower in organic cultures than in conventional ones and introduction of irrigation for the specific country where sugarcane is produced. We can also note that the adaptation leads to a decrease of 22% of the impact of the product on water ecotoxicity. It can be easily explained by the adaptation of pesticides use that was removed for organic cultures. However, this change cannot be considered as significant regarding the high level of uncertainty on this indicator.
It is interesting to note that even in a cradle to gate system (including packaging fabrication and end-of-life, biscuit production, distribution, etc.) the parameters related to agricultural step can have significant influence on the final results. In the case of generic agricultural data adaptation, LCA practitioners should pay attention with priority to main parameters (here, yield, irrigation and pesticides) and main ingredients.

References

Gopinath, K.A., 2011. Organic Sesame Production. Technical Bulletin, Central Research Institute for Dryland Agriculture, Santoshnagar, Hyderabad, Andhra Pradesh, India. 34p.
Gudoshnikov Organic Sugar, 2001. Niche Commodity In The Mainstream Market, FO LICHT International Sugar and Sweetener Report, Vol 133, No 22 of 24th July 2001.
Nemecek, T., Kägi, T. Life cycle inventories of agricultural production systems. Data v2.0 (2007), ecoinvent report n°15, December 2007.
Toolseeram Ramjeawon, 2004. Life Cycle Assessment of Cane-Sugar on the Island of Mauritius, Int J. of LCA, 9 (4), pp. 254-260.

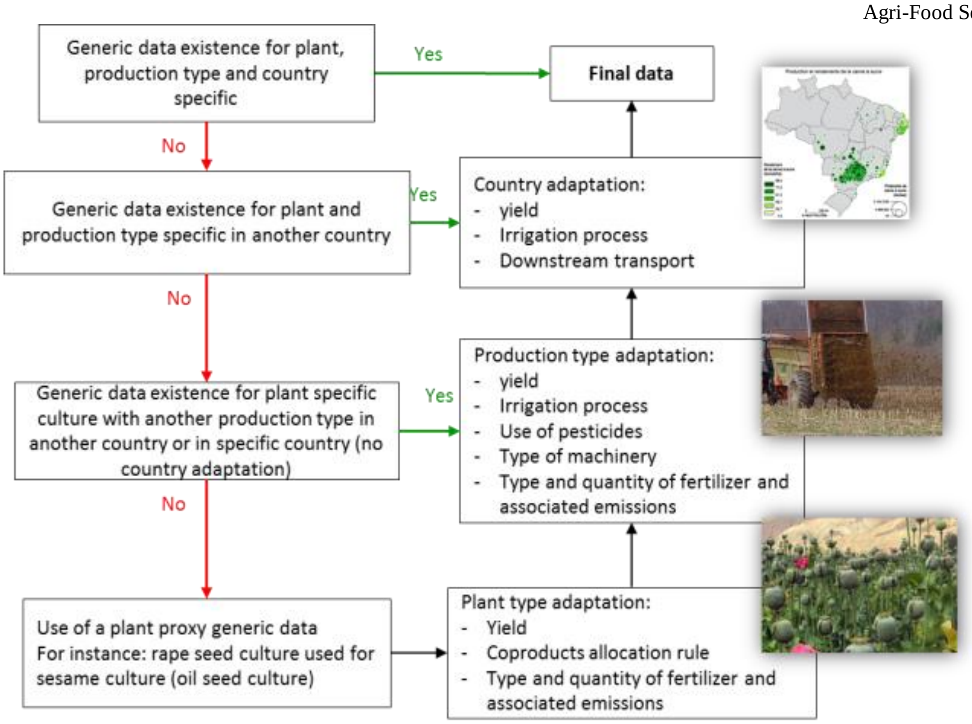


Figure 1. Process for adapting data

Table 1. For two ingredients used in the biscuit, the processes included, the generic data used, and the main adaptations performed on these data.

Ingredients	Ingredients production steps	Generic data used	Adaptations
Organic brown sugar	Sugarcane culture	Ecoinvent data Sugarcane, at farm/BR	Country specific yield for organic production
			Country specific irrigation process
			Mineral fertilizer substitution by organic fertilizer
			Fertilizer needs (N, P, K) for organic production and related field emissions adapted
			Removal of pesticides inputs
	Sugar production	Ecoinvent data Sugar, from sugarcane, at sugar refinery/BR	Country specific yield for organic production
Organic toasted complete sesame seeds	Organic sesame culture	Ecoinvent data Rape seed organic, at farm/CH	Country specific yield for organic production
			Fertilizer needs (N, P, K) for organic production and related field emissions adapted
	Toasting	Data creation	

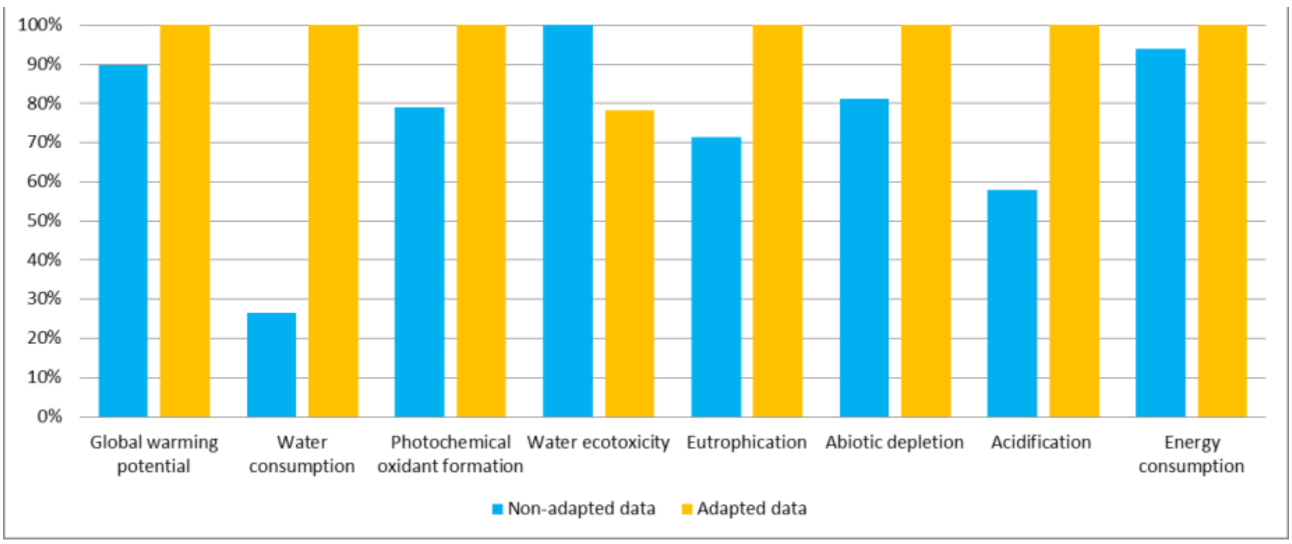


Figure 2. Environmental impacts of a packet of biscuit with non-adapted data or data adapted for cultural steps. LCIA performed with SimaPro 7 software, according BPX30-323 guidelines (2011)

125. Prioritising retail food waste prevention - potatoes, tomatoes or carambolas?

Ingrid Strid

Swedish University of Agricultural Sciences, Dept of Energy and Technology, P.O. Box 7032, SE-750 07 Uppsala, Sweden, E-mail: *ingrid.strid@slu.se*

Loss of food is a major problem world-wide with regard to the environment and to food security. To prevent waste efficiently, a better understanding of the conditions behind the wastage is needed. This study is part of a research project focusing on food wastage at the retail level of the Swedish food supply chain, conducted by the Swedish University of Agricultural Sciences during 2010-2013 (www.slu.se/foodwastage). After identifying the main products driving the food wastage, the question of how to prioritise prevention options arises. In the present study, the fruit and vegetable department at six Swedish retail stores were studied. Data on sold and wasted quantities for 2010 and 2011 were obtained from the retail company. The resulting top lists of wasted items, in tonnes and as a percentage of sold volumes, gave a quantitative background necessary for the subsequent task to priorities between mitigation targets. Potatoes and lettuce dominated the wastage by mass, while rare exotic fruits had the highest waste percentage. In the next phase, the product specific carbon footprints were multiplied with the wasted amounts to quantify the carbon footprint of each waste fraction. These LCA-based results shifted the list, so that tomatoes and sweet peppers now dominated the impact. In absolute numbers, the carbon footprint of food wastage was highest for tomatoes and peppers, leading to a possible conclusion to target these products for waste prevention measures at the fruit and vegetable departments studied. However, an alternative evaluation method was also tested in order to relate the burdens from the wasted fraction to the benefit of the respective product. The benefit of each product was indicated with its sold volume, reflecting the food supply capability of the product. In this additional analysis the wastage carbon footprint was divided by the sold quantify of each product type to give an index of the un-necessary environmental impact per kg sold product. The result gave that rare exotic fruits totally dominated this recalculated list, where the carbon footprint of the wastage from bulk products added grams to the total results, while the corresponding figure for each kg sold rare exotic fruit was 7.2 kg CO_2-eq extra to the (already high) product specific carbon footprint of 11 kg CO_2-eq. When relating the environmental burden of the wastage to the sold quantity of products, the conclusion became that rare exotic fruits should be prioritised for waste prevention measures at the vegetable departments of the retail chain studied.

References

Carlsson-Kanyama, A., D Gonza´lez, A., 2009. Potential contributions of food consumption patterns to climate change. The American Journal of Clinical Nutrition 89(5), 1704-1709.

Cellula, M., Ardente, F., Longo, S., Mistretta, M., 2010. Life Cycle Assessment (LCA) of protected crops: an Italian case study, 449-454, in: Proceedings from LCAfood 2010 VII International conference on Life Cycle Assessment in the agri-food sector, volume 1. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.). Universita Degli Studio di Bari Aldo Moro, Bari, Italy.

Davis, J., Wallman, M., Sund, V., Emanuelsson, A., Cederberg, C., Sonesson, U., 2011. SIK report 828, Emissions of Greenhouse Gases from Production of Horticultural Products. The Swedish Institute of Food and Biotechnology, Gothenburg, Sweden.

Karlsson, H., 2011. Seasonal Vegetables - An Environmental Assessment of Seasonal Food. Master Thesis, Norwegian University of Life Sciences, Department of Plant and Environmental Sciences.

Müller-Lindenlauf, M. and Reinhardt, G., 2010. Food import versus regional production: comparison of energy demand and greenhouse gas emissions 487-492, in: Proceedings from LCAfood 2010 VII International conference on Life Cycle Assessment in the agri-food sector, volume 1. Notarnicola, B., Settanni, E., Tassielli, G., Giungato, P. (Eds.). Universita Degli Studio di Bari Aldo Moro, Bari, Italy.

Röös, E., Sundberg, C., Hansson, P-A., 2010. Uncertainties in the carbon footprint of food products: a case study on table potatoes. The International Journal of Life Cycle Assessment 15(5), 478-488.

Table 1. Yearly sold and wasted fruit and vegetables at six Swedish retail stores

Product	Sold (tonne/yr)	Wasted (tonne/yr)	Waste quota (%)	Product carbon footprint kg CO_2-eq/kg product	Wastage carbon footprint tonne CO_2-eq of yearly wastage	Wastage carbon footprint per quantity sold g CO_2-eq/kg product
Potato	1616	10	0.62	0.12[1]	1.2	0.7
Lettuce	349	7.3	2.1	1.0[2]	3.6	10
Tomato	743	6.8	0.90	0.9[3]	6.1	8.2
Sweet pepper	271	5.4	1.9	1.1[4]	5.9	22
Carrot	439	4.5	1.0	0.18[5]	0.8	1.8
Banana	768	4.4	0.57	1.1[6]	4.8	6.3
Rare exotic fruits (Tamarillo, Pithaya, Pepino, Prickly pear, Carambola, Rambutan)	0.6	0.39	39	11[7]	4.3	7200

[1]Röös et al., 2010
[2] Müller-Lindenlauf and Reinhardt, 2010
[3] Karlsson, 2011
[4] Cellula, et al., 2010
[5] Davis et al., 2011
[6] www.dole.com (accessed 2012-02-10)
[7] Carlsson-Kanyama and D. González

126. LCA of waste from olive oil manufacturing

Bruno Notarnicola[1,*], Bernardo De Gennaro[2], Giuseppe Tassielli[1], Pietro Alexander Renzulli[1]

[1] Ionian department of Law, Economics and Environment, University of Bari, Italy, [2] Department of Agro-Environmental and Territorial Sciences, University of Bari, Italy, [*] Corresponding author. E-mail: *brunonotarnicola@gmail.com*

Competition on both national and international olive oil markets is becoming increasingly intense, thus forcing organisations to identify new and diverse competitive advantages, particularly by increasing the efficiency within the firm. So far, the prevailing action undertaken by firms, with regards to this, has been to search for process innovations that also entailed a reduction of unit production costs (Notarnicola et al., 2003). However, the current development paradigms show that in order to obtain the best results, the entire supply chain must be designed as a whole, trying to foresee the flows going through the whole economic system rather than just one firm or process.

This aspect becomes particularly relevant for the olive oil supply chain, in which alternative options can be found in both the cultivation stage (traditional, super-intensive) (De Gennaro et al., 2012) and the industrial processing stage (two or three phases) and the alternative oil products (Nicoletti et al., 2001). Each option generates final outputs together with solid and liquid waste with very different characteristics.

In this research a comparison between various alternative options for producing olive oil – discontinuous, two-phase and three-phase systems – is performed by means of LCA. Particular attention is placed on the treatment of waste that is generated in each processing stage of the olive oil supply chain, for each alternative option.

These processes differ in terms of the yields and the organoleptic quality of the finished product, but the principal differences regard the quantity and quality of waste: variation of the humidity levels of the pomace and the concentration of vegetation water.

The outcomes of this study show the environmental profile of the systems considered, and they are expected to contribute to the current debate on whether a three phases-system, which is widely adopted in some areas like the Apulia region in Italy, should be transformed in a two phases-system, by means of economic incentives supporting the replacement of the old plants too.

References

De Gennaro B., Notarnicola B., Roselli L., Tassielli G., 2010. Innovative olive-growing models: an economic and environmental assessment. J. Cleaner Production, doi:10.1016/j.jclepro.2011.11.004

Nicoletti G. M., Notarnicola B., Tassielli G., 2001. Comparative LCA of virgin olive oil vs. seed oils. The International Conference LCA in Foods, Goteborg 25-27 April 2001.

Notarnicola B., Tassielli G., Nicoletti G. M., 2003. LCC and LCA of extra-virgin olive oil: organic vs. conventional. 4th International Conference: Life Cycle Assessment in the Agri-food sector, Horsens, 6-8 October 2003.

127. Environmental impact of animal food products and their substitutes

Trudy Rood[1], Durk Nijdam, Henk Westhoek, A. Leip[2], J.P. Lesschen[3]

[1] PBL Netherlands Environmental Assessment Agency, Bilthoven, The Netherlands, [2]European Commission - Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy, [3]Alterra, Wageningen-UR, Wageningen, The Netherlands, * Corresponding author. E-mail: *trudy.rood@pbl.nl*

Land use, greenhouse gas emissions (GHG) and nitrogen (N) emissions are the main causes of global loss of biodiversity and damage to ecosystems (Rockstrom et al., 2009). Major impacts on the environment, and thus also on biodiversity, are caused by the production of meat, dairy and fish. In this study, different protein sources, such as meat, dairy, fish, eggs and meat substitutes, are compared for their environmental impacts.
In order to identify the range of impacts, their most important related factors, as well as the main causes of the differences between products, 44 life-cycle assessment (LCA) studies were analysed, containing 96 LCAs of animal and vegetal sources of protein. Moreover, the results for agricultural products were compared to results from the model Miterra-Europe. The Miterra-Europe model was used to calculate greenhouse gas and nitrogen emissions from agriculture, following a life-cycle approach that reached 'up to the farm gate' (Lesschen et al., 2011).
Compared to other review studies, such as De Vries and De Boer (2010), Yan (2011), Roy et al. (2009), Flachowsky and Hachenberg (2009), and González et al.(2011) containing a selection of LCA studies on animal products and mainly focused on greenhouse gases, our review study presents a broader view.
There are very large differences in carbon footprints and land requirements between the various protein sources in the human diet. Greenhouse gas emission levels from the most climate-friendly protein sources are up to 100 times lower than those from the most climate-unfriendly protein sources. For land use, comprising both arable land and grasslands, this varies even more strongly. In the case of grasslands, there are also large differences in the quality of land use in terms of biodiversity. Vegetal sources, poultry products and certain seafood have well below average environmental impacts, while those of ruminant meat and some other types of seafood are well above the average.
The impact differences between the various products were found mainly to be due to differences in production systems. In the life cycle of protein sources, in general, the farm phase is the most important. Further processing, transportation and packaging are of less importance.
The differences in scores, both between and within the various product categories, offer chances for lowering the environmental impact of our protein consumption. Shifting consumption towards other sources of protein has a large potential for reducing the impacts on biodiversity and climate change.

References

De Vries, M., de Boer, I.J.M., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livestock Science 128, 1-11.

Flachowsky, G., Hachenberg, S., 2009. CO2 Footprints for Food of Animal Origin – Present Stage and Open Questions. Journal für Verbraucherschutz und Lebensmittelsicherheit 4, 190-198.

González, A.D., Frostell, B., Carlsson-Kanyama, A., 2011. Protein efficiency per unit energy and per unit greenhouse gas emissions. Food Policy, In Press.

Lesschen, J.P., van den Berg, M., Westhoek, H.J., Witzke, H.P. and Oenema, O., 2011. Greenhouse gas emission profiles of European livestock sectors. Animal Feed Science & Technology 166-167, 16-28.

Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sorlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature 461, 472-475.

Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T., 2009. A review of life cycle assessment (LCA) on some food products. Journal of Food Engineering 90, 1-10.

Yan, M.J., Humphreys, J, Holden, N. M., 2011. An evaluation of life cycle assessment of European milk production. Journal of Environmental Management 92, 372-379.

128. Industrial ecology and agri-food sector. Perspectives of implementation in an Italian regional cluster

Raffaella Taddeo*, Alberto Simboli, Anna Morgante

University "G. d'Annunzio", Department of Economic Studies, V.le Pindaro 42, 65127 Pescara, Italy, * Corresponding author. E-mail: *r.taddeo@unich.it*

Over the last fifty years, the development of the agri-food sector has been characterised, in many developed countries, by significant structural, technological and organisational changes, which have led to the involvement and integration, in traditional food production chains, of a number of activities, such as: food processing, manufacturing of technical equipment, packaging industry, transportation, storage, distribution, marketing, catering. The spontaneous agglomeration of agricultural activities in favorable geographic areas, has often led to the development of clusters of SMEs, recognised as agri-food clusters. Agri-food clusters are characterised by typical environmental impacts, such as: land use, CO_2 emission, energy and water consumption, use of agricultural inputs such as fertilisers, pesticides, feed additives and irrigation water. However, the involvement of agro-industrial activity in the same area highlights other significant sources of waste related to auxiliary materials and different types of packaging used during agricultural activities and food processing (materials such as polystyrene, polyethylene, polypropylene, wood, paper). Industrial Ecology, through a more efficient management of material and energy flows, helps to reduce loads and environmental impacts of production activities without compromising their competitiveness. Major applications of Industrial Ecology principles in the agri-food sector, concern the valorisation of animal and vegetable by-products, leading to the implementation of so-called Agro-Eco-Industrial Parks. The purpose of an Agro-Eco-Industrial Park is to provide a base for companies and service organisation in achieving a transition to sustainable farming, improving the value of their output and gaining market channels. In an agri-food cluster, alternative and effective solutions can be also implemented to manage waste flows deriving from auxiliary materials used in agro-industrial activities, through the adoption of closed loop approaches, especially considering technical features of such flows: high volumes, high percentage of non-hazardous materials, homogeneity in composition, and regular (or cyclic) flows. This paper presents a preliminary analysis of one of the most representative agri-industrial clusters in the production of horticultural products, the area of Fucino in Abruzzo Region (Italy). The cluster covers an area about 15,000 ha, for a total of 3,700 small and micro-sized enterprises that produce mainly carrots, potatoes, endive and lettuce (Fig. 1). The study aims to analyse the main vegetable and not vegetable waste flows to propose alternative options for managing them in the perspective of an Agro-Eco-Industrial Park. The preliminary qualitative analysis shows that efficient solutions can be potentially implemented through recycling, recovery and repair activities, materials substitution and alternative energy production, exploiting synergies of the existing cluster (Fig. 2).

References

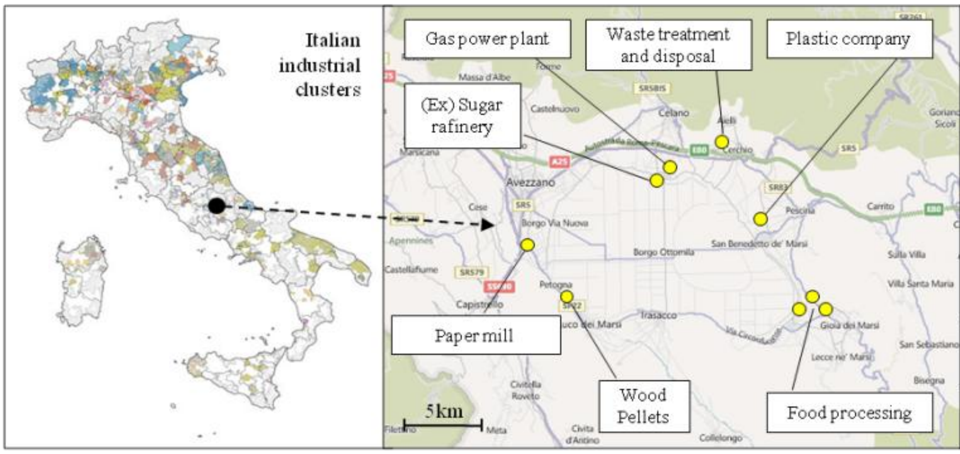
Ayres, R., Ayres, L. W., 1996. Industrial Ecology: Towards closing the material cycles, first ed. Edward Elgar, Cheltenham.

Indigo Development, 2005. Creating systems solutions for sustainable development through Industrial Ecology. Developing Agricultural Eco-Industrial Parks in China. Available at http://indigodev.com/AEIPwhitepaper.html. Accessed January 2012.

Kroyer, G. Th., 1995. Impact of food processing on the environment – an overview. LWT – Food Science and Technology 28, 547-552.

Nguyen Ngoc, U., Schnitzer, H., 2010. Environmental understanding of agro-based industrial systems and sustainable development. International Journal of Environment and Sustainable Development 9, 149-180.

Taddeo, R., Simboli, A., Morgante, A., 2011. Symbiotic exchanges and hazardous wastes in Chemical Industry. A "Path-Dependent" Case Study, 21th Setac Europe Annual Meeting, Milano Convention Centre, 15-19 May. Proceedings.



CROP	ha
Corn	975,6
Potato	4378,0
Grass	381,9
Carrot	2303,9
Lettuce	1660,8
Other	3387,0

Figure 1. The Fucino agri-food cluster and its main productions (2011).

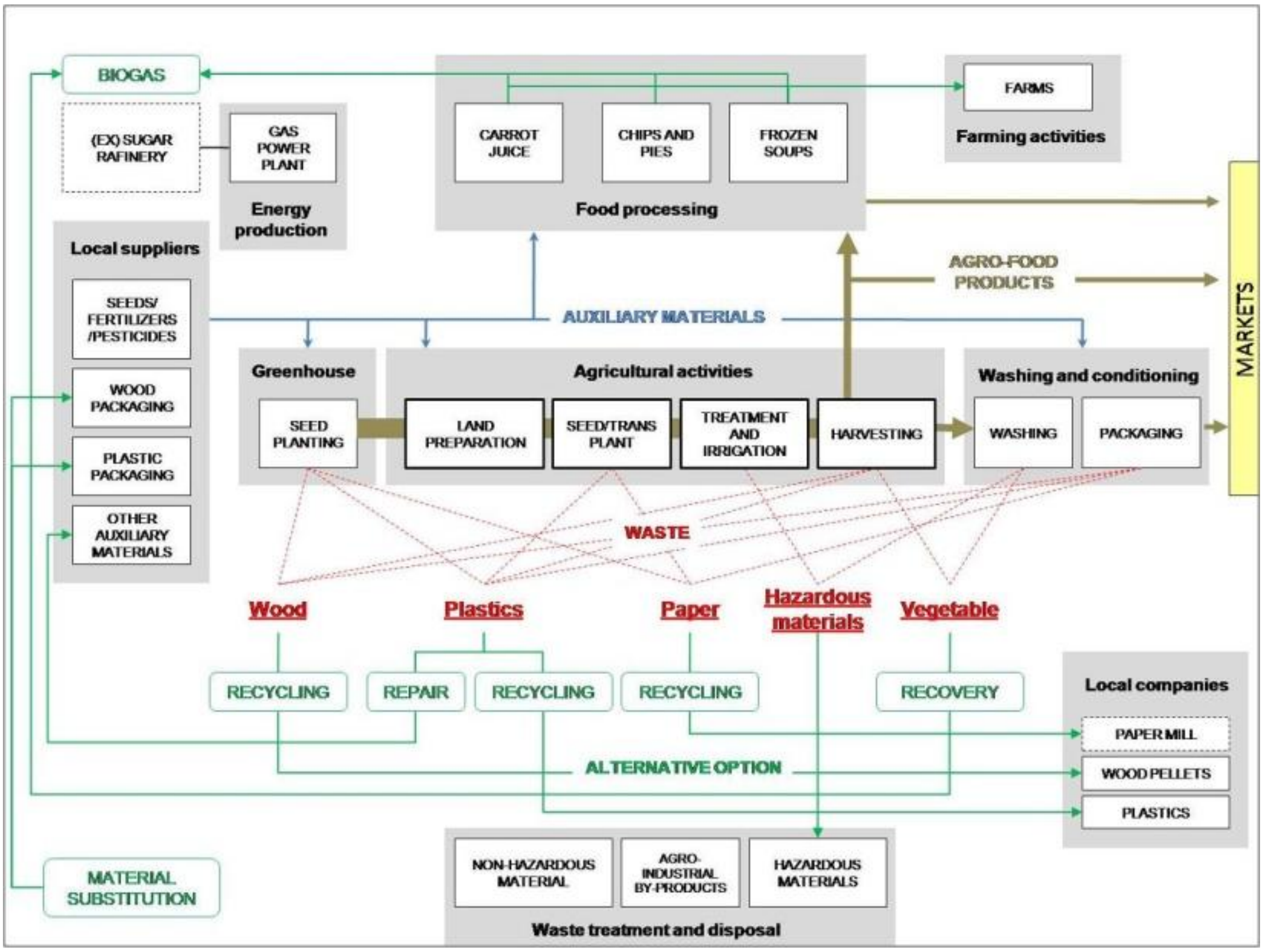


Figure 2. Potential solutions in the Fucino agri-food cluster.

129. LCAs for animal products pork, beef, milk and eggs in Finland

Kirsi Usva*, Jouni Nousiainen, Helena Hyvärinen, Yrjö Virtanen

MTT Agrifood Research Finland, * Corresponding author. E-mail: kirsi.usva@*mtt.fi*

This paper presents results of LCA for pork, beef, milk and eggs produced in Finland. Environmental impact categories assessed were climate change, eutrophication and acidification and, additionally, land and energy use. Several LCAs had been implemented on Finnish food products earlier, but there was a need for comprehensive environmental impact assessment of most important food products. Both plant and animal products were assessed.

The functional unit of LCAs is kg of a product at the farm gate (without packaging). System boundaries include animal production (heating, lightning, mechanical work) on the farm, as well as complete chains of the animal feeds, fuels and power which supply that. The supply chains include all significant industrial processing, product packaging, energy and transports. Data for the supply chains were obtained from the industry, which produce the majority of the inputs used in the animal production in Finland. Energy was assumed according to the Finnish average grid. Data on the use of inputs at crop and grass production were based on the national agricultural database consisting of data on the cultivation instances of various crop species and grass, i.e. it is primary data. Data on mechanical work were based on physical models.

Animal production models were used to assess partly the inventory data and partly the impacts of animal production. Animal models consist of animal population age-classes, their specific feed consumption and diet composition. This information in utilised in simple balance calculations (nutrient intake in feeds – nutrient retention in growth and products) of nitrogen and phosphorus and gross energy intake that is necessary in methane production estimation. Models were based on e.g. national statistics and calculation based on feeding norms. Methane emissions were estimated according to models used in Finnish greenhouse gas inventory. Nitrogen amount in excrement and urine was assessed by animal model (nitrogen balance) and assessment of NH_3 and N_2O emissions in animal shelter and manure storage were based on this.

Emissions from manure storage were allocated to animal production. Emissions from manure spreading on the field were allocated to the those plants the manure was used as fertiliser for. Allocations were needed also to allocate inputs and emissions especially in beef and milk –case between milk and meat. In Finland most of the beef production is connected to milk production. For pork and beef allocations were done between different qualities of meat. Allocations were calculated accomplished economic values of different products. In egg production all inputs and emissions were allocated to eggs.

Results of LCAs in terms of climate change, eutrophication and acidification are shown in Fig. 1-3. Environmental impacts of beef are more than twice as much as impacts of pork. In case of climate change the methane emissions from bovine are higher than from pigs. In terms of eutrophication, most of the impacts derive from feed production in Finland. Pigs and chickens use soy which do not cause as much eutrophication impact as Finnish feeds. NH_3-emissions from manure are the main reason for acidification impact. An important thing is, that feed conversion ratio of pigs and chickens are better than bovines'.

Food stuffs are one of the most important consumer goods in terms of environmental impacts of consumption. Animal models are a strong base for emission assessment in animal shelters and manure storage. Machinery work models represent a typical situation in Finland as well as feed production models based on national agricultural data. As such the results of the LCAs represent the typical Finnish animal products. Together with LCAs of plant products they are very valuable in comparing the environmental burdens of different food stuffs in Finland. Results may be utilised in communication to consumers, political decision-making and improvement of animal product supply chains and production systems.

References

Grönroos, J., Mattila, P., Regina, K., Nousiainen, J., Perälä, P., Saarinen, K. and Mikkola-Pusa, J. 2009. Development of the ammonia emission inventory in Finland. pp. 15-19.

Saarinen, M., Virtanen, Y. and Hyvärinen, H. 2012. LCAs for a large repertoire of Finnish outdoor plant products. Abstract for LCA Food 2012 Conference.

Statistics Finland 2011. Greenhouse Gas Emissions in Finland 1990-2009. pp. 248-253.

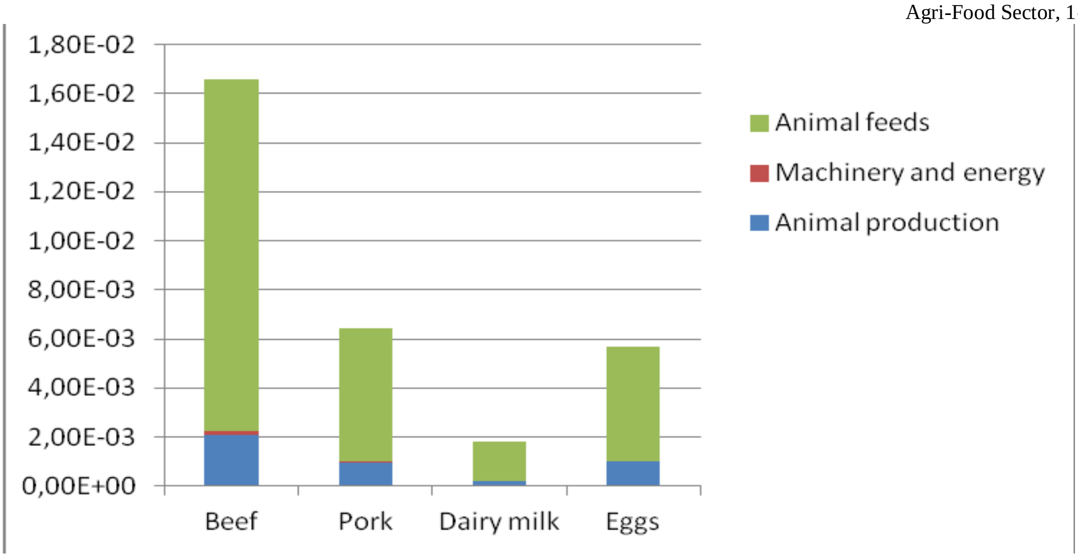


Figure 1. Eutrophication impact of animal products (kg PO_4 eq./product kg).

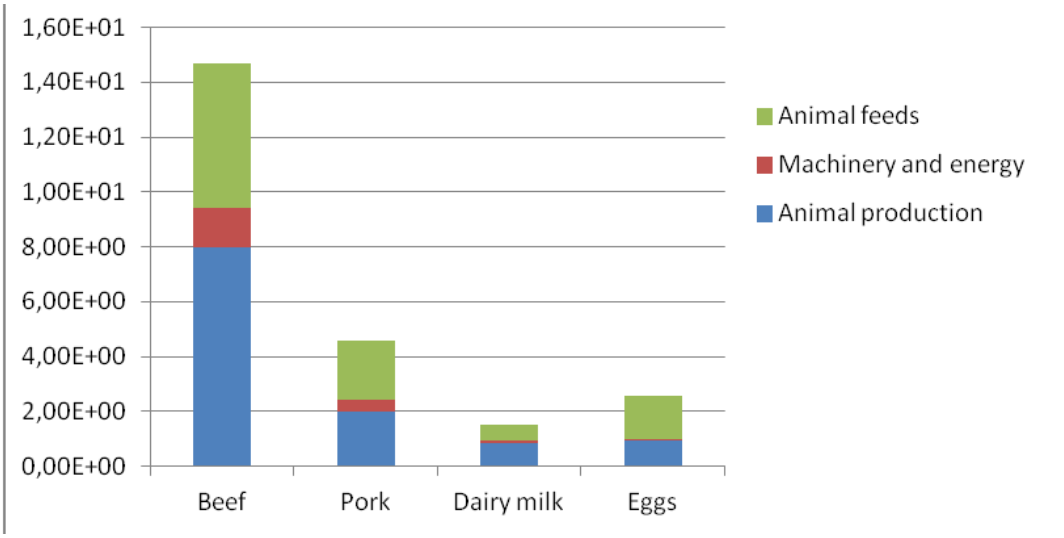


Figure 2. Climate change impact of animal products (kg CO_2 eq./product kg).

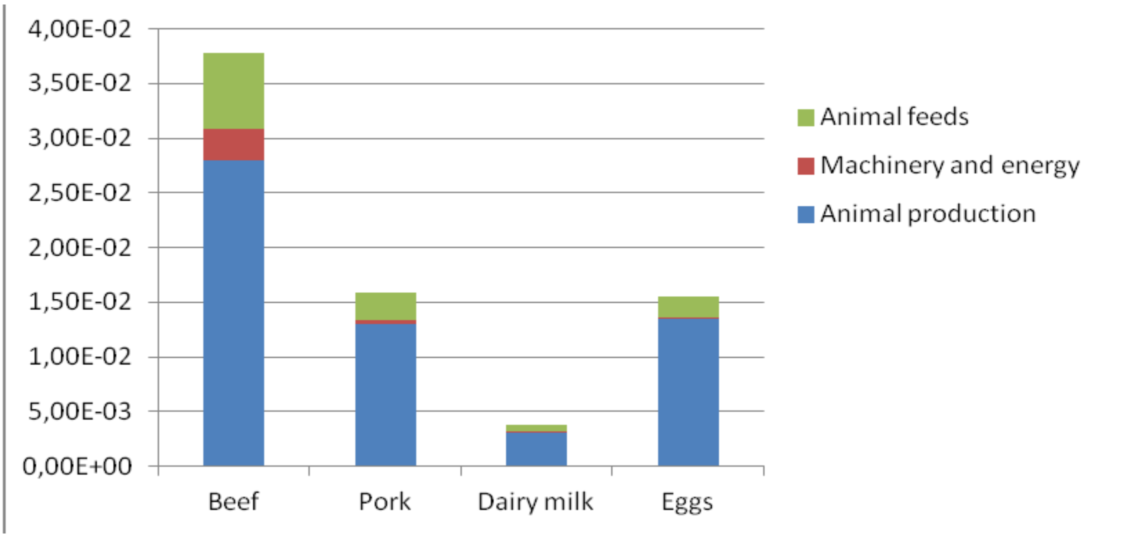


Figure 3. Acidification impact of animal products (AE eq./product kg).

130. Greenhouse gas emissions related to saturated fat, sodium and dietary fibre content of food products

Corné van Dooren* , Annely Douma

Netherlands Nutrition Centre (Voedingscentrum), P.O. Box 85700, 2508 CK The Hague, The Netherlands, * Corresponding author. E-mail: *dooren@voedingscentrum.nl*

The Dutch Health Council concludes in her 2011 advice that a healthy diet according to the Wheel of Five model is usually also environmentally friendly. Furthermore, she states that there is a strong correlation between various indicators of sustainability, such as greenhouse gases (GHG), energy and land use (Health Council 2011). The Italian double pyramid shows that there is an inverse relationship between breakdown in health gain and environmental impact, expressed as carbon, water and ecological footprint (Buchner, Fischler et al. 2010). Our study investigates and quantifies this hypothesis with Dutch data.
Data sources: food composition (RIVM 2011); land use and energy use (Gerbens-Leenes 2006) water use (Hoekstra and Chapagain 2004) and GHG (CLM, klimaatweegschaal.nl; 2009 not published). Correlation between GHG, energy use, water use and land use was calculated with a Spearman rank correlation test. Further analyses were only done with the GHG data as indicator (n=403). Products were divided into 6 categories according to the Wheel of Five education model. A Kruskal Wallis and a Mann-Withney U test were performed to determine differences between groups of products (Fig. 1).
To find whether a healthy food pattern could be in line with a sustainable food pattern, the products were divided into a broad accepted division of Preference, Neutral and Exception, based on saturated fat, sodium, added sugar and fibre content (Voedingscentrum 2011). Differences in GHG of these groups were explored. To further investigate whether this mentioned nutrients affects the emission of GHG, the nutrients were included in a regression analysis (for this analyses GHG data was log transformed).
We find a correlation between GHG and land, energy and water use (all $r>0.354$, $p<.01$; Table 1). There is a significant difference in GHG between the Wheel of Five groups ($\chi2(5)= 175.51$, $p<.001$). Follow up analyses show a significant difference between the animal protein rich product group and all other groups ($p<.001$). Analyses indicate that less healthy food items are also less sustainable. A difference in GHG was found between the Preference, Medium and Exception category ($\chi2(2)= 30.131$, $p<.001$). The categories low in GHG (Cat. A & B) consist almost entirely of preferred products, whereas exceptional foods mostly fall into higher emission categories (Cat. D & E) (Fig. 2). The products in the Exception category has a 2 times higher median compared to the products in the Preference category (200g vs 408g CO_2eq/100g $p<.001$). The finding that unhealthy foods have a higher GHG was confirmed in the regression analyses; in the model saturated fat and sodium are positively associated with GHG, whereas dietary fibre was negatively associated with GHG. An exception was added sugar which was also negatively associated with GHG.
GHG (CO_{2eq}) = 10×((2.356 + (saturated fat g *.019) + (sodium g *.279) – (dietary fibre g *.024) – (added sugar g *.021)).
About 23% of the variability in the GHG can be explained by these predictors. The results strongly support the concept that Dutch health advices are in line with sustainability indicators. Lowering consumption of products high in animal protein, saturated fat and sodium, is a clear consumer advice. Using more Preference products, like fruits and vegetables rich in dietary fibre, instead of Exception products helps consumers to eat a more healthy and sustainable diet.

References

Buchner, B., C. Fischler, et al. (2010). Double Pyramid: healthy food for people, sustainable food for the planet. Milan, Barilla Center for Food & Nutrition**:** 150 p.

Gerbens-Leenes, P. W. (2006). Natural resource use for food: land, water and energy in production and consumption systems: 155p.

Health Council (2011). Guidelines for a healthy diet: the ecological perspective. The Hague, Gezondheidsraad**:** 92p.

Hoekstra, A. Y. and A. K. Chapagain (2004). Water footprints of nations: Volume 1: Main Report, Value of Water Volume 2: Appendices. Delft, Unesco-IHE**:** 80 p.

RIVM (2011). NEVO-tabel: Dutch nutrient files (in Dutch). Bilthoven, RIVM.

Voedingscentrum (2011). Guidelines for food choices (in Dutch). The Hague, Voedingscentrum**:** 61p.

Table 1. Indicators of sustainability are strongly correlated, except water use with energy use.

	GHG	Virtual water		Energy use		Land use	
GHG	1.000	**0.668**	**	**0.584**	**	**0.354**	**
Virtual water		1.000		0.132		**0.794**	**
Energy use				1.000		**0.294**	*
Land use						1.000	

*** p<0.01; * p<0.05*

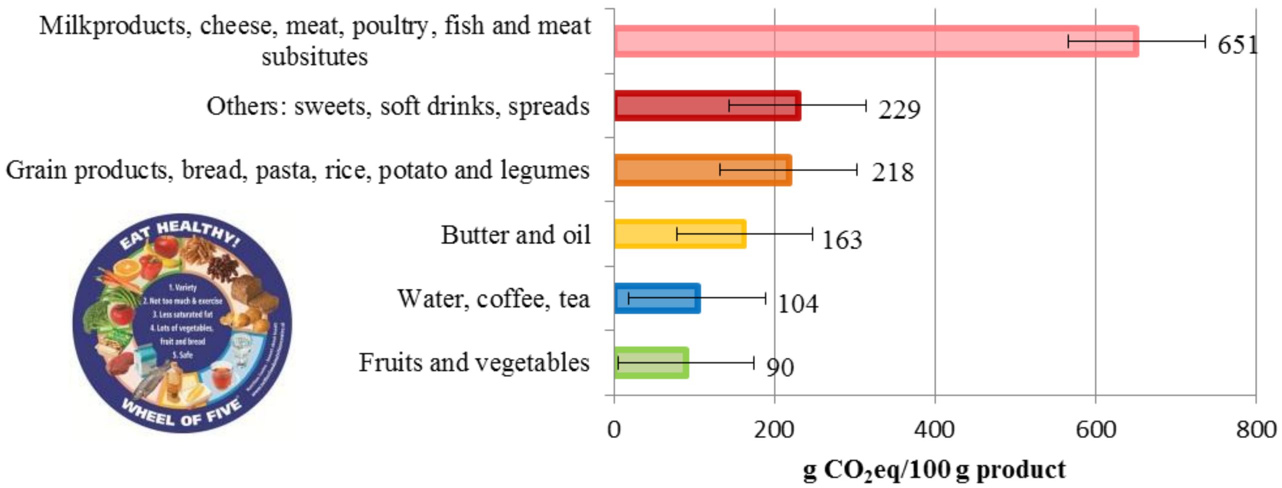


Figure 1. Protein rich animal products have significant (3 to 7 times) higher GHG emissions than other food groups.

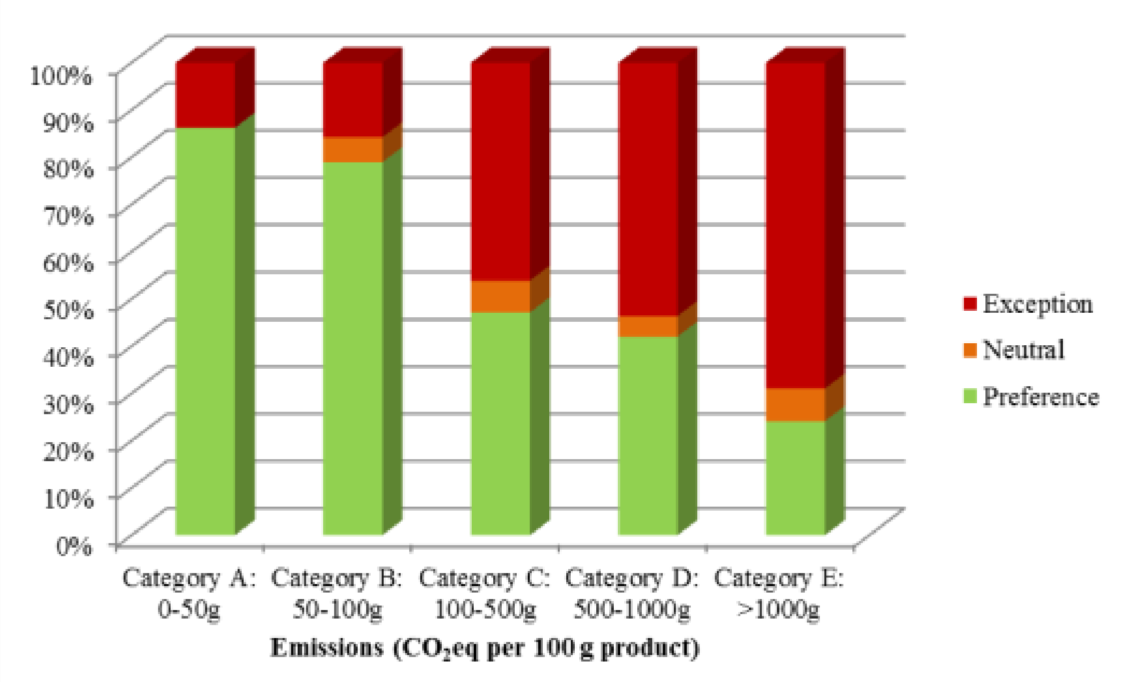


Figure 2. Exceptional foods from health point of view have higher GHG emissions (Cat. D & E).

131. Harmonising LCA methodology: A collaborative approach in a search for allocation rules for food sector

Hanna Hartikainen*, Juha-Matti Katajajuuri, Frans Silvenius

MTT Agrifood Research Finland, 00790 Helsinki, Finland, * Corresponding author. E-mail: *hanna.hartikainen@mtt.fi*

There are several general and more specific life cycle assessment (LCA) calculation guides. However, the food sector is lacking a common guide that is taking into account the special features of the sector and, thus, giving practical and harmonised guidelines for calculating food products' footprints. There are few initiatives, such as, the methodology for food products structured by the European Food SCP Round Table. Another initiative is the Finnish project called "Foodprint" (2009–2012). The project's aim is to create harmonised and practical attributional LCA methodology and guidelines for, particularly Finnish, food sector. The key objectives of the project are: 1) to actively follow and influence the harmonisation of LCA methods both internationally and nationally, 2) to involve Finnish food chain actors into the development process (via pilots, steering group meetings, workshops, seminars), and 3) to attain more supply chain specific data from the entire food chain. The last target is both to improve the food chain's operations and to give more both reliable and comparable information to consumers of the environmental impacts of the food product in question.

When drafting the Finnish LCA methodology for food products one central question found out to be the allocation rules, especially in the multi-output systems. As well known, LCA's aim is to assess environmental impacts associated with all the stages of a product's lifecycle. The problem is that allocation is lacking unambiguous basis and, thus, jeopardising the credibility of the LCA methodology. Allocation decisions are easily influenced by the analysts' perspectives and worldviews and, thus, there are arguments from different angles whether, for instance, economic, biophysical or mass-based allocations are suitable or not in different case-studies/decision-making situations.

The discussion about allocation methods has been going on and on in the research community but methods' usefulness in reality is not always fully considered. Therefore, the actual barriers for the acceptable (depends on the viewer) allocation methods are easily forgotten. In the Foodprint project, besides a broad literature review and active following of international discussion on allocation methods, there have been several discussions with the Finnish food chain actors to discover allocation methods that are simultaneously comprehensive, suitable and practical. In these discussions, for instance, biophysical allocation (read e.g. IDF 2010: allocation between milk and beef (Appendix B)) raised interest but its complexity and limited use were considered its definite drawbacks. Additionally, internationally widely preferred economic allocation was seen problematic for many reasons. First of all, the market prices fluctuate and are easily influenced by various external factors. Secondly, if earlier prices (e.g. products' values prior to any further processing) are used, these prices depend on producers' pricing strategies, and thus, not on the 'actual values' of products. These prices are also often trade secrets. All in all, in economic allocation the allocation proportions are usually heavily influenced by many factors. Therefore, in order to improve the appropriateness of the use of economic allocation one should pay more attention to its uncertainties and weaknesses - at least uncertainties should be revealed when communicating the results.

Another important issue brought into the discussion by the Finnish food chain actors was that in order to receive harmonised results one needs strict allocation rules instead of rules that are loose and open to interpretations. It was stated that since food sector comprises several different food products, it would be best to agree on the best allocation methods or on the least bad alternatives in a more case specific level. This leads us closer to the product category rules' (PCRs') ideology, i.e. closer to product specific rules. Furthermore, it was stated by food chain actors that these more case specific discussions should also take into account different types of food product chains in order to attain more uniform allocation rules.

Altogether, the aim of the discussions and preferred practices in Finland is not to differentiate Finnish practices from international practices. On the contrary, the aim is to strengthen the international harmonisation and, thus, to share Finnish experiences and discuss them. We believe that the attempt to harmonise LCA calculation, to fully understand the weaknesses of allocation practices, and to find more appropriate approaches requires collaboration among the research community and, moreover, strong inclusion of the food chain actors into these discussions.

References

IDF (International Dairy Federation). A common carbon footprint approach for dairy, The IDF guide to standard lifecycle assessment methodology for the dairy sector, Bulletin of the International Dairy Federation 445/2010.

132. Assessment of the life cycle of a Portuguese wine: from viticulture to distribution

Belmira A. F. Neto[1,*], Ana Cláudia Dias[2], Marina Machado[3]

[1] CEMUC, Faculty of Engineering, University of Porto, 4200-465 Porto, Portugal, [2] CESAM & Department of Environment and Planning, University of Aveiro, 3810-193 Aveiro, Portugal, [3] Faculty of Engineering, University of Porto, 4200-465 Porto, Portugal, [*] Corresponding author. E-mail: *belmira.neto@fe.up.pt*

This study assesses the life cycle assessment of a wine produced in Portugal, i.e., the white green wine, designated as vinho verde branco. This is to identify the environmental impacts occurring along the wine life cycle. The stages considered include activities taking place during 1) Viticulture, 2) Wine production (from vinification to storage), 3) Distribution and 4) Bottles production.
Materials and energy consumption as well as the emissions to air, soil and water from the wine campaign of 2008/2009 were reported to the functional unit (0.75 litres of white green wine). A Portuguese company, located in the northern part of Portugal, responsible for the production of about 20% of the current total production of white green wine, supplied specific life cycle data, including information regarding transportation of grapes, wine, must and other wine production related products. Information concerning the distribution of this wine consumed worldwide is also made available by the company in terms of the amount sold to each country and the transportation mode.
The life cycle approach taken shows Viticulture as the stage that mostly contributes to most of the impact categories. The production of glass bottles appear as the second larger contributor and Wine Production and Distribution appeared as the third larger contributors. The Production of wine products and the transportations of grapes, wine products and wine and must have a comparatively negligible effect. Sensitivity analysis results show that some parameters are very influential.

133. Can you afford to make an eco-diet?

Sirpa Kurppa*, Kirsi Silvennoinen

MTT Biotechnology and Food Research, Team of Sustainable Bioeconomy FIN-31600 Jokioinen, Finland, * Corresponding author. E-mail: *sirpa.kurppa@mtt.fi*

The EU Commission has set the agricultural sector a recommended target of reducing greenhouse gas emissions by 50% by 2050, which would require a 35% reduction by 2030. Our aim was describe some possible changes in the Finnish diet that can help cut the emissions, while at same time benefitting public health, by e.g. preventing heart diseasesand overweight. Meanwhile, big changes in food patterns are usually difficult to achieve, we created several small adjustments all through the diet, keeping all the time the nutritional recommendations in mind. We assumed that this approach would be much more successful than recommending for example a vegan/vegetarian diet for everyone.

Our Eco-Diet's main aim is to decrease the environmental impacts of food chain, while it simultaneously encourages healthy eating patterns, is more economical to the consumer, and improves animal welfare.

In Eco-Diet, we propose that meals in general contain less meat and more vegetables, and we prefer seasonal products. Our Eco-Diet is a model that is easy to adapt to everyday life by everybody, without requiring any specific knowledge or education, or particular motivation for shifting towards vegan/vegetarian diet. Both of our benchmarked diets, the eco-diet and conventional diet used as a reference include meat, fish and dairy products.

We termed our conventional diet Basic Diet, and it was composed based on average Finnish diet with average amount of calories. The other one we termed Eco-Diet, which took into consideration climate change, water eutrophication, nutrition recommendations including appropriate amount of calories, and food waste. Both diets covered a full week of five working days and two days of weekend; each day having three meals: breakfast, lunch and dinner. Breakfast consisted of juice, bread or porridge, vegetable oil margarine, and some vegetables or fruit. Lunch and dinner consisted of meat, fish or bean main course and a side dish such as potatoes or rice. In addition, lunch and dinner included also a salad, bread, spread and drink.

The Eco-Diet included various kinds of meals; home cooked meals, convenience food and meals cooked in school canteens in communal food services. In our model we used school meals for lunch, home cooked meal or convenience food for dinner, and a home cooked breakfast. Here school meals were seen as an equivalent to office lunch. For the weekends, lunch and dinner were home cooked or convenience food meals, and breakfast was made at home.

We demonstrated that by following Eco-Diet for one week there was ca 40% decrease in carbon footprint, while the impact on water eutrophication was even more significant. Differences in environmental impacts between single food plate portions are remarkable: the highest animal based portions can have 5 times the environmental impact when compared to the lowest vegetarian/vegan portion. In Eco-Diet we managed cut off the highest impacts of the single food plates to such an extent that the difference between lowest and the highest impacts was only about three times, in both carbon footprint and water eutrophication.

We compared the cost of the Eco-Diet to the conventional diet, and found that for a consumer economy the Eco-Diet is a feasible, money saving alternative. Thus, we could oppose the preconception and correct the thinking that environmental enhancive diet would be more expensive and as such regarded as a premium diet. At the same time with saving the environment we can also save our money.

In Finland households are wasting about 5% of all purchased food[8]. Production of food that is lost in the food supply chain causes remarkable, unnecessary environmental and economic impacts. We assumed that Eco-Diet does not waste any food.

134. Value of a life cycle approach in evaluating the environmental impacts of packaging for food and beverage applications

Sonia Valdivia[1], Bruce Vigon[2], Jim Fava[3], Laura Flanigan[3,*]

[1] UN Environment Program, [2] Society of Environmental Toxicology and Chemistry, [3] PE INTERNATIONAL, Inc., [*] Corresponding author. E-mail: *l.flanigan@fivewinds.com*

The nexus between security and safety and environmental performance of packaging in food and beverages is a key question driving the market for sustainable packaging today. Therefore, an articulation of the benefits of the life cycle approach to design, manufacturing, use and end of life management of packaging for food applications is important to further examination of the role of packaging.

Key items of focus for this research and analysis included:

- What is the value of a life cycle approach for beverage and food products and packaging?
- What is the value of including all life cycle stages in evaluating the packaging/food systems to reduce overall life cycle impacts?
- What is the value of including multiple impacts in evaluating the packaging/food systems to reduce overall life cycle impacts?
- What is the value of including the food and/or beverage into an evaluation of the packaging life cycle impacts?
- What characteristics of future LCA studies should be considered when evaluating the food/packaging life cycle?

Examples of how the waste management hierarchy and LCA results interface/connect

This presentation will present the results of study to examine the value of a life cycle approach in evaluating the environmental impacts of packaging for food and beverage applications. The UNEP/SETAC Life Cycle Initiative brought to the project a unique combination of benefits that cannot be obtained from other sources. These benefits include:

- Neutral, objective, authoritative, and recognised forum for advancing understanding of packaging life cycle for food applications
- Global dissemination of the report
- Proven 10-year history of solid project deliverables

Acknowledgement: The authors thank the sponsoring organisations for their support.

References

Valdivia, S., Vigon, B., Fava, J., Flanigan, L. Value of a Life Cycle Approach in Evaluating the Environmental Impacts of Packaging for Food and Beverage Applications. UNEP/SETAC Life Cycle Initiative. Expected publication June 2012.

135. Spatial food-print to supply livestock products to Paris, 19th-21st centuries

Petros Chatzimpiros

Paris Interdisciplinary Energy Research Institute (PIERI), Université Paris Diderot, France, E-mail: *petros.chatzimpiros@univ-paris-diderot.fr*

When cities import food to sustain their metabolism, they virtually import land resources from the surrounding local and distant rural territories that generate the food. We refer to this vital physical substrate of urban metabolism as the spatial urban food-print (or food-print). The food-print locates where the food and animal feed crops are grown and its size depends on various parameters that can be divided into three categories: urban diets, crop yields and "feed to food" conversion ratios (Chatzimpiros and Barles, 2010).

In this proposal we determine the food-print to supply cereals and livestock products (beef, pork, chicken, dairy products and eggs) to the Paris metropolitan area since the early 19th century. Over this period, the food-print of fast growing occidental cities like Paris was the substrate of fundamental changes in the structure of agro-ecosystems and the common ground for transformations in both agricultural productivity and urban diets. Until the early 20th century, food supply to Paris is calculated from data records of the city's food markets. After that date, data become scarce due to progressive increases in the number of food supply chains and retail markets and, as for today, urban food consumption is not specifically known for any French city. Since the 1960s though, dietary discrepancies across France are low enough to allow deriving urban food supply from data on national food availability (production, plus imports, minus exports). We thus derived a time series of food supply to Paris since the early 19th century which we express as nitrogen (e.g. protein) and convert into land requirements for food production – the food-print of Paris – using data on food and feed crop yields (Statistique agricole annuelle) and nitrogen conversion efficiencies (NCE) in livestock production. For pork, beef and dairy production we used model-derived data of NCE for the early 19th, 20th and 21st centuries (Chatzimpiros, 2011). For chicken and egg production, we used data covering the second half of the 20th century (Lambier and Leclercq, 1992, Smith, 1997, Smil, 2002). We interpolated/extrapolated data on NCE over time proportionally to key variables such as biomass production rates.

Fig. 1 shows Paris population and its food-print since the early 19th century. Between 1850 and 2008, population grew 7-fold, food supply 8.5-fold and the food-print 2-fold. In Fig. 2, per capita supply (kg N/cap) is plotted with land requirements for production (ha/kg N) (land requirements decrease with time). The resulting curves show increases in the consumption of livestock products with low land requirements. As long as beef was the cornerstone of agrarian systems (<1950), urban consumption of beef was high.

References

Chatzimpiros P. and Barles S., 2010. Nintrogen, land and water inputs in changing cattle farming systems. An historical comparison for France, 19th–21st centuries, Science Tot. Environ., 408(22), 4644-4653.

Chatzimpiros P., 2011. Les empreintes environnementales de la consommation alimentaire: Paris ses viandes et lait, XIXe – XXIe siècles, Thèse de doctorat: Université Paris Est, Marne-la-Vallée, 368 p.

Larbier M. and Leclercq B., 1992. Nutrition et alimentation des volailles, INRA, Paris.

Smil V., 2002. Nitrogen and food production: Proteins for human diets. Ambio 31: 126-131.

Smith A. J., 1997. L'élevage de la volaille. Maisonneuve et Larose, Paris.

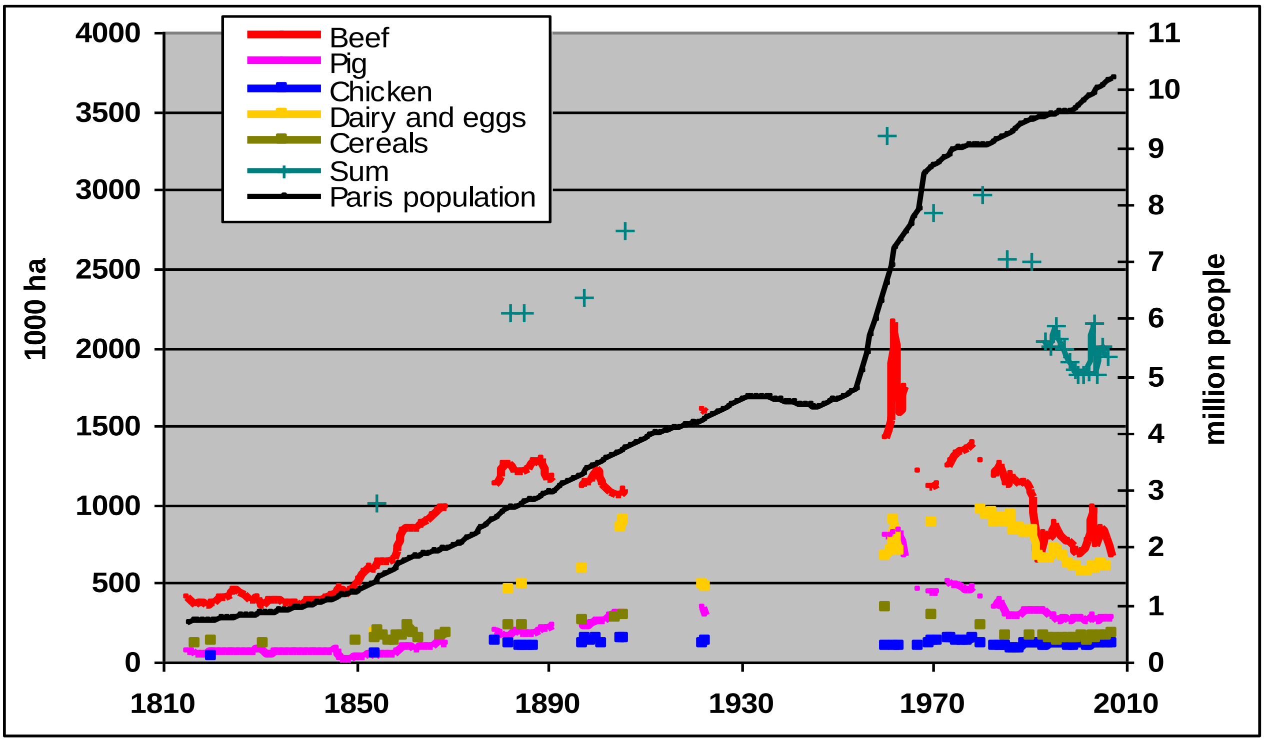


Fig 1: Food-print of Paris for cereals and livestock

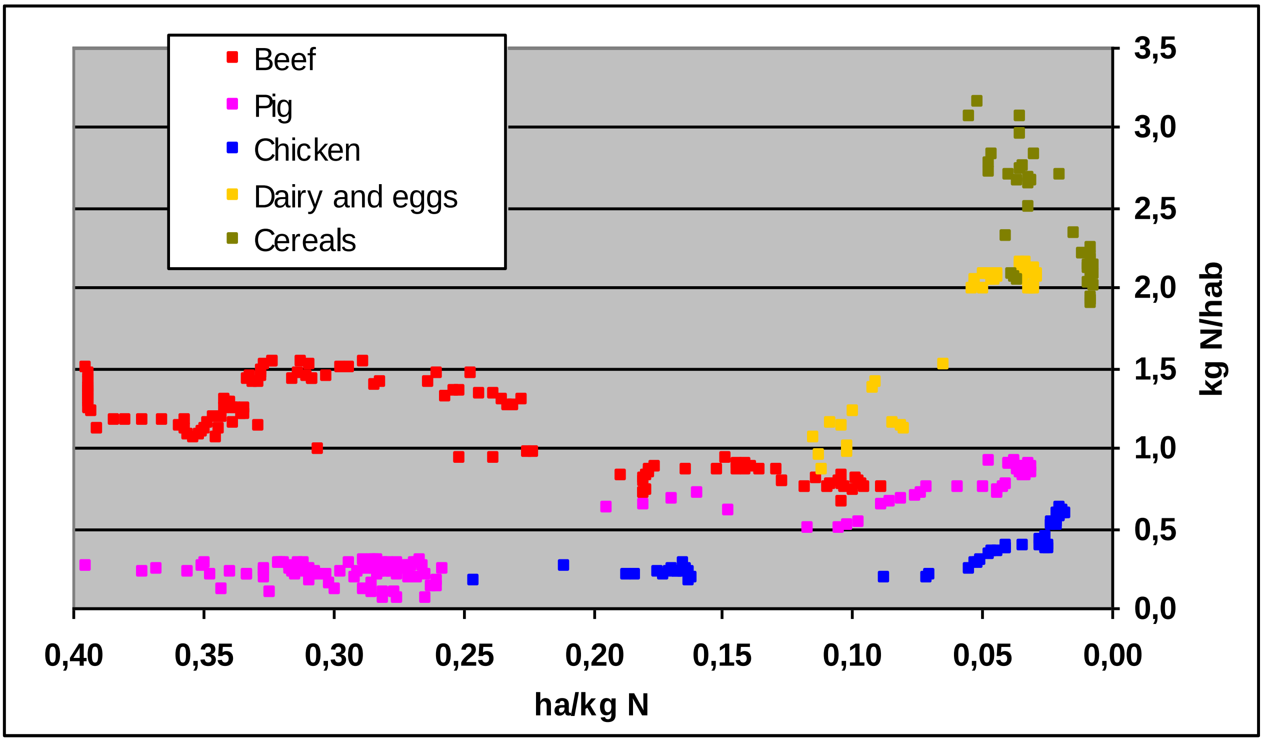


Figure 2. Land requirements and per capita consumption

136. Analysis of material and energy flow associated with food production and consumption in Japan

Naoki Yoshikawa[1,*], Koji Amano[1], Koji Shimada[2]

[1]Ritsumeikan University, College of Science and Engineering, Kusatsu, Japan, [2]Ritsumeikan University, College of Economics, Kusatsu, Japan, [*]Corresponding author. E-mail: *n-yoshik@fc.ritsumei.ac.jp*

The potential load on the environment due to food systems has been discussed for years. In the existing research (Shindo et al., 2010; Oda, 2006) on food systems, nitrogen accumulation in Japan is attributed to increase in food import, increase in chemical fertiliser input, and changes in food consumption patterns. Life cycle thinking plays an important role in understanding the comprehensive flow of energy, carbon, water, and other important materials associated with the systems for food production/consumption and biomass (food waste) utilisation. Such evaluation is required for estimating the comprehensive effect of future changes in consumption patterns, biomass policy, and agricultural technologies.

In this study, the material flow associated with food systems, including waste treatment and recycling, is evaluated. In particular, we focus on the energy flow, including some indicators that reflect the energy effectiveness of food systems and the utilisation level of biomass and food waste.

The energy flows investigated in this study are related to the following types of energy:

1. Cumulative non-renewable energy
2. Feedstock energy related to food and biomass
3. Nutritionist's calories of food

In this study, we evaluated 47 commodities of food. They were evaluated on the basis of the following indicators:

a. Effectiveness of utilisation of food waste and co-products (EF-w)

$$(EF - w) = \frac{EP + AE - EC + FSE_2}{FSE_1}$$

Here, FSE_1: Feedstock energy before treatment/utilisation of food waste and co-products; *EP*: Energy production via utilisation of food waste/co-products; *AE*: Alternative effect of material recycling; *EC*: Energy consumption during treatment/utilisation of food waste and co-products; FSE_2: Feedstock energy after treatment/utilisation of food waste and co-products.

b. Energy effectiveness of food supply/consumption (EF-f)

$$(EF - f) = \frac{FE}{TI - EP - AE}$$

Here, *TI*: Cumulative non-renewable energy used; *FE*: Total energy intake for a food system

The data pertaining to material flow and food transportation are collected from statistics, reports, and experts. The data pertaining to energy consumption during food production are collected from papers or calculated from production-cost data.

Fig. 1 shows the material flow of food products in Japan in 2005. The amount of input from crop farming to animal farming is roughly the same as that to that from food manufacturing. The flow from food manufacturing to animal farming denotes the flow of rice and wheat bran. The internal flow related to crop farming consists of the incorporation of residues. The comprehensive approach employed in this study can be used to evaluate the effect of active utilisation (for example, bioethanol production or composting of straw) by considering alternatives for organic materials used in farmlands and future change in grain consumption, which are necessary for deciding long-term policies.

References

Shindo J., Okamoto K., Kawashima H., Konohira E, 2010. Nitrogen flow associated with food production and consumption and its effect on water quality in Japan from 1961 to 2005. Soil Sci. Plant Nutr., 55(4), pp. 532-545

Oda K., 2006. Trends in nitrogen flow since the 1980s in the food and feed system in Japan. Jpn. J. Soil Sci. Plant Nutr., 77, pp. 517-524 (in Japanese with an English abstract)

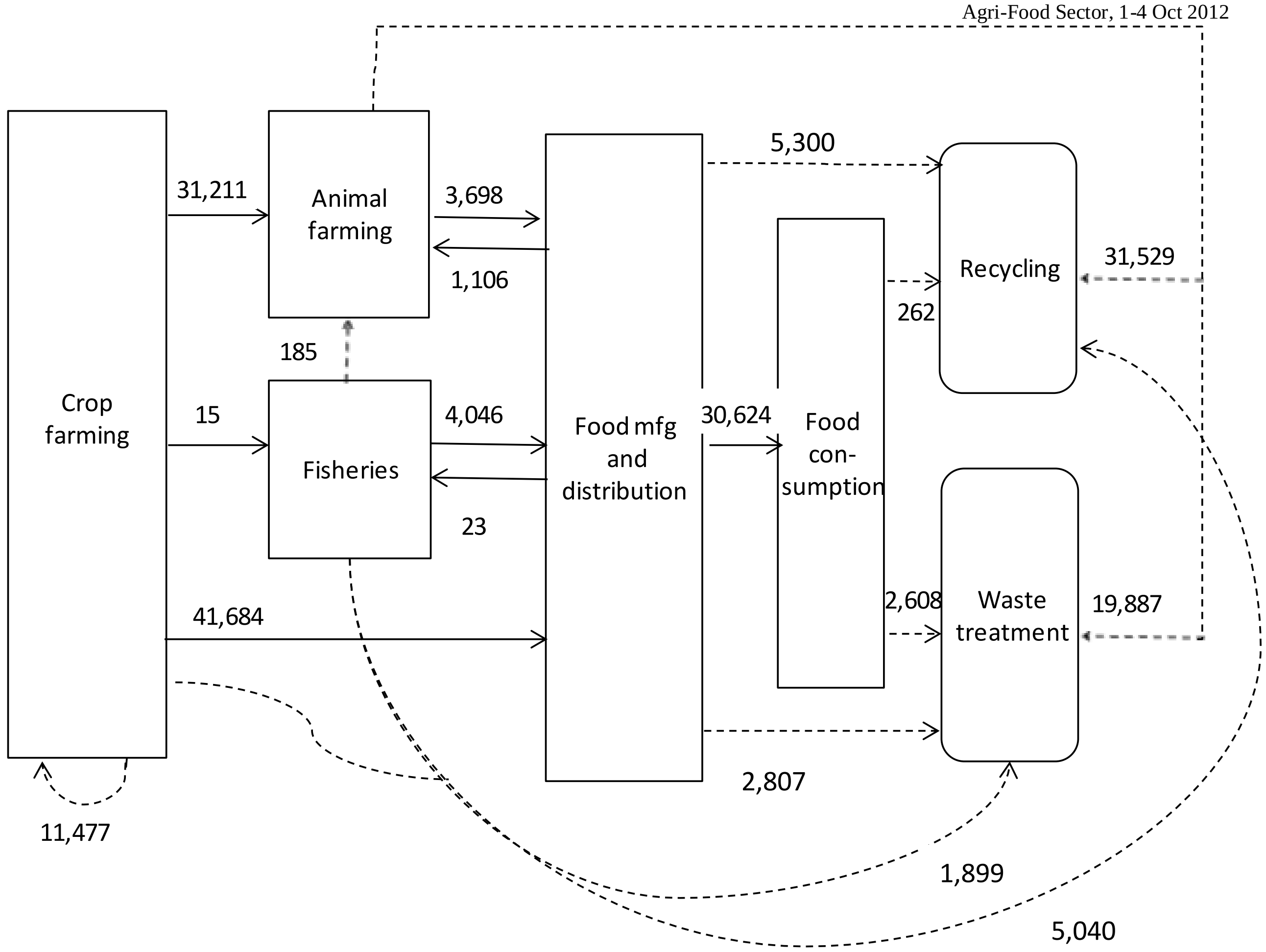


Figure 1. Material flow associated with food production and consumption in Japan in 2005 (Unit: thousand tonnes, wet).

137. Environmental impacts of pasta cooking

Luca Ruini[1], Roberto Bassi[1], Massimo Marino[2], Sonia Pignatelli[2,*]

[1] Barilla G. R. Fratelli S.p.A, Italy, [2] Life Cycle Engineering, [*] Corresponding author. E-mail: *pignatelli@studiolce.it*

Sustainability has now entered in the agendas of companies, policy makers and a fraction of "green consumers". Companies work to achieve environmental impact reduction, while policy-makers work to define strategies aimed at improving the sustainability of production and consumption chains. Consumers are asked to prefer products that demonstrate compliance with these requirements.
Though the production phase always seems to be the most important in terms of environmental impacts, in some cases the consumer use bears even more impact than production itself: pasta falls into this case and, therefore, this paper presents elaborations aimed to the calculation of the environmental impacts of the cooking phase.
The starting point is constituted by a full life cycle assessment of Barilla's pasta production that was published in a verified Environmental Product Declaration in which the carbon, ecological and water footprint were illustrated in utmost clarity.
Aside from this, a detailed study on the cooking impacts was made, and the carbon footprint of different cases evaluated. Normally pasta makers recommend using 10 times the water in comparison to the amount of product being cooked: 500 g should therefore use 5 litres of water. It is interesting to consider how the use of different amounts of water can affect energy consumption and the relative impacts in terms of CO_2 equivalents.
At this point, it is interesting to examine how the various environmental impacts vary in relation to the amount of water used for cooking. The diagram below represents the impacts for both a smaller and a larger amount of water used for cooking 500 grams of pasta; some considerations were made changing the quantity of water used to cook pasta, moving from 4 to 6 litres This abstract also accounts for the Italian energy-mix for electricity production. Data about energy production and use come from ecoinvent database.
It is interesting to assess how a variation of the quantity of water used yields significant differentiations of impact: -20% water corresponds to -7% GHG emissions for gas cooking procedures (Fig. 2).
The carbon footprint of the pasta cooking phase is similar to that of production. That is why correct consumer behaviour is as important as corporate efforts aimed at reducing impacts. In particular, it is quite important to use the right amount of water, cover the pot while waiting for the water to boil, and add salt only when the water is boiling. This aspect of the cooking phase is directly linked to the consumer's behaviour. That is the reason why proper consumer information is crucial for achieving sustainability, learning how to avoid useless waste and be more environmentally friendly.

References

Baldo, GL, M. Marino, S. Rossi, 2008. Analisi del Ciclo di Vita LCA, nuova edizione aggiornata, Edizioni Ambiente.

Barilla, 2011. Environmental Product Declaration of durum wheat semolina dried Pasta in paperboard box (brand Barilla), Published on the International EPD system register. www.environdec.com (Rev. 2.1; approved the 10/03/2011, Reg. S-P-217).

Barilla Center for Food and Nutrition (BCFN), 2011. 2011 Double Pyramid: Healthy Food for People, Sustainable for the planet, Supporting technical paper Version 2 of 14 July 2011.

Ecoinvent Swiss database, v.2.0, www.ecoinvent.ch

Foster C., Green K., Blea m., Dewick P., Evans B., Flynn A., Mylan J., 2006. Environmental impacts of Food Production and Consumption – a Report to the Department of the Environment, Food and Rural Affairs (DEFRA), Manchester Business School, London.

Nilma, Italian manufacturer data sheets, www.nilma.it

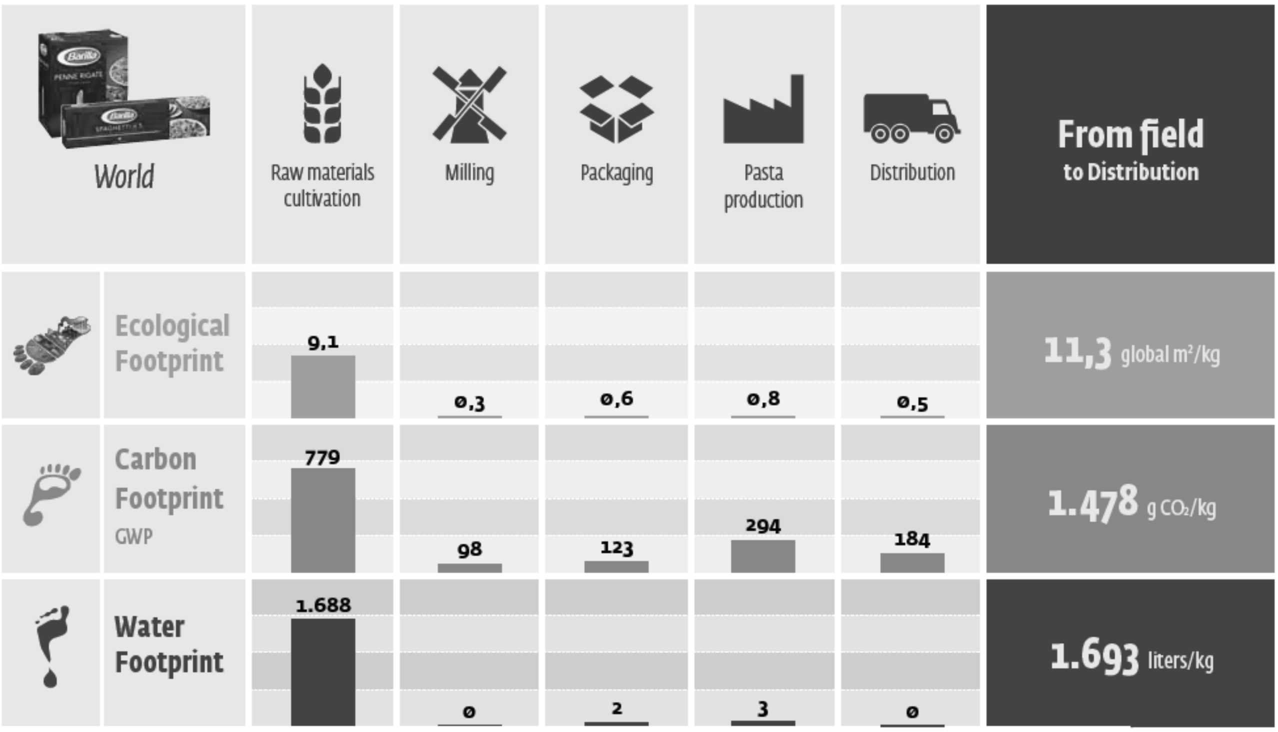


Figure 1. Footprint of pasta production (Barilla, 2011)

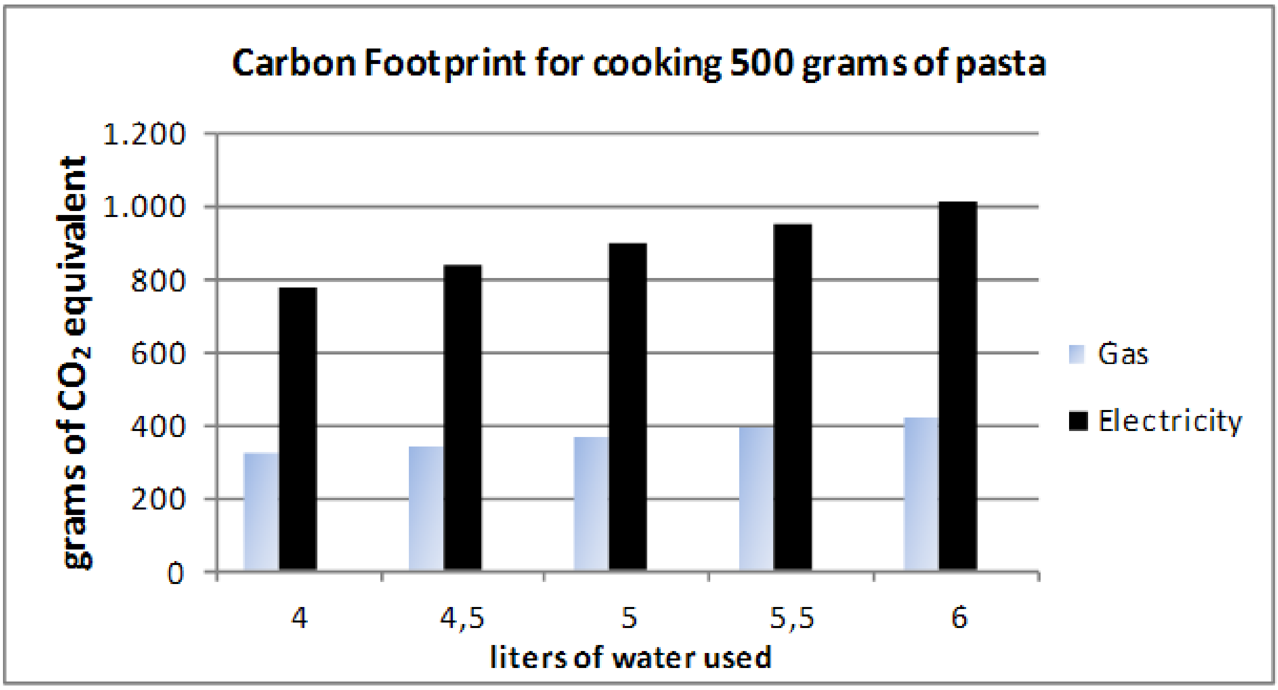


Figure 2. Carbon footprint for cooking 500 grams of pasta assuming a variable pasta/water ratio of ± 20% and a cooking time of 10 minutes. A total of 5 litres of water were used as recommended by the producer (BCFN, 2011).

138. The development of robust Life Cycle Analysis for mycoprotein and the meat free brand Quorn™

Wayne Martindale[1,*], Louise Neville[2], Muyiwa Akintoye[2], Tim Finnigan[2]

[1] Centre for Food Innovation, Sheffield Hallam University, Sheffield, S1 1WB UK, [2] Quorn Foods Ltd, Stokesley, UK, [*] Corresponding author. E-mail: *w.martindale@shu.ac.uk*

The global food supply system is experiencing increased protein demand pressures that are dependent on feed and livestock production systems that have received much attention in LCA research (FAO, 2006). Several internationally led assessments of wider impacts of livestock production raise the importance of Indirect Land Use Change (ILUC) associated with production and trade in livestock products. An area that has received less attention and one where we believe we are ready to report robust LCA data is that of industrially produced proteins (Finnigan et al., 2010 and Toumisto 2010). We believe that industrially produced proteins offer significant benefits to the current world protein supply system that are currently unrealised by meat producers, food manufacturers and policy makers.

To define the potential of meat free ingredients we have developed a LCA based approach to defining the production impact of the mycoprotein ingredient for the Quorn™ brand of food products that is retailed in 22 countries. Mycoprotein is produced industrially from the fungal fermentation of wheat derived glucose in the United Kingdom. An LCA programme within the mycoprotein and Quorn™ manufacturing facilities provided GWP measurements of 3.1 for mycoprotein and typically 4.2 for Quorn™ products (see **Table 1**). This initial LCA provided important targets for future investigation within the Quorn™ supply chain. These were (1) energy balance and the use of co-product steam in Quorn™ manufacture from nearby ammonia fixation plant, and, (2) the use of Egg White Protein (EWP) in the manufacture of Quorn™ from mycoprotein. A further outcome, was a more detailed investigation of the Quorn™ ingredient supply chain in the terms of embodied GHG's and energy.

Development and improvement of the existing LCA has identified GWP reductions of at least 30% over a three year period committing to the company to significant investment in LCA based on the very clear business case that implementing LCA procedures improves production efficiencies and identifies cost reduction. Furthermore, Quorn Foods Ltd has aligned current methodologies with the Carbon Trust Footprint Expert LCA Model. This approach has further detailed knowledge of the mycoprotein and Quorn™ supply chain in terms of embodied resources and environmental impact associated with the product. It has further identified the reality of fixing LCA boundaries around a brand that has over 90 discrete Stock Keeping Units (SKU's).

We show that aligning commercial and marketing information with a international standard such as PAS2050 is still in a developmental stage. There is a requirement to develop applied statistical methods so that companies can obtain typical data for supply chains that are not just 'snapshots' but represent integrative data of supply chains over realistic commercial time periods accounting for production, waste and proportion of product consumed by consumers. This is critical to food supply chains and others where there are seasonal changes in the LCA of ingredients and selective consumption of specific parts of products. We present research that defines our approach in developing the functional unit of initial LCA of 1 tonne of mycoprotein to a 300g of retail product purchased. In achieving this we have identified important considerations for the global protein production system where industrially produced proteins have a critical role to play in optimising land use.

References

FAO, 2006. Livestock's Long Shadow. ISBN 9789251055717

Finnigan T., Lemon T., Allan B., and Paton I., 2010. Mycoprotein, Life Cycle Analysis and the Food 2030 challenge. Aspects of Applied Biology 102, 81-90.

Tuomisto H., (2010). Food security and protein supply- cultured meat a solution? Aspects of Applied Biology 102, 99-104.

Table 1. The protein and eco-system service attributes of mycoprotein compared to wheat and beef

Protein source	Protein g/100g	GWP	Land use
Wheat	12.7	0.80	0.53
Beef	22.5	15.80	3.44
Mycoprotein	11.0	3.11	0.53

139. Environmental evaluation of French olive oil production

G. Busset[1,2,3,4,*], JP Belaud[3,4], F. Clarens[5], J. Espi[5], M. Montréjaud-Vignoles[1,2], C. Sablayrolles[1,2]

[1] Université de Toulouse, INP-ENSIACET, LCA (Laboratoire de Chimie Agro industrielle), Toulouse, France, [2] INRA, UMR 1010 CAI, Toulouse, France, [3] Université de Toulouse, INP-ENSIACET, LGC (Laboratoire de Génie Chimique), [4] CNRS UMR 5503, Toulouse, France, [5] CTM Centre Tecnològic, Manresa, Spain, [*] Corresponding author. E-mail: *guillaume.busset@ensiacet.fr*

Olive oil production is an important agro-industrial sector in the Mediterranean region of Europe, which has to face environmental issues. The OiLCA (2011) research program has been elaborated partly in order to reduce the carbon footprint and to optimise waste management of the olive oil sector in SUDOE area (Spain, Portugal and France). However, French production is different when compared to Spanish or Portuguese because it is more traditional. The present work proposes a life cycle assessment of the French olive oil production sector.
The study is led in accordance to the ISO 14040 (2006) and ISO 14044 (2006) standards. Functional unit is the production of one litre of olive oil. Transport of workers, olive tree culture, harvest, cold extraction, solid residues (leaves, branches and grounds) management, wastewater management, storage and bottling are the main identified steps of the system. Valorisation of waste is accounted like avoided impacts of a product with the same function. Matter and flow data come from 21 French enterprises contacted for the OiLCA project. All indirect extraction and emissions is calculated with the ecoinvent database. Then, impact assessment is realised with the impact methods Impact 2002+ and ReciPe 2008. The SimaPro® software is used for the impact calculation.
First results have permitted to define all the scenarios for olive oil production in France based on the different olive production techniques (with or without irrigation, mechanical or not, organic or not), the different extraction processes (pressing, centrifugation two phases or centrifugation three phases) and the different waste management (incineration or spreading). Raw data from enterprises have been collected. Expected results are the comparison of all the scenarios in order to identify parameters that influence environmental consequences of olive oil production.

References

OiLCA, 2011. Enhancing the competitiveness and reducing the carbon footprint of the olive oil sector through waste management optimization and the establishment of an ecological label, http://www.oilca.eu/index.htm

140. LCA of food packaging: soy-protein based films versus PP film

I. Leceta[1], P. Guerrero[1], S. Cabezudo[2], K. de la Caba[1,*]

[1] University of the Basque Country, UPV/EHU. Chemical and Environmental Engineering Department. Polytechnic School. Donostia-San Sebastián. Spain, [2] University of the Basque Country, UPV/EHU. Business Organization Department. Polytechnic School. Donostia-San Sebastián. Spain, [*] Corresponding author. E-mail: *koro.delacaba@ehu.es*

Food packaging protects food against external environment to maintain its quality during storage and extend its shelf life. Conventional packaging plastics impact heavily on the environment because of its end of life and, furthermore, they are derived from non-renewable sources, so biobased materials could be considered friendlier for the environment than petroleum-based materials. In this context, soy proteins could be a potential replacement of conventional polymers because they are abundant, renewable, and biodegradable. Moreover, soy protein-based films can be processed to exhibit good mechanical and barrier properties for food packaging (Guerrero et al., 2011), using techniques such as extrusion (Guerrero et al., 2012), which are employed for industrial applications.

In this study, a comparative life cycle assessment was carried out between two different food packaging systems: a commercial food packaging film based on polypropylene (PP) and a new biodegradable soy protein-based film manufactured in our labs. The functional unit was 1 m^2 of packaging film. Three main stages were considered: resource extraction, film manufacture, and waste disposal. The data relating to PP packaging films were obtained from Ecoinvent v2.0 database. The life cycle inventory for soy protein production was taken from literature and film manufacture data was measured directly by our research group in the lab. The selected method for comparison of the films was EcoIndicator 99. The disposal scenario considered for soy protein-based biofilm was composting, due to the fact that the film is based on natural raw materials, while different waste scenarios were studied for conventional PP film: landfilling, incineration, and recycling.

As seen in Fig. 1, impact categories in which the biofilm exhibited a significant environmental charge were carcinogens, respiratory inorganics, climate change, land use, and fossil fuels. The responsible for the main impact in carcinogens and the high environmental burden associated to land use category was the cultivation of soybeans and the use of pesticides, fertilisers, diesel, and machinery associated to this cultivation. The environmental load in respiratory inorganics and climate change categories were principally owing to the emissions from wood burning and clear-cutting for land transformation, and the fuel consumption in this land transformation was the main cause of the impact in fossil fuels. Regarding to PP film, films disposed in landfill exhibited higher impact in carcinogens. Emissions of carbon dioxide, hydrocarbons, methane, and volatile organic compounds (VOCs) generated by waste treatment in landfilling were the main responsible for this impact (Sundqvist, 1999). In addition, the environmental burden in fossil fuels impact category was also very high for landfilling. On the other hand, when the end of life was incineration, the film showed high impact in climate change and fossil fuels categories, being raw materials extraction the stage that originated this environmental burden. Recycling waste scenario showed the lowest impact due to the fact that raw material was considered as avoided product in the recycling of the film, so that the emission associated to the extraction of PP was much lower than in the emission associated to the other end of life scenarios.

References

Sundqvist, J.O., 1999. Life cycle assessments and solid waste – Guidelines for solid waste treatment and disposal in LCA. Final Report. IVL, Swedish Environmental Research Institute.

Guerrero, P., Beatty, E., Kerry, J.P., de la Caba, K., 2012. Extrusion of soy protein with gelatin and sugars at low moisture content. Journal of Food Engineering, 110, 53-59.

Guerrero, P., Stefani, P.M., Ruseckaite, R.A., de la Caba, K., 2011. Functional properties of films based on soy protein isolate and gelatin processed by compression molding. Journal of Food Engineering, 105, 65-72.

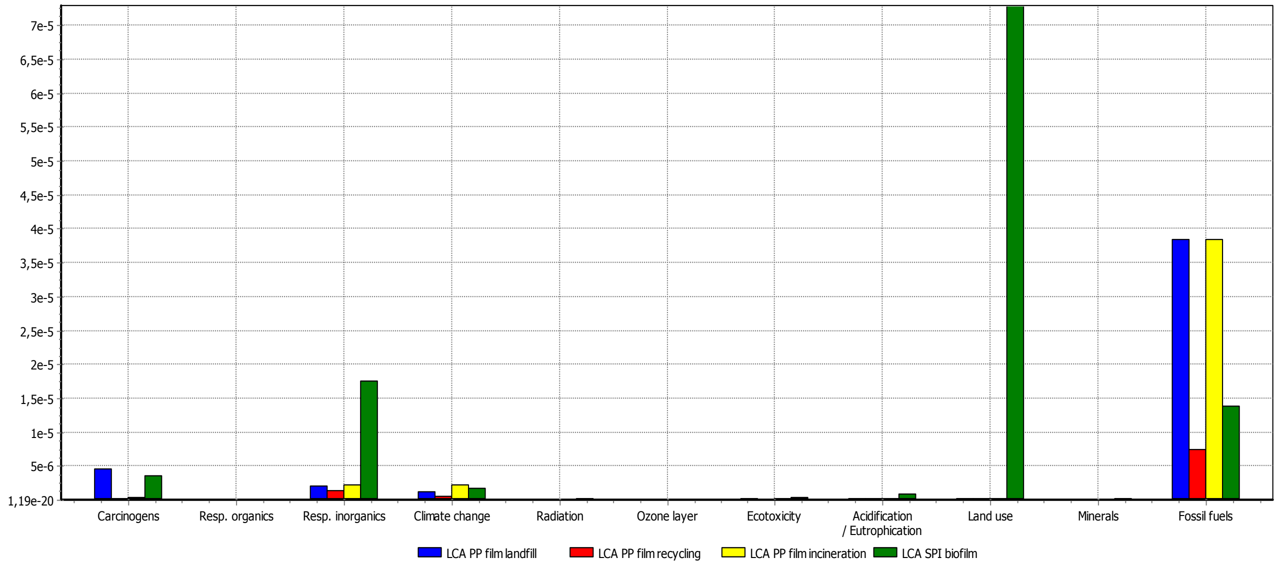


Figure 1. Normalised results of the comparative environmental assessment.

141. Comparative LCA of reconstituted milk and recombined milk processing

Farroudja Younsi[1,*], Krim Louhab[2], Chawki Belblidia[3]

[1] Laboratory of ESTs, Development, Physical Chemistry of Materials and Biodiversity, Boumerdes University, Algeria, [2] Food Technology Research Laboratory, Boumerdes University, Algeria, [3] Pasteurized Milk Production Department LFB Boudouaou, Algeria, [*] Corresponding author. E-mail: *younsi_umbb@yahoo.com*

The Algerian dairy industry operates mainly on the basis of milk powder and anhydrous milk fat (AMF). Technologically, two transformation processes allow us to obtain pasteurised milk: reconstitution and recombination. The first process involves rehydrating the whole milk powder while in the second process, the finished product is obtained from a mixture of reconstituted milk, based on skim milk powder and AMF.
This study, carried out in a dairy processing situated in Boudouaou (Algiers), aims to provide a comparative analysis of environmental impacts generated by these two processes. The approach used here is based on the life cycle assessment (LCA). This powerful and recognised method, standardised in the International Organization for Standardization (ISO 14040 to 14043) is used in practice.
Only the most significant impacts were considered in this study: Respiratory inorganic, terrestrial ecotoxicity, terrestrial acidification, land occupation, global warming and non-renewable energy consumption. The results obtained demonstrate the advantage of the recombination process compared to the reconstitution of milk. Indeed, the addition of AMF in the transformation process has reduced the impact by a factor of 3 to 6%. These results can be explained by the substitution in the scenario 2 a volume (5 tons) of milk powder by the AMF and also by its nature. Since it is a co-product, the impacts attributed to the AMF are negligible compared to those assigned to the milk powder, which is the main product. In the light of the obtained results, it seems more optimal to favour the recombination of milk-based AMF.

References

Cederberg, C., Mattsson, B., 2000. Life cycle assessment of milk production- a comparison of conventional and organic farming. Journal of Cleaner Production 8, 49-60.

Feitz, A.J, Lundie, S., Dennien, G., Morain, M., Jones, M., 2007. Generation of an industry-specific physico-chemical allocation matrix. Application in the dairy industry and implication for systems analysis. International Journal of LCA 12 (2) 109-117.

Høgaas Eide, M., 2002. Life Cycle Assessment (LCA) of Industrial Milk Production. International Journal of LCA 7: 115-126.

Hospido, A., Moreira, M.T., Feijoo, G., 2003. Simplified life cycle assessment of galician milk production. International Dairy Journal 13, 783-796.

Jensen, G.K., Nielsen, P.M. (1982). Milk powder and recombination of milk products. Reviews of the progress of Dairy Science. Journal of Dairy Research 49, 515-544.

Thomassen, M.A., Van Calker, K.J., Smits, M.C.J., Iepema, G.L., De Boer, I.J.M.(2008) Life Cycle Assessment of conventional and organic milk production in The Netherlands. Agric Syst 96:95–107.

Table 1. AMF contribution to different impact categories. (for 1 kg of milk)

Impact category	Unit	Scenario1	Scenario2	Difference	%
Respiratory inorganic	kg PM2.5 eq	6.93E-04	6.67E-04	2.64E-05	3.82
Terrestrial ecotoxicity	kg TEG soil	2.38E+00	2.22E+00	1.59E-01	6.67
Terrestrial acidification	kg SO_2 eq	6.39E-02	6.13E-02	2.58E-03	4.04
Land occupation	m^2org.arable	4.33E-01	4.15E-01	1.77E-02	4.10
Global warming	kg CO_2 eq	5.22E-01	4.96E-01	2.65E-02	5.07
Non-renewable energy	MJ primary	3.84E+00	3.62E+00	2.18E-01	5.67

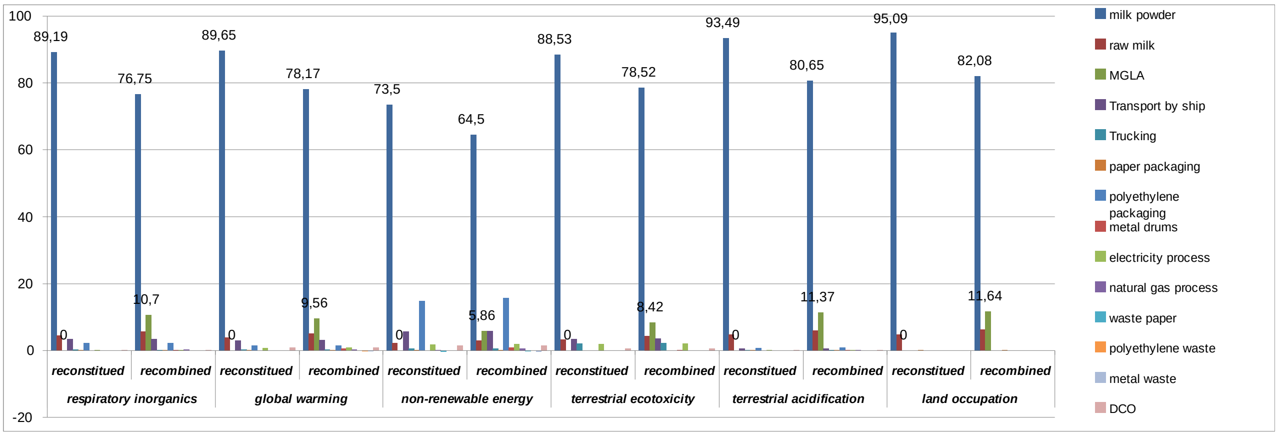


Figure 1. Contribution of the two processes to various impacts

142. Improving the efficiency of agricultural and food systems by introducing biodegradable and compostable products in strawberry supply chain: experience from northern Italy

Francesco Razza[1,*], Francesco Degli Innocenti[2], Cristiana Peano[3],Vincenzo Girgenti[3], Michele Bonous[3]

[1] Novamont, Piazz.le Donegani 4, 05100 Terni – Italy, [2] Novamont, Via G. Fauser 8, 28100 Novara, Italy, [3] Dipartimento di colture arboree Università degli Studi di Torino, Via Leonardo DaVinci 44, 10095 Grugliasco (TO), Italy, [*] Corresponding author. E-mail: *francesco.razza@novamont.com*

Bioplastics can be used to make biodegradable products for agriculture. Biodegradability can contribute to alleviating the waste problem of this sector. The FRUITGEST project aimed to increase the efficiency of strawberry supply chain by introducing biodegradable products at different levels of the supply chain itself. The first task interested the strawberry agricultural phase. Here the traditional mulch film made of traditional plastic (non-biodegradable) is replaced with the biodegradable one. By doing so the waste passes from 260 kg per ha up to zero. The second task was focused on the development of innovative strawberry packaging systems (i.e. flowpack). Aim was to increase the shelf life of the product, thus decreasing food losses, and to increase the efficiency of waste management at supermarket level. As a matter of fact, the packaging and its content (i.e. expired strawberry) are suitable to be collected along with organic fraction without any operations since the packaging is compostable according to EN13432. These changes in the strawberry supply chain were assessed by means of LCA using a consequential approach. Results have shown that the most important benefits were those related to the increase of shelf life of the product; this avoids wasting food (i.e. strawberry). Furthermore, also the use of biodegradable mulch film and packaging has positive consequences. Basically this project has demonstrated that, in particular circumstances and applications, the use of biodegradable bioplastics is beneficial, especially for the effects in waste treatment systems.

143. LCA of one cup of espresso coffee: how to collect and validate LCA data along the coffee supply chain

Stefania Furfori[1,*], Cristiano Portis[1], Alberto Cabilli[1], Alfredo Vanni[1], Leo Breedveld[2], Rosalina Barrale[3]

[1] Lavazza SpA, [2] 2B Srl, [3] Goglio Cofibox SpA, [*] Corresponding author. E-mail: *s.furfori@lavazza.it*

The management of life-cycle inventory data is a complex and time-consuming task in LCA, especially due to upstream and downstream processes outside the company's system boundaries. The data collection is often hampered by the complexity of these processes, a lack of involvement of supply chain stakeholders and associated confidentiality issues. Being a key-player in the coffee supply chain Lavazza SpA, in collaboration with one of its suppliers, has attempted to overcome these barriers as illustrated in the LCA of espresso coffee.

The LCA of espresso coffee has been conducted in order to support Lavazza's ecodesign activities, a main part of the corporate social responsibility strategy and in order to create a truthful and correct environmental communication. Espresso coffee is a rather elaborated food system with a diversified supply chain, involving coffee plantations around the whole world (e.g. Brazil, India, Vietnam, Ethiopia), green coffee suppliers, packaging suppliers (especially Goglio Cofibox SpA), the coffee manufacturer (Lavazza SpA), the distribution chain, consumers and finally waste disposal management.

A crucial moment of the LCA execution has been the choice of Lavazza and Goglio Cofibox to work together, which has proved to be a successful partnership. The handling of the complexity of the multi-layer packaging is guaranteed by the Goglio Cofibox's packaging experts, while Lavazza is in the position to involve other actors in the supply chain.

The data used in the LCA comes directly from coffee manufacturers and suppliers. Energy consumption and greenhouse gases emissions impact categories are screened using IPCC 2007 and CED assessment methods, following the international standards ISO 14040 (2006) and ISO 14044 (2006). The results, consistent with other studies published on coffee (Humbert et al., 2009), show that about half of the environmental footprint occurs at a life cycle stage under the control of the coffee producer or its suppliers (coffee cultivation, treatment, processing, packaging up to distribution) and the other half at a stage controlled by the user (appliances manufacturing, use and waste disposal).

The success of this project depends on the company's ability to able to increase the awareness and commitment of the key players involved in the entire supply chain in supporting the improvement solutions. The validation and yearly updating of LCI data is also made easier in this respect, transforming initial barriers into mutual opportunities to understand the potential ecological footprint impact of each phase and to enhance the environmental market position of all chain stakeholders.

References

Humbert , S., Loerincik, Y., Rossi, V., Margni, M., Jolliet, O., 2009. Life cycle assessment of spray dried soluble coffee and comparison with alternatives (drip filter and capsule espresso). Journal of Cleaner Production 17, 1351-1358.

ISO 14040, 2006. Environmental Management – life Cycle Assessment – Principles and Guidelines. International Organisation for Standardisation (ISO), Geneva, Switzerland.

ISO 14044, 2006. Environmental Management – life Cycle Assessment – Requirements and Guidelines. International Organisation for Standardisation (ISO), Geneva, Switzerland.

144. Life cycle assessment of an artisanal Belgian blond beer

Raphaëlle Melon[1,*], Roberto Renzoni[1], Angélique Léonard[1], Vincianne Wergifosse[2]

[1]Laboratory of Chemical Engineering, University of Liège, 17 allée de la Chimie, 4000 Liège, Belgium, [2] Brasserie des Légendes, 19 rue du Castel, 7801 Irchonwelz, Belgium, * Corresponding author. E-mail: *raphaelle.melon@ulg.ac.be*

In the framework of the Wal-Aid project funded by Wallonia aiming namely at developing valorisation means for co-products of the agro-food industry, a life cycle assessment applied to the production of artisanal Belgian blond beer of ‘ Brasserie des Légendes’ was carried out.
This analysis focuses on the production of a golden triple beer and the packaging step. The functional unit is the production of 1 litre of beer and the used method is ReCiPe endpoint. This production is carried out in six steps: (1) culture of barley, malting, grinding and transport of malt flour at the brewery, (2) brewing (heating and mixing) of malt flour and water in two steps, heating at 62-63°C during 70-90 minutes, then 73°C for 10-15 min. (3) Boiling wort at 105°C during 2h30 with steam generator. Hop is added to wort but only transport of this material is considered. (4) Then beer is kept at 4°C during 25 days, (5) packaging in brown glass bottle of 33 cl with a crown cap chrome steel. The mass of steel capsule is estimated at 2 g and 0.3 g of label’s paper. (6) Washing of spent grain recovered after brewing with H_2SO_4.
Most used data in this analysis come from the industrial site except for the production of barley malt and packaging. Information about these steps are taken from the literature. Generic data come from the ecoinvent v2.2 database (www.ecoinvent.org). Analysis was performed using SIMAPRO 7.3.2 software from PRé Consultants.
Electricity consumptions were calculated on the basis of energy balances and Belgian energy mix of 2008 released by the International Energy Agency. This mix shows that Belgian electricity is mainly produced from nuclear energy (57%), gas (31%) and coal (9%).
Figure 1 shows the percentage contribution of environmental impact for production’s steps of one litre beer in bottles. The most penalising step is glass packaging with 74.1%; then we find the preparation stage of malt flour with 19.8%. This stage represents culture of barley, malting, malt grinding and its transport at the brewery. Washing of spent grain step does not appear in this table because its impact can be considered as negligible.
Figure 1 shows the single score results of the production of one litre of beer and compares this production with and without packaging in glass bottle. First, we note that the packaging step greatly increases the overall impact of production process. Secondly, whatever the system boundaries, i.e. with or without packaging, the most important impacts are fossil depletion and climate change human health due to the energy demand of packaging step for the production of glass bottles and energy for growing, grinding and transporting malt flour. Impact of agricultural land occupation remains the same whatever the system boundaries as the packaging step is primarily energy consumption and no culture is associated.
For the rest, climate change ecosystems, particulate matter formation and human toxicity are consequences of use of fossil resource to produce electricity and fuel.
This analysis shows that the most penalising step in the production of this artisanal beer is the packaging. So, it is on this step we have to work to reduce the environmental impact of this beer. In this study, we consider a packaging in brown glass bottles and it seems interesting to analyse a packaging of 20 litres in a keg and compare the results.
Acknowledgments: The authors gratefully acknowledge the Walloon Region of Belgium for the financial support of Wal-Aid Wagralim project.

References

Kloverpris, J. H., N. Elvig, et al. (2009). Comparative life cycle assessment of malt-based beer and 100% barley beer, Novozymes A/S, Harboes Bryggeri.

Table 1. Environmental contribution of production stages.

Step	Contribution (%)
Production of malt flour	19.8
Brewing	0.1
Boiling wort	3.9
Keep at 4°C	2.1
Glass packaging	74.1

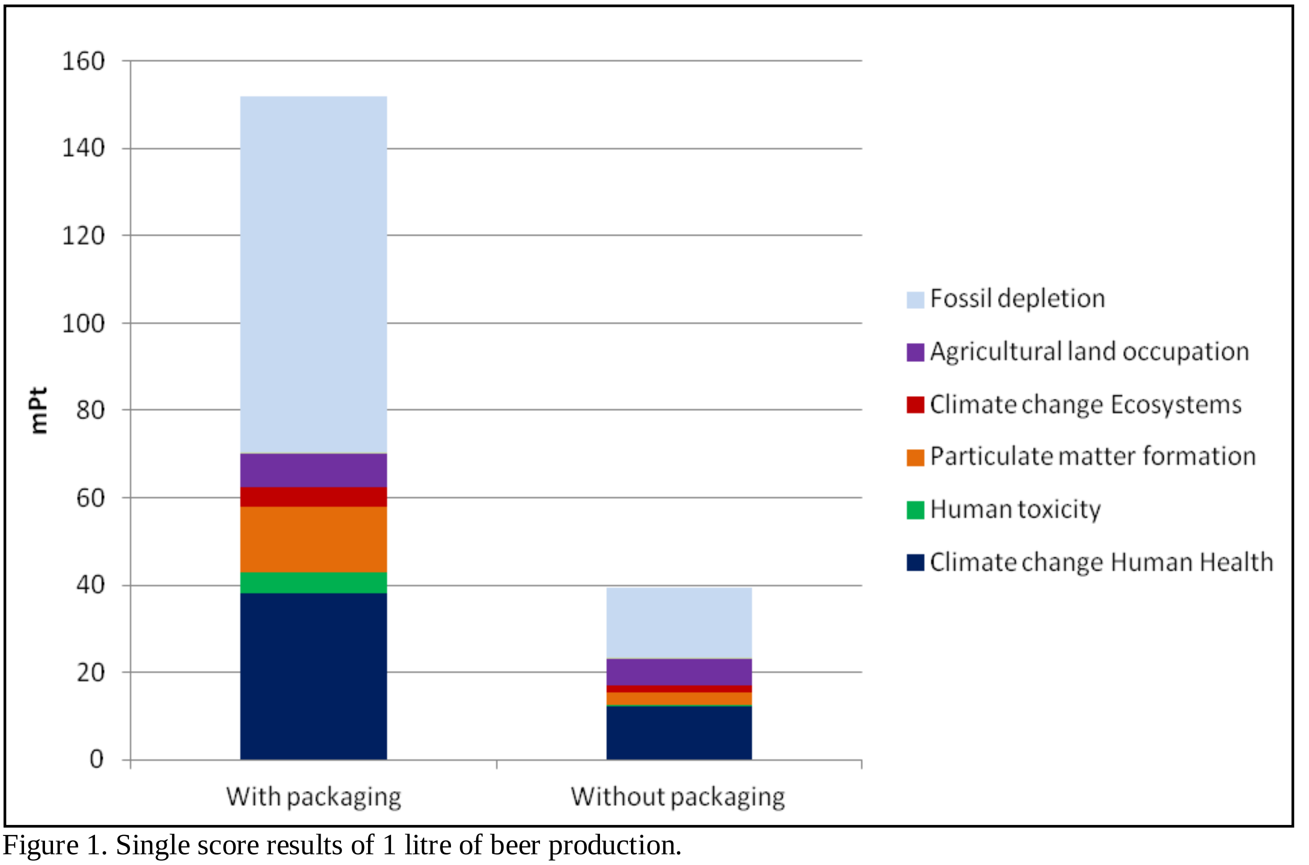


Figure 1. Single score results of 1 litre of beer production.

145. Tracking environmental impact of food production chain: a case study on fresh carrot chain in Italy

Simona Bosco[1,*], Giorgio Ragaglini[1], Giuseppe Garcea[2], Roberta De Natale[3], Enrico Bonari[1]

[1] Institute of Life Sciences, Scuola Superiore Sant'Anna, Pisa, Italy, [2] CCPB Srl, Bologna, Italy, 3 Auchan SpA, Rozzano Milanofiori, Italy, * Corresponding author. E-mail: *s.bosco@sssup.it*

It is recognised that food product supply chains contribute to the environmental impact of goods production and consumption. In the last years, an increasing number of food chain partners and public authorities have promoted many initiatives to provide information about the environmental performance of food products (Peacock et al., 2011), with an initial interest focused only on the carbon footprint analysis (Hillier et al., 2009; BSI 2011). Adoption of differences approaches, methodologies and objectives could confuse consumers and stakeholders, while scientifically defensible information concerning food production and food product and production systems are required by policymaker and producers (Schau and Fet 2008).

Here we proposed a new method specific for food products and production chains eco-labelling based on LCI built on modular phases integrated within questionnaires. The food chain system boundary was defined so as to include four main phase: agricultural phase, processing and packaging, delivery to platform and to retail store. Consumer phase was not included, except for waste management. The LCI considered all the inputs, both energy and material, involved in the production process, and the outputs for each phase of the food chain, including all transports until the supermarket. Special attention was devoted to the agricultural phase, for all field operations data were tracked in a field trial notebook and all the direct field emissions were included. This method was applied to a complex food product chain of a large-scale retail trade company. The examined production was fresh carrot within a real case study in Italy, using an approach similar to that of food traceability. 1 kg fresh carrot packed in a plastic tray was chosen as functional unit (FU), and the production chain was organised in farm production, processing and packaging in Abruzzo Region, delivery to distribution platform in Lombardy Region and to one single retail store located in the same region (Fig. 1). The data collection was referred to 2009; primary data were collected by questionnaires, filled out by farmers or technicians responsible of each phase. Secondary data were from GaBi embedded database and literature (GaBi, 2012). CML 2001 method was used for impact assessment for seven main impact categories (IC): Global Warming (GWP), Acidification (AP), Eutrophication (EP), Freshwater Aquatic Ecotoxicity (FAETP), Terrestric Ecotoxicity (TETP), Photochem. Ozone Creation (POCP), Human Toxicity (HTP). Results presented in Fig. 2 show that packaging and storing phases were responsible of the largest impact level in each considered ICs, mainly due to the estimated electricity consumption for refrigeration. It is important to notice that also the distribution at platform, rarely included in the literature, could significantly affect the results in some IC, 33% in FAETP and 27.5% in HTP. On the contrary, the cultivation phase showed low impact levels in average, with exception for EP, where it was responsible for the 39% of the estimated impact. The proposed approach was able to describe and analyse the impact of a real food production chain. The questionnaires allowed building a simple and clear method for a detailed data collection. This approach helped to develop a standardised system boundary for food products and subsequently to carry out robust and easily comparable LCA analyses.

References

BSI, 2011 PAS 2050:2011 - Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. BSI British Standards, London.

GaBi, 2012. GaBi software. http://gabi-software.com

Hillier, J., Hawes, C., Squire, G., Hilton, A., Wale, S., Smith, P., 2009. The carbon footprints of food crop production. Int J Agric Sustain, 7(2), 107-118.

Peacock, N., Camillis, C., Pennington, D., Aichinger, H., Parenti, A., Rennaud, J.-P., et al., 2011. Towards a harmonised framework methodology for the environmental assessment of food and drink products. Int J Life Cycle Ass, 16(3), 189-197.

Schau, E. M., Fet, A. M. 2008. LCA Studies of Food Products as Background for Environmental Product Declarations Int J Life Cycle Ass, 13, 255-264.

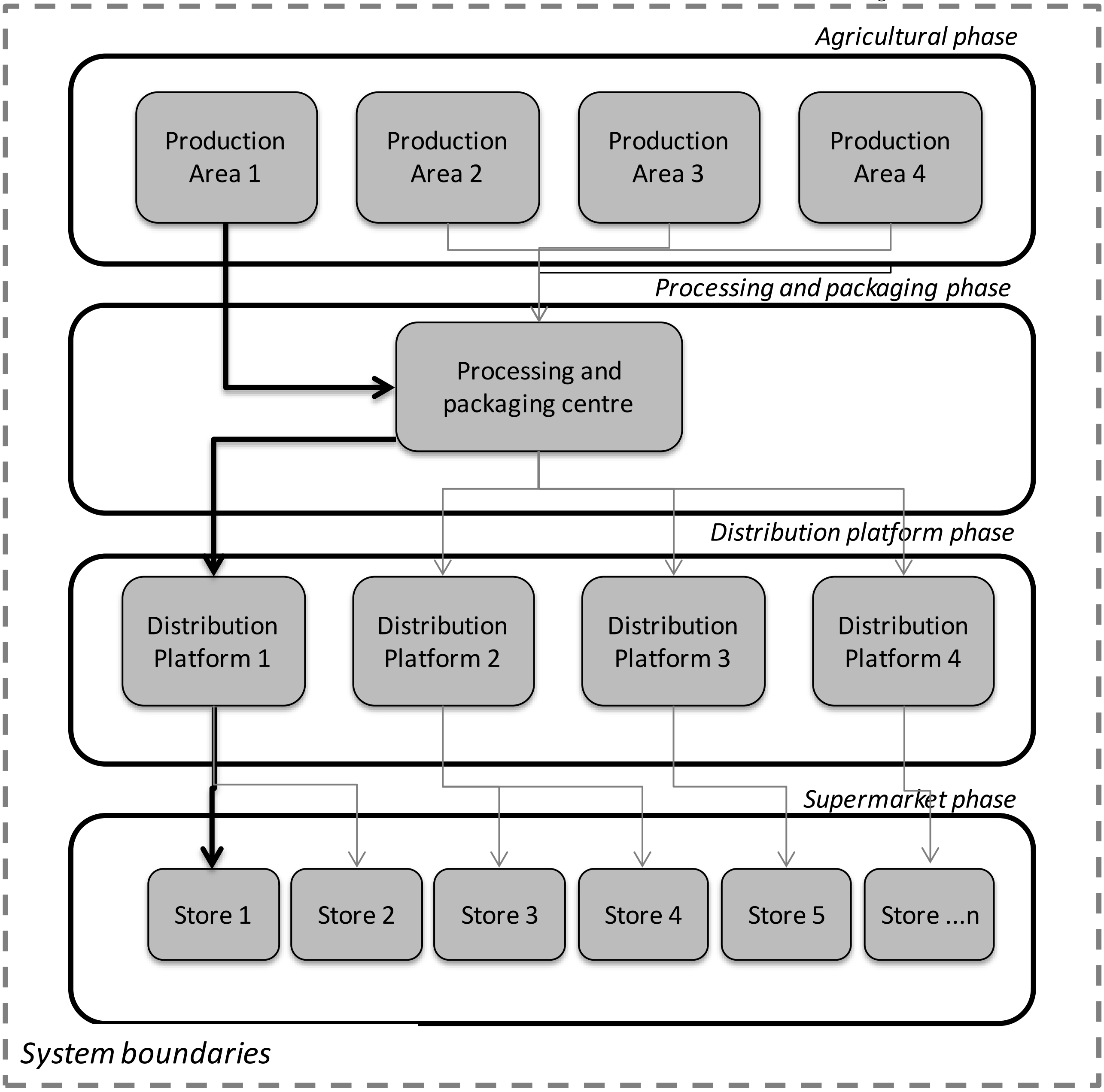


Figure 1. System boundaries of 1 kg fresh carrot plastic tray production chain. The black arrow identifies the system analysed.

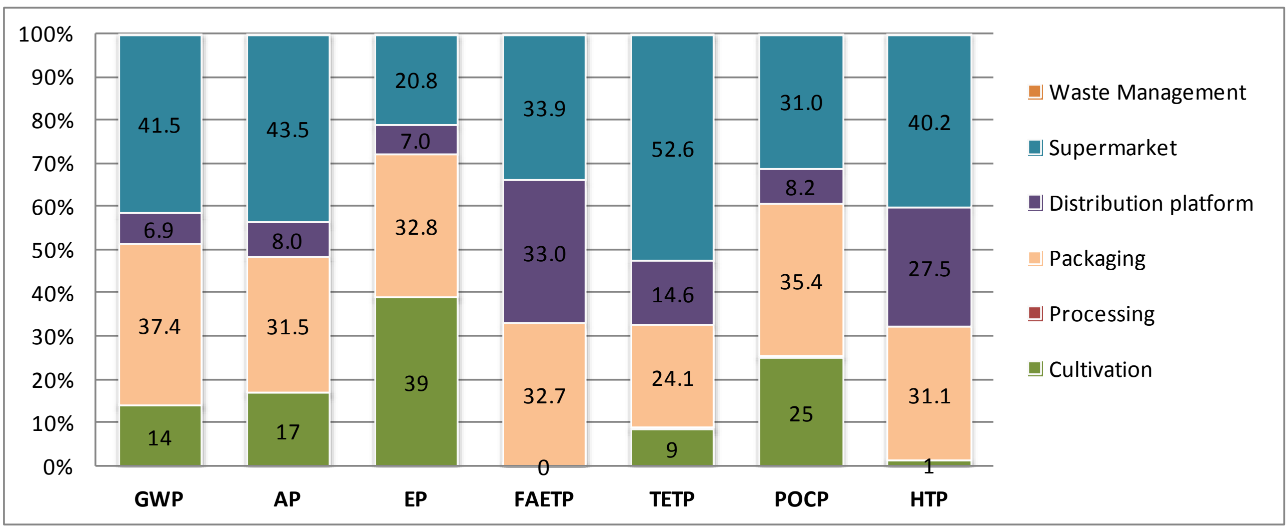


Figure 2. Impact assessment for 1 kg fresh carrot plastic tray for food chain phase.

146. Life cycle assessment of an Italian sparkling wine production

Paola Masotti[1,*], Luigia Petti[2], Serena Melchiori[3]

[1] Department of Economics, University of Trento, Via Inama 5, 38122 Trento, Italy, [2] DASTA Department, "G. d'Annunzio" University, 65127 Pescara, Italy, [3] University of Trento, Via Inama 5, 38122 Trento, Italy [*] Corresponding author. E-mail: *paola.masotti@unitn.it*

The environmental impacts of food products and methods to identify and minimise those impacts are drawing the attention of producers. Wine producing firms are particularly interested in these topics in order to demonstrate that along with the quality of the product they are also committed in developing a proper environmental management of the production chain. To meet the requirements of an important Italian wine producer the environmental performance of a sparkling wine made by a winery of a Trentino province, has been investigated.

This paper presents the results of the a life cycle assessment (LCA) applied to an Italian sparkling wine from vineyard operations, winery activities from grape crushing to bottling, and to packaging.

The main objective of the study was to quantify input flows of raw materials and energy in order to detect the environmental impacts associated with output flows and releases at any stage, from cradle to gate, of wine to evaluate the environmental performance. Furthermore, quantification of environmental impacts enables the identification of the hot spots of the activities and stages and the definition of management practices to improve environmental performance.

Foreground data were collected from Cantina Rotari (Gruppo Mezzacorona) and refer to production year 2005. Background data were derived from different databases such as EcoInvent and Corinair. The inventory table was constructed for one bottle of wine (0.75 L) using the LCA software SimaPro and data were assessed using the Eco-Indicator 99 method.

The characterisation result graph shows that bottle production contributes to all the 11 impact categories considered with high percentage values for Ecotoxicity (88%) and Acidification (55%). A further step to better understand the magnitude of the category indicator results is normalisation, which shows (Fig. 1) that fossil-fuel extraction is the prominent impact category being bottle production and pesticides the major contributors.

As regards the single process contribute to the three damage categories (Human Health, HH; Ecosystem Quality, EQ; Resources, R) glass production is the first process for both HH and EQ whereas crude oil utilisation is the first process for R followed by glass production.

Finally, the Ecoindicator single score assessment (Fig. 2) shows the impact of each process to all the damage categories being the glass production and the crude oil utilisation the ones that more contribute to the damage categories considered.

This LCA study pointed out that in this specific sparkling wine production there are two processes that mainly affect the overall environmental impact: the bottle production that accounts for 36% of the total impact and the pesticides production and utilisation that account for 27.3%.

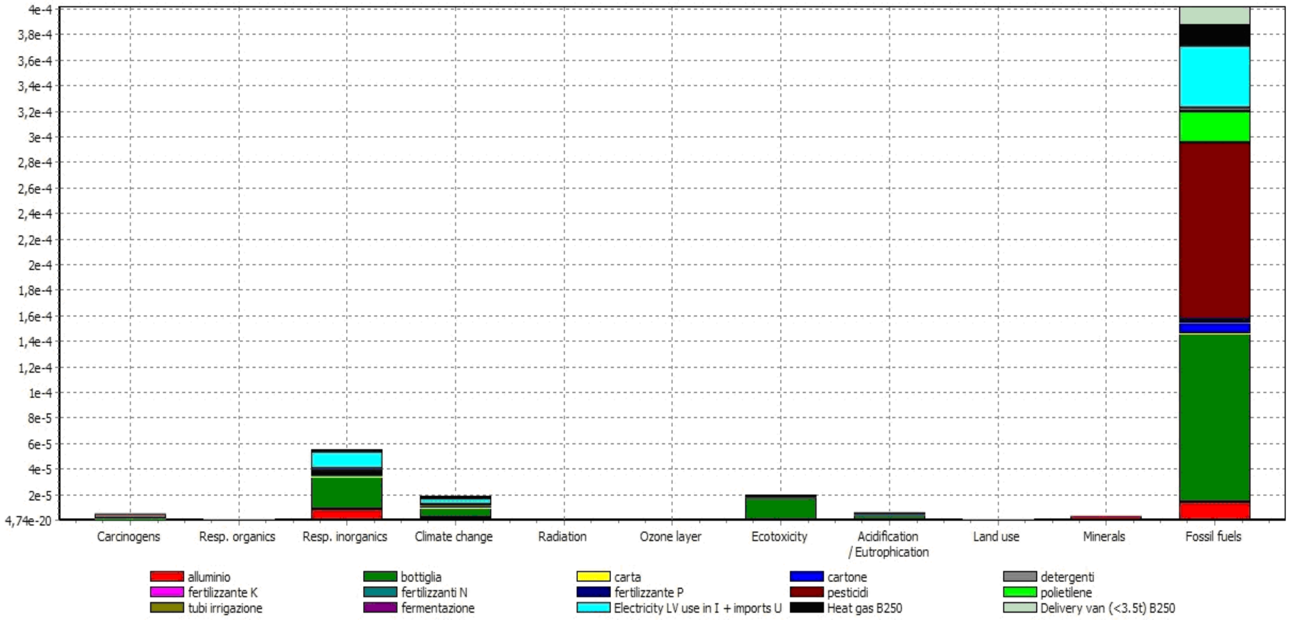


Figure 1. Sparkling wine production – Impact assessment normalisation.

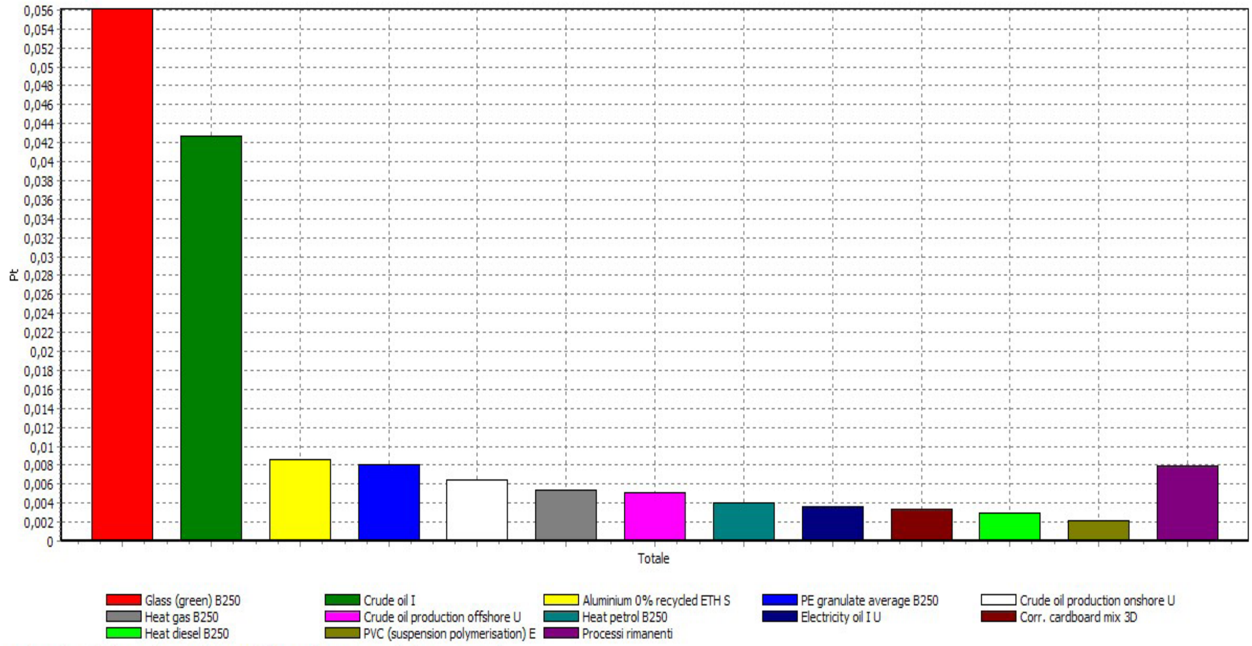


Figure 2. Sparkling wine production - Process contribution, single score.

147. Influence of transport on environmental impacts of the production chain of poultry in Brazil

Sebastião Roberto Soares[1,*], Vamilson Prudêncio da Silva Junior[2], Edivan Cherubini[1], Marco Antônio Santos[3]

[1] Universidade Federal de Santa Catarina, Florianópolis, Brazil, [2] Empresa de Pesquisa Agropecuária e Extensão Rural de Santa Catarina, [3] Eng. Agrônomo - Especialista em Gestão Ambiental Agroindustrial, [*] Corresponding author. E-mail: *soares@ens.ufsc.br*

Brazil is among the five largest exporters of poultry meat (Burnquist et al., 2011, USDA, 2011) solidifying as one of the majors exporter of livestock products, Brazilian's production is benefited by favourable climatic conditions and its vast territory, besides the high government investment in the sector. These production chains, such as poultry, pigs and cattle, in particular, are able to produce all raw materials that it's used in animal feed. That feature brings interesting advantages for Brazilian companies when evaluated the environmental impacts of its products since livestock production is responsible for a major impact on the environment as noted by de Vries and de Boer (2010), so the choice for a more sustainable product can mitigate some of these impacts. Although the chain can be self-sufficient in production of grains, the large distances of a continental country between the stages in the life cycle added to the lack of kinds of transportations may eventually supplant that environmental gain. Thus the aim of this paper is to evaluate from the environmental point of view, different kinds of transportation and their influences on the productive chain of frozen chicken for export. We evaluated two scenarios: (A) Poultry produced and exported in South with the grains coming from the Central West; and (B) Poultry and grains produced in Central West with the exportation placed in South East. The functional unit is a ton of poultry delivered in the port for exportation. The method used through the software SimaPro® was the CML 2000 with modifications, from where we choose by the easier level of communication with the stakeholders, the impact categories of global warming and total cumulative energy demand. The results of the LCA of the usual scenarios, indicates South Poultry as that with the worse environmental performance, due the transportation of grains. For the category of global warming the difference between the South and the Central West, in tons of CO_2 equivalent emitted, only from the transportation of grains and the final product are about 49.6%, with the possibility of this difference decrease 19%.

References

Burnquist, H. L., da Costa, C. C., de Souza, M. J. P., Fassarella, L. M., 2011. Composite Index of Market Access for the Export of Poultry from Brazil. International Centre for Trade and Sustainable Development (ICTSD), Geneva, Switzerland.

United States Department of Agriculture (USDA), 2011. Livestock and Poultry: World Markets and Trade. Foreign Agricultural Service.

de Vries, M., de Boer, I.J.M., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livestock Science 128, 1-11.

148. A methodological comparison of Life Cycle Sustainability Assessment of fruit and vegetable logistics

Pere Fullana i Palmer[1,*], Alba Bala[1], Tabea Beck[2], Stefan Albrecht[2]

[1] UNESCO Chair in Life Cycle and Climate Change, Escola Superior de Comerç International (Universitat Pompeu Fabra); Passeig Pujades 1, 08003 Barcelona, Spain, [2] Department Life Cycle Engineering, Chair of Building Physics (University of Stuttgart); Hauptstrasse 113, 70771 Echterdingen, Germany, [*] Corresponding author. E-mail: *pere.fullana@esci.upf.edu*

For packaging fruit or vegetables, wooden boxes, cardboard boxes and plastic crates are most commonly used. While the first two are non-returnable packaging systems and normally disposed of or partly recycled after one use, plastic crates as a rule are returnable packaging, washed and reused many times.

The environmental impacts of the European wide fruit and vegetable distribution have been analysed by different institutions in different countries during the last decade. Some studies have analysed different transport packaging systems, but their understanding of the important life-cycle aspects and the sensitivity of the parameters within the life-cycles of the different packaging solutions is not homogeneous.

In this review, we analyse mainly the reports of three studies performed by Ecobilan (2000), ITENE (2005) and University of Stuttgart (2009). It is very interesting to note the differences from the commissioning organisation to how the study has been communicated. In between, methodological differences can be found in relation to each methodological step. Differences can be found in: the goal and scope of the study; how sustainability is addressed; the assumptions to key parameters (such as number of rotations of plastic crates, from 10 to 100); how complex and close to reality is logistics taken (from single trips to international networks); how open-loop recycling is calculated; which combination of data sources is used, the treatment of biogenic CO_2; the extend of the sensitivity analysis or the inclusion of different scenarios and, in fact, how the interpretation phase of LCA is addressed; who and how performed a critical review; etc.

As per the results and the ones from the Fenix project, it is relevant to see that LCSA does not give a unique and static answer. While the study in 2000 showed that the multiple-use option was environmental preferable to the cardboard format for most of the environmental impact categories assessed and quite similar to the wooden study, the study in 2005 showed the opposite, i.e. that the environmental impact of single use cardboard boxes is always lower than that of reusable plastic in six of the 10 categories analysed. Finally, the study in 2009 shows a very similar result (with not totally equal numbers) than the first one.

Will a new study arise soon to balance the match scores? Is this a game or is it a serious matter with millions at stage? How should scientists respond to market pressure? Is LCA still being used as a throwing weapon nowadays? We are looking forward to a “fruit”-full discussion.

References

Ecobilan, 2000 (in French). Life cycle assessment of wood, corrugated board and plastic crates for apples.

ITENE, 2005 (in Spanish). A comparative study of the environmental and economic characteristics of corrugated board boxes and reusable plastic crates in the long distance transport of fruit and vegetables, Universitat Politècnica de València.

University of Stuttgart, 2009 (in English). The Sustainability of Packaging Systems for Fruit and Vegetable Transport in Europe based on Life-Cycle-Analysis – Update 2009.

149. Comprehensive life cycle assessment for cheese and whey products in the U.S.

Greg Thoma[1,*], Darin Nutter[2], Rick Ulrich[1], Dae-Soo Kim[1], Greg Norris[3], Franco Milani[4], Dean Sommer[4]

[1] Ralph E. Martin Department of Chemical Engineering, University of Arkansas, Fayetteville, AR, USA, [2] Department of Mechanical Engineering, University of Arkansas, Fayetteville, AR, USA, [3] Walton College of Business, University of Arkansas, Fayetteville, AR, USA, [4] Department of Food Science, University of Wisconsin – Madison, Madison, WI, USA, [*] Corresponding author. E-mail: *gthoma@uark.edu*

A comprehensive life cycle assessment (LCA) has been carried out to determine a benchmark for the environmental impacts associated with cheddar and mozzarella cheese consumption in the United States. This includes specifically product loss at various stages of the supply chain, as well as consumer transport and storage of products. The scope of this study was a cradle-to-grave assessment with particular emphasis on the unit operations of typical cheese processing plants.

A functional unit of 1 metric ton of cheese consumed (dry solids basis), or 1 metric ton of whey delivered (dry basis) was adopted. The ecoinvent database was used for 'upstream' unit processes. Operational data from 17 cheese manufacturing plants representing 24% of mozzarella production and 35% of cheddar production in the US. Allocation procedures follow the ISO 14044 hierarchy. System separation was used when sufficient information was available, primarily as plant-specific engineering estimates. Incoming raw milk, cream or dry milk solids are allocated using a milk solids mass balance. Revenue-based allocation was used for remaining in-plant processes.

Life cycle impact assessment was conducted using the ReCiPe and USEtox frameworks. Greenhouse gas (GHG) emissions are of significant interest, and on a dry milk solids basis, the carbon footprint of both cheddar and mozzarella is approximately 13.0 metric tons CO_2e (IPCC 2007 factors) per metric ton of cheese consumed (inclusive of product loss across the supply chain). The 95% confidence interval ranges from 9 to 18 metric tons CO_2e per metric ton of cheddar cheese (dry solids basis) consumed. For an average solids content of 63.2% for cheddar as sold at retail, the cumulative GHG emissions are 8.5 kg CO_2e per kg cheddar cheese consumed, and for an average solids content of 51.4% for mozzarella, the GHG emissions are 7.3 kg CO_2e/kg consumed. Fig. 1 shows the relative contribution from different supply chain stages to both the cradle to grave impacts and the post-farm gate impacts (that is impacts excluding milk).

This study provides a benchmark for the US cheese manufacturing industry to gauge improvement over time and showed that energy use, especially electricity, across the supply chain is relevant to several impact categories, including climate change, cumulative energy demand, photochemical oxidant formation, and human toxicity (USEtox). The impacts to ecosystems (ReCiPe) are driven almost exclusively by agricultural land occupation (63%) while the aquatic ecotoxicity (USEtox) impacts are driven significantly by fly control pesticides which are used in dairy operations and to lesser extent crop protection chemicals. Improvement opportunities focused on reducing energy consumption will have broad beneficial impacts, both economic through cost savings, as well as environmental due to the emissions reduction.

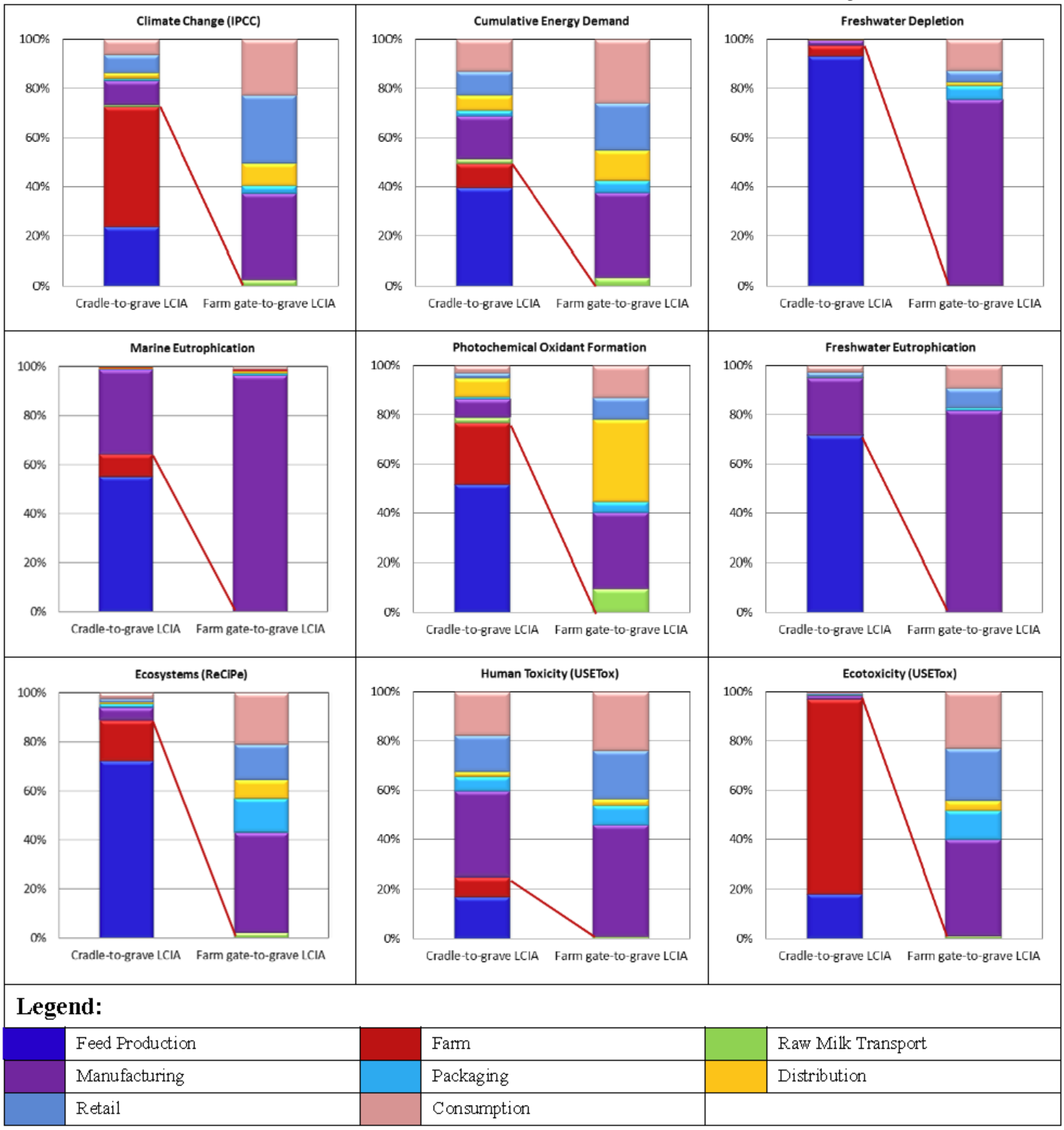


Figure 1. Summary of cradle-to-grave and farm gate-to-grave life cycle impact assessment results for cheddar cheese supply chain.

150. Comprehensive life cycle assessment of fluid milk delivery systems

Greg Thoma[1,*], Darin Nutter[1], Jasmina Burek[1], Dae-Soo Kim[1], Susan Selke[2], Rafael Auras[2], Bev Sauer[3], Sarah Cashman[3]

[1] University of Arkansas, Fayetteville, AR, USA, [2] Michigan State University, East Lansing, MI, USA, [3] Franklin Associates, Kansas City, MO, USA, [*] Corresponding author. E-mail: *gthoma@uark.edu*

In 2009 U.S. Dairy Management Inc. has set up a target to reduce greenhouse gas emissions of fluid milk supply chain of 25 percent by 2020, as a part of U.S. Dairy Sustainability Commitment. The focus of this research is the LCA of fluid milk delivery systems emphasising assessment of environmental impacts of 18 fluid milk packaging and delivery options. The study provides results of the life cycle impact assessment for various standard and emerging fluid milk packaging options.

The objective of this project was to conduct a cradle-to-grave LCA of fluid milk delivery systems focused on quantifying use of non-renewable energy sources, emissions to air, water, and land, consumption of water and other natural resources; and assessment of the impacts of these inventory flows on climate change, resource depletion, human health, and ecosystems. The LCA provides data for the dairy industry enabling the industry to identify and engage in more sustainable approaches and identify opportunities for improvements leading to mitigation of environmental impacts across the dairy delivery life cycle.

The main goal of this work was to equip milk delivery industry stakeholders (milk processors, packaging material manufacturers and retailers) with timely, science-based information in order to incorporate environmental performance into decision-making and drive innovative new products, processes, and services. Fluid milk delivery systems can be distinguished by their final consumption, delivery type, container material composition and size. The life cycle impact assessment methods chosen include: ReCiPe Midpoint, ReCiPe Endpoint, and USEtox. They were used to create results that include the relevant inventory indicators and range of midpoint/impact and endpoint/damage categories. The selection includes two inventory indicators: ReCiPe's Water depletion [m^3] and Cumulative Energy Demand Non-renewable, fossil [MJ]. For the purpose of result interpretation and clarification of the importance of certain impact category in the context of dairy delivery systems, each system was analysed using normalisation step of the IMPACT 2002+ Method for U.S., and World ReCiPe normalisation.

A summary of the fluid milk delivery systems under study based on their final consumption function, delivery option, container composition, size, and total weight is presented in Fig. 1. Fig. 2 presents the overall results for gallon sized HDPE container based delivery, as this represents approximately 65% of fluid milk consumption in the United States.

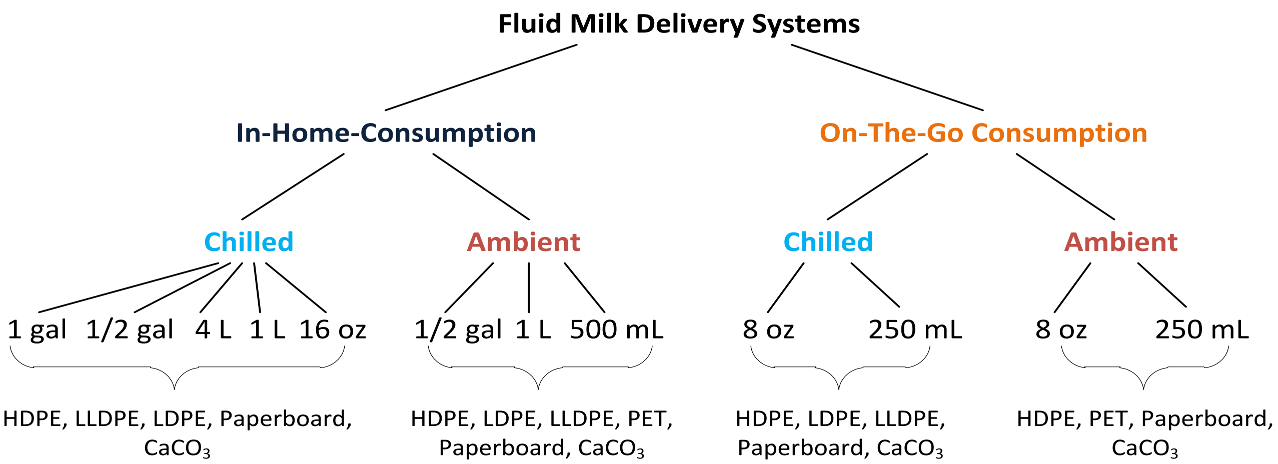


Figure 1. Summary of fluid milk delivery systems under study.

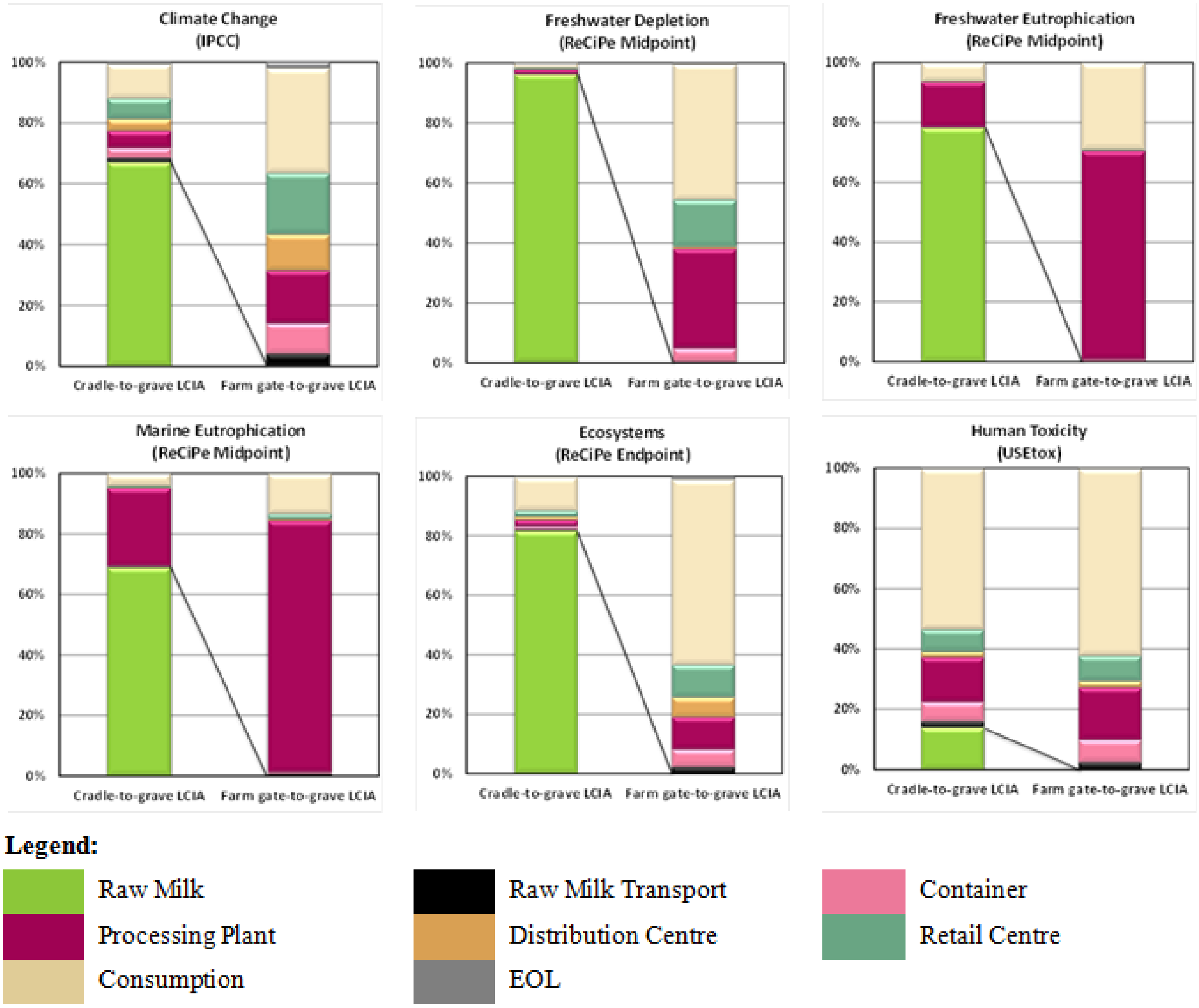


Figure 2. Cradle-to-grave and farm-gate-to-grave life cycle impact assessment results [%] of mono-layer HDPE fluid milk delivery system for nine impact categories

151. AusAgLCI – building national lifecycle inventory for Australian agriculture

Sandra Eady[1,*], Tim Grant[2], Simon Winter[3]

[1] CSIRO, Locked Bag 1, Armidale NSW 2350, Australia, [2] Lifecycle Strategies, Swanston Street, Melbourne VIC 3000, [3] Rural Industries Research and Development Corporation, Canberra ACT 2600, Australia, [*] Corresponding author. E-mail: s*andra.eady@csiro.au*

There has been substantial growth in the development of Life Cycle Assessment (LCA) in agriculture in Australia and globally over the past five years. A recent survey of activity by University of Queensland identified more than 20 LCAs of Australian agricultural commodities; many of these sponsored by industry associations. However, the consistency of these studies is low in terms of their system boundaries, impact categories, modelling of co-products and data quality. There is a need to develop a national life cycle inventory (LCI) for agricultural products to support environmental impact studies and to provide data for important export commodities, so that importers of Australian agricultural commodities can access representative data to complete LCAs.

The Rural Industries Research and Development Corporation has initiated a project to develop an agricultural LCI database (AusAgLCI) in alignment with the Australian Life Cycle Assessment Database Initiative (AusLCI). The project began in December 2011 and will run for 20 months. The project will draw on the extensive research and data collection undertaken in Australia by industry research bodies, state departments of industry, universities and CSIRO. The first task will be to define important industry sub-sectors to give appropriate representation of agricultural products that reflect differences in environmental impact and market segments required by downstream users of the data. Using existing data, the project will align data quality, breadth of data coverage in terms of the impact categories included (global warming, water and land use, eutrophication, ecotoxicity) and standardise the documentation of inventories. Resulting unit processes from cradle-to-farm gate will be reviewed and published. As part of the project, approaches to key methodological issues are being resolved such as joint production from mixed farming systems, scope of water flows to be included, carbon fluxes and land use for future impact assessment developments. At the completion of the project priority areas for additional data collection will be identified.

The project will assess and develop tools to quickly and accurately determine flows to the environment and inputs from the techno sphere. These will explore the use of spatial data to inform inventory development, such as spatial layers for nitrogen and phosphors flows from agricultural land use and spatial layers for soil type for determining fuel consumption for land cultivation.

152. The French agricultural LCA platform

Anne Bouter[3,4,5], Armelle Gac[1], Aurélie Tailleur[2,*], Sandrine Espagnol[5], Aurélie Wilfart[3,4]

[1] Institut de l'Elevage, Monvoisin, F-35652 Le Rheu cedex, France, [2] ARVALIS-Institut du vegetal, Service Agronomie Economie Environnement, Boigneville, F-91720, France, [3] INRA, UMR1069 Sol Agro et hydro-système Spatialisation, F-35000 Rennes, France, [4] AGROCAMPUS OUEST, F-35000 Rennes,France, [5] IFIP - Institut du Porc, La Motte au vicomte BP35104, F-35651 Le Rheu cedex, France, [*] Corresponding author. E-mail: *a.tailleur@arvalisinstitutduvegetal.fr*

The French Agricultural LCA Platform has been created in 2009 under the impetus of the French technological mixed network on Livestock and Environment (Réseaux Mixte Technologique Elevage et Environnement). This network gathers research, technical and education institutes and aims at diffusing management tools for animal production systems to improve their environmental performances. This open LCA platform is gathering among 40 Agricultural LCA-practitioners from different institutes (agricultural research institutes, agricultural technical institutes, agriculture council institutes, professional organisations). The platform purpose is sharing collectively knowledge, data and methodological positions to manage LCA on agricultural products and systems, in taking into account French specificities of this sector. This includes actions to ensure common knowledge among the members by training organisation session and experiences sharing, to mutualise material resources (Using Simapro Multiuser license) and to validate collectively methodological choices (organisation of workshops about allocations, references for imported products in particular concerning land use, biodiversity and water impacts, and watch on international initiatives of mutualising LCA data). A specific action is to mutualise LCA and LCI data from 8 partners in a common database, and to work for the recognition of LCI data into international databases. This implies first collecting data among partners, describing precisely the methodology that has been involved, qualifying data quality and assessing their limits. This methodology will allow identifying the best references for users' purpose.

153. A software and database platform for multi-criteria assessment of agri-food systems

Hayo van der Werf[1,2,*], Julie Auberger[1,2], Coralie Haese[1,2], Geneviève Gésan-Guiziou[2,3], Joël Aubin[1,2]

[1]INRA, UMR1069 Sol, Agro and hydroSystem, 35000 Rennes, France, [2]Agrocampus Ouest, 35000 Rennes, France, [3] INRA, UMR1253 Science et Technologie du Lait et de l'Œuf, 35042 Rennes, France, [*] Corresponding author. E-mail: *hayo.vanderwerf@rennes.inra.fr*

INRA, the French National Institute for Agricultural Research, aims to improve the environmental, societal and economic performances of agriculture, and to favour healthy and sustainable food systems. Research conducted to reach these objectives should consider and integrate many criteria. The development of tools for the multi-criteria assessment of the production and transformation of agricultural products is of strategic importance, to guide the evolution towards sustainable agri-food systems. INRA has therefore decided to develop an in-house platform for the multi-criteria sustainability assessment of agri-food systems. The objectives of the platform are:

- To develop user-friendly multi-criteria assessment tools and associated databases for agri-food systems,
- To train future users, give support to users
- To provide a science and technology watch on multi-criteria assessment

Many methods have been proposed for the multi-criteria assessment of the environmental, societal and economic impacts of farming systems (van der Werf and Petit, 2002; Bockstaller et al., 2008). These methods present a similar structure, often consisting of eight stages (Acosta-Alba and van der Werf, 2011): 1) Definition of the sustainability dimensions to be assessed, 2) Identification of objectives for each dimension, 3) Selection or conception of indicators for each objective, 4) System definition, 5) Definition of calculation algorithms for each indicator, 6) Technical description of the system, 7) Calculation of indicator values, 8) Interpretation of results, identification of improvement options (Table 1).

In spite of their similar structure, these methods present an amazing diversity in their implementation, and the outcome of studies using different methods to assess contrasting systems depends to a large extent on the characteristics of the methods used (van der Werf et al., 2007). Methods differ amongst others with respect to system definition (inclusion of inputs to the system assessed), issues of concern considered, type of indicators (means-based vs effect-based) and expression of results (per ha or per kg of product) depending on identified functions.

Initially the platform will focus on the multi-criteria assessment of environmental impacts through Life Cycle Assessment (LCA). The implementation of societal and economic dimensions in methods for multi-criteria assessment will be pursued, in particular, but not exclusively, via the concept of Life Cycle Sustainability Assessment. The platform will allow the implementation of both LCA-type methods and other methods for the multi-criteria assessment of agri-food systems (e.g. Ecological Footprint, Emergy). This will be done by conforming the architecture of the platform to the eight stages of multi-criteria assessment methods outlined above (Fig. 1).The platform will be complementary to other INRA software platforms, and in particular to the RECORD platform (http://www4.inra.fr/record), for the modelling and simulation of crop and farm systems.

References

Acosta-Alba, I., van der Werf, H.M.G., 2011. The use of reference values in indicator-based methods for the environmental assessment of agricultural systems. Sustainability, 3, 424-442.

Bockstaller, C., Guichard, L., Makowski, D., Aveline, A., Girardin, P., Plantureux, S., 2008. Agri-environmental indicators to assess cropping and farming systems. A review. Agron. Sustain. Dev. 28, 139-149.

Van der Werf, H.M.G., Petit, J., 2002. Evaluation of the environmental impact of agriculture at the farm level: a comparison and analysis of 12 indicator-based methods. Agr., Ecosyst. Environ., 93, 131-145.

Van der Werf, H.M.G., Tzilivakis, J., Lewis, K., Basset-Mens, C, 2007. Environmental impacts of farm scenarios according to five assessment methods. Agr., Ecosyst. Environ., 118: 327-338.

Table 1. Stages of multi-criteria assessment methods for farming systems

	Stage	Description
1	Sustainability dimensions defined	Some methods only assess environmental impacts; other methods also assess societal and/or economic impacts and/or overall sustainability.
2	Objectives identified	Dimensions of sustainability cannot be directly quantified; a set of more specific objectives (issues of concern) for each dimension is required.
3	Indicators defined	To quantify the extent to which objectives are attained, indicators serving as assessment criteria are required. This stage may involve the definition of thresholds or reference values, which help to interpret indicator values.
4	System definition	The system is characterised, its boundaries and functions are defined.
5	Calculation algorithms defined	This stage involves the determination of calculation algorithms for the indicators. Calculation algorithms can be very simple (e.g. emission factors) or much more sophisticated (dynamic simulation models).
6	System technical description	Farmer production practices including amounts and timing of inputs used are described, technical parameters (feed efficiencies, crop yields) are quantified, pedoclimatic conditions are defined.
7	Indicators calculated	Indicator values are calculated for each of the systems or scenarios to be compared. A partial or total aggregation of indicator values may facilitate their interpretation.
8	Interpretation	Interpretation of results, identification of improvement options. This stage obviously is crucial; unfortunately few methods explicitly include this stage.

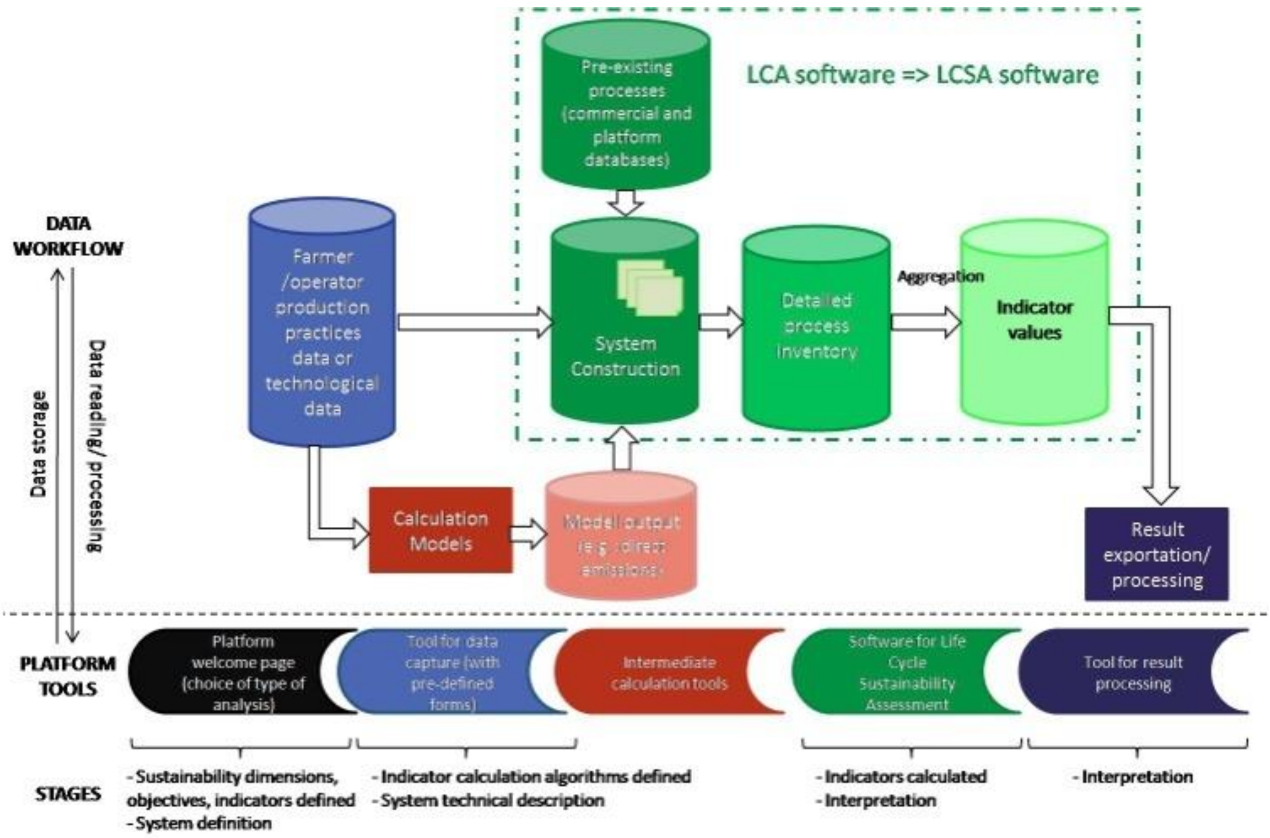


Figure 1. Representation of the platform's architecture and tools, allowing the implementation of different methods for the multi-criteria assessment of agri-food systems.

154. Availability and completeness of LCI of cereal products and related databases

Evelyne Laurent, Roberto Bellino, Stéphane Le Pochat*

EVEA, 56 Bd de la Fraternité, 44100 Nantes, France, * Corresponding author. E-mail: *s.lepochat@evea-conseil.com*

In 2011, EVEA carried out a study on behalf of Intercéréales (Inter-branch association of the French cereal sector). The aim of this study was first to analyse the different product labelling systems of cereals products around the world, and second to analyse available data in existing LCI databases, regarding cereals (wheat, barley, maize, sorghum, rice and rye) and cereal-containing products.

The study provided information on the existing databases that contain LCI of cereals and cereals-containing products. The analysis was conducted on ten French and international databases, eight of which include cereals-related data (ecoinvent, DiaTerre, LCA Food, Bilan Carbone®, AUSLCI, CPM Database, USLCI, and Agri-Footprint; Probas and BUWAL 250 do not include cereals-related data).

Each database was analysed by gathering general information (country, public or private status) and a description of the available data in the cereals sector. The following data are to be found in the databases: raw materials, inputs, processes, and cereal-based finished products (Table 1, Fig. 1).

The ecoinvent Database is by far the most complete database, with Swiss and European data for agricultural raw materials, inputs and processes. Data about some cereals-based finished products can be found in the LCA Food Database (wheat, bread, pastries, oat flakes) and in the French Bilan Carbone® database. However, very little data can be found in the databases about agricultural processes, food industry processes, storage or mass-market retailing, and the lack of data makes it difficult to implement an environmental labelling of cereal-based products based on present data available. Only ecoinvent Database and LCA Food Database provide specific geographic data: Swiss data in ecoinvent and Danish data in the LCA Food Database. The existing databases lack specific French data.

The study raised the issue of methodological comparability: all databases set their own hypothesis and methodological rules (allocation, cut-off rules…) and major differences can be found between data from different databases.

For example, the allocation method is not the same in all major databases. Among two of the main databases for cereals products, the ecoinvent database resorts to economic allocation whereas the LCA Food Database uses substitution (avoided impacts). As a consequence, for example, the impact of one kg of wheat, which is evaluated in 5 different databases, ranges from 0,401 kg CO2 eq. (USLCI) to 0,959 kg CO2 eq. (CPM Database), but this variability cannot be explained solely by obvious differences in yield, agricultural techniques or climate. This variability makes it difficult to implement an environmental labelling of cereal-based products with sufficient accuracy and comparability.

This study lead to the conclusion that existing databases cannot be used as a strong and detailed basis about cereal product environmental labelling. Two major programs have been launched in France in order to collect data for the environmental assessment of agricultural-based products:

- Agri-Balyse for the creation of French average data about the environmental impacts of agricultural productions.
- Acyvia about the impacts of agri-food transformation processes.

The conclusion of the Intercéréales study is that its members should take part in the Acyvia project in order to improve the completeness of the data about their products.

References

Abelmann, A. 2005. Environmental Potential of Increased Human Consumption of Grain Legumes: An LCA of food products. Report No. 2005:10.

Blonk, H. et al. 2011. The Agri-footprint method Methodological LCA framework - assumptions and applied data. Report Agri FootPrint

Nemecek, T., Kägi, T., 2007. Life Cycle Inventories of Swiss and European Agricultural Production Systems. Final report, Ecoinvent v2.0, No. 15a, September 2007.

LCA Food Database, 2007. www.lcafood.dk.

Table 1. Data availability for cereals and cereal-based products in 8 LCI databases

Database	**Country of origin**	**Generic or specific to the agriculture sector?**	**Agricultural raw materials**	**Fertilisers**	**Other agricultural inputs**	**Processes**	**Finished products (cereal-based)**	**Total number of data**
ecoinvent	Switzerland	Generic	21	25	110	3	1	160
DiaTerre	France	Specific	10	10	24	0	3	47
LCA Food	Denmark	Specific	10	7	5	0	11	33
Bilan Carbone®	France	Generic	2	8	3	0	9	22
AUSLCI	Australia	Generic	1	18	1	0	0	20
CPM Database	Sweden	Generic	1	8	0	0	3	12
USLCI	United States	Generic	3	2	0	0	0	5
AGRI FOOTPRINT	Netherlands	Specific	0	0	0	0	2	2
Total	48	78	143	3	29	301		

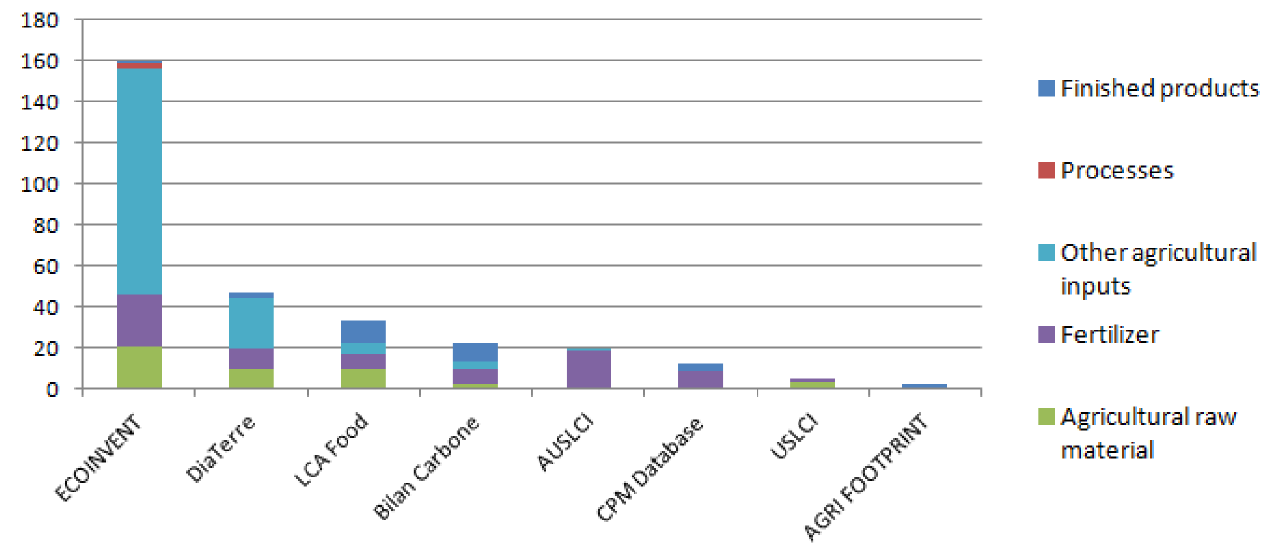


Figure 1. Data availability for cereals and cereal-based products in 8 LCI databases

155. The LCA world food database project: towards more accurate LCAs on agricultural and food products

Laura Peano[1,*], Yves Loerincik[1], Sebastien Humbert[1], Gérard Gaillard[2], Thomas Nemecek[2]

[1] Quantis, Parc scientifique EPFL, Bât. D, 1015 Lausanne, Switzerland, [2] Agroscope Reckenholz-Tänikon Research Station ART, Reckenholzstrasse 191, 8046 Zürich, Switzerland, [*] Corresponding author. E-mail: *laura.peano@quantis-intl.com*

The food and beverage sector is moving rapidly regarding sustainability issues (such as labelling purposes or "food eco-design"). This concerns both institutional and private organisations; consumers and environmental organisations also claim for more transparency on the environmental performance of food and beverage products. Life cycle assessment (LCA) has proven to be an effective method to assess the environmental impact of a product or service throughout its life cycle. However, currently, major limitations in doing such analyses are the lack of inventory data on food products and processes and a lack of consistency between existing food datasets. Therefore, there is a need to develop detailed, transparent, well documented and reliable data in order to increase accuracy of food LCA.

In this context, Quantis, the Agroscope Reckenholz-Tänikon Research Station ART and some leading companies in the food sector have decided to launch in 2012 the LCA world food database project.

The database will include datasets concerning agricultural raw materials (including, when possible and relevant, differences between production systems such as organic or non organic, intensive or extensive), inputs (such as pesticides and fertilisers), infrastructures (agricultural buildings, equipment and machinery), processes, processed food products, food storage, food transportation and food packaging. A consistent, but transparent, consideration of deforestation will also be included giving the possibility to assess it using different allocation rules. The data will come from existing LCAs on food products (partners' LCA, ART and Quantis existing databases), literature review on LCA of food products, statistical databases of governments and international organisations (such as FAO), environmental reports from companies, technical reports on food and agriculture, partners' information on food processes as well as collected primary data. Background datasets from the ecoinvent database will be used and new datasets will be compatible with ecoinvent.

To guarantee its transparency, the database will be fully documented, unit processes will be visible (except for confidential data provided by the companies) and information sources identified. The user will be able to differentiate among different stages of the process (e.g. agricultural production vs. food product manufacturing) and to identify the main contributors of a specific dataset (e.g. irrigation, fertiliser use, etc.).

The food database will be updated each year and will be compliant with quality requirements from major standards (ILCD Handbook, Sustainability Consortium, United Nations Environment Programme (UNEP), etc.) as well as with ecoinvent.

The project will start in March 2012 and will be completed in March 2015. The presentation will present the project (results of the literature review of the existing food datasets, involved companies, time schedule) as well as current state of the results (existing and new datasets, influence of consistent incorporation of deforestation, results and challenges associated with organic and non organic as well as extensive and intensive agricultural and animal systems).

References

Ecoinvent, 2010. ecoinvent database v 2.2. Available at www.ecoinvent.ch

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., 2003. IMPACT 2002+: a new life cycle impact assessment method. International Journal of Life Cycle Assessment 8 (6), 324–30. Version adapted by Quantis (http://www.quantis-intl.com).

European Commission, Joint Research Centre, Institute for Environment and Sustainability, 2010. International Reference Life Cycle Data System (ILCD) Handbook, General guide for Life Cycle Assessment, Detailed guidance. Publications Office of the European Union.

156. Seeds4green, a collaborative platform for LCA

Hélène Teulon*, Benjamin Canaguier

Gingko 21, Paris, France, * Corresponding author. E-mail: *helene.teulon@gingko21.com*

In 2009 and 2010, DEFRA (UK) joined forces with ADEME (French Environmental and Energy Management Agency) to sponsor the development of a collaborative LCA website with the technical support of the EU's Joint Research Centre. Seeds4Green is a wiki platform that aims to provide an easy way gather and share documents linked to environmental sustainability. Both agencies support the transparency and sharing of data and view this as a one of the solutions that allows a wider range of users to acquire LCA information more easily and promote sustainable goods and services.

We anticipate that the information stored here could be used by many audiences - from purchasers to eco-designers, businesses, eco-labelling teams within public authorities as well as LCA practitioners, researchers and students throughout the world. Currently summaries are in both French and English.

The purpose of the platform is to collaboratively build knowledge on the environmental quality of goods and to diffuse the results of LCA studies. It provides purchasing guidelines and systemised criteria making green purchasing operational and eco-labels even more transparent and comprehensive. Seeds4green intends to cover all product categories. Several LCA studies are available for agriculture goods. The platform provides a large range of topics such as organic vs conventional product or links towards food product LCA database.

The presentation/poster will explain the features of the platform including detailed examples of data summary sheets making it immediate accessible and showing its ease of integration in individual work habits. http://seeds4green.net/

Date de mise à jour	Titre	Type	Code
12/24/2011 - 08:33	Life cycle greenhouse gas emissions of 96 foods in US (per mass and per calorie)	ACV	Food, Plants
10/13/2011 - 00:46	Analyse de Cycle de Vie : production et transformation du lait	ACV	Food, Plants
10/10/2011 - 21:39	Life-cycle-assessment of industrial scale biogas plants	ACV	Gardening/Agriculture /Industry Energy Food, Plants
08/25/2011 - 17:17	Soil and Plant Nutrition Adapted to Organic Agriculture	ACV	Gardening/Agriculture /Industry Food, Plants
08/25/2011 - 15:53	Évaluation environnementale de systèmes de production laitiers : comparaison des systèmes conventionnels et biologiques avec l'outil EDEN	ACV	Gardening/Agriculture /Industry Food, Plants
08/24/2011 - 16:50	Environmental Evaluation of Greenhouse tomato production in France	ACV	Food, Plants
08/24/2011 - 09:38	Evaluation environnementale de la production de tomate en serre en France	ACV	Food, Plants
08/23/2011 - 17:45	Energy and environmental burdens of organic and non-organic agriculture and horticulture	ACV	Gardening/Agriculture /Industry Food, Plants

Figure 1. Extract of the available studies for the food-related products

157. ACYVIA: creation of a public LCI/LCIA database of the French food industry

F. Bosque[1], O.Réthoré[2], S. Kerner[3], H. Bortoli[2], Laura Peano[4], Sébastien Humbert[4,5], G. Gaillard[6], J. Lansche[6]

[1] ITERG, rue Monge, 33600 Pessac, France, [2] ADEME, 27 rue Louis Vicat, 75737 Paris Cedex 15, France, [3] IFV, Domaine de l'Espiguette, 30240 Le Grau-du-Roi, France, [4] Quantis, Parc scientifique EPFL, Bât. D – CH-1015 Lausanne, Switzerland, [5] Quantis, 42 boulevard de Sebastopol, 75003 Paris, France, [6] Agroscope Reckenholz-Tänikon Research Station ART, CH-8046 Zürich, Switzerland, [*] Corresponding author. E-mail: *f.bosque@iterg.com*

In the context of the Grenelle laws aiming at an environmental labelling on consumer goods, the ADEME, the French Environment and Energy Management Agency, has been mandated to set up a national LCI/LCIA database, in parallel with the development of Product Category Rules by the ADEME/AFNOR platform (Fig. 1). In terms of format, ADEME's database is based on the ILCD Dataset format defined by the JRC. In terms of content, three methods will be used to feed the database. The first and main one relies on the adaptation of existing processes and on the purchase of the requisite rights from data developers through framework contracts. For sectors with lacks of data, the ADEME is setting up collaborative projects to develop sectorial databases that will be merged to ADEME's database. The third mode will allow third-parties to propose the integration of isolated supplementary data.
A first three years collaborative project called Agri-BALYSE has been launched in 2010 with INRA, ART, CIRAD and ten agricultural institutes to develop a public LCI/LCIA database for the main French agricultural productions as well as some importations (see http://www2.ademe.fr/servlet/KBaseShow?sort=-1&cid=96&m=3&catid=12908). ACYVIA is a similar project aiming to deal with the next step of the food supply chain: transformation.
ACYVIA will apply a consistent methodology for the establishment of life cycle inventories of food transformation processes from the farm gate to the exit gate of the transformation plant. The methods used and the data format will be in accordance with ISO norms, the ILCD handbook and the general methodology adopted for the environmental labelling of consumer goods in France (BPX 30-323). The inventory data should allow the calculation of the indicators identified for environmental labelling, but also of other frequently used LCA impact categories.
The project will be carried out in collaboration by the following partners:

- ADEME will fund the project and assure its leadership and co-ordination.
- Food technological institutes (wine, spirits, dairy products, fats and oils, baking and pastry, meat industry, appertised or dehydrated food) will collect representative data and contribute to the methodological developments and the establishment of the inventories.
- Quantis and ART will lead the methodological developments and the computations of the life cycle inventories.

ACYVIA will systematically look for a consensus among the concerned experts, not only for the data used to produce the LCI, but also about major methodological questions such as allocation procedures. This is seen as a prerequisite for the success of the database: the involvement of interested stakeholders should favour the broad adoption of ACYVIA across the food chain. ACYVIA's database will be made of three levels of deliverables, corresponding to the needs of each partner:

- aggregated and averaged processes for ADEME's database;
- disaggregated and un-averaged processes to help the technical institutes assessing and lowering the environmental impacts of their supply chain;
- disaggregated and averaged processes.

This "vertical" construction of the database will be completed by a "horizontal" one consisting in the generation of "child" processes based on "parent" processes and ensuring a greater level of homogeneity for the database. For each child process, the values and relative proportions of the parameters of a given parent process are adapted to fit with the technical, geographical and temporal representativeness of the child process. For instance, a "parent" process "processing and packaging of milk" can be declined in several "child" processes specifically constructed to represent the geographical and/or technological specificities of a producer (e.g. PAST vs. UHT during sterilisation process)
The presentation will show the latest developments and the architecture of the database, the structure of the collaboration, and the main features of the database, such as:

- the expected processes to be included in the database
- the public availability;
- the homogeneity;
- the link with the ADEME database
- the wide range of usage: supporting the environmental labelling for customers and helping producers to decrease their impacts.

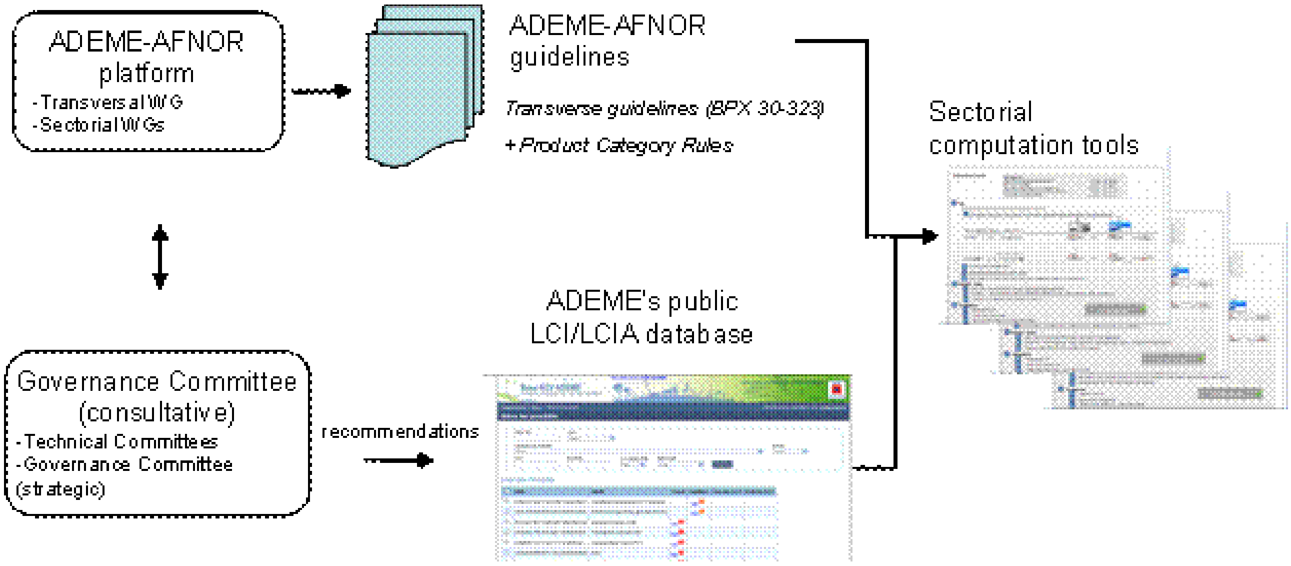


Figure 1. Overall context of ADEME's database

158. LCA as an evaluation tool for new food preservation techniques

Anna Aronsson*, Karin Östergren, Christoffer Krewer

Swedish Institute for Foods and Biotechnology (SIK), * Corresponding author. E-mail: *anna.aronsson@sik.se*

Alongside with new technology development it is becoming more common to evaluate the environmental impact using LCA. (e.g. Hospido et al., 2010, Sonesson, 2010).The demand for environmentally sustainable food products is increasing and it is a challenge for the food and drink sector to reduce environmental impact along the food chain while at the same time maintaining and improving quality. These demands have resulted in an increased focus on environmental quality aspects during the development of new food products and new technologies. Freeze protection with anti-freeze protein (AFP) (Phoon et al., 2008) and carbon dioxide (CO_2) drying (Agterof et al., 2005) are new preservation technologies under development. Soft vegetables and fruits such as strawberries (*Fragaria ananassa*) are suitable for both preservation methods. Both technologies lead to complete new products as freeze protected fruits can be thawed without cell damage and can be used as fresh fruits and dried fruits can be rehydrated resulting in recovered natural turgor. The environmental benefits of the new preservation technologies is the possibility of extending shelf life and substitution of imported fresh fruit and vegetables when out of season. The aims of this study were to (1) compare the environmental impact from cradle to retail of fresh, AFP freeze protected and CO_2-dried strawberries and (2) identify the key environmental issues after "farm-gate" to be able to evaluate the supply chain from processing to retail to support the technology development. Frozen / CO_2-dried and fresh strawberries were assumed to be cultivated and processed in Sweden and Egypt respectively. The study was performed using LCA methodology with a functional unit of 1 kg of ready to use strawberries. Global warming potential (GWP) and fossil energy use (MJ, data not shown) per functional unit in terms of primary energy was calculated. The resulting preliminary GWP for fresh, freeze protected and CO_2-dried strawberries after LCA was carried out were 4.6, 0.63, and 2 kg CO_2eq/kg strawberry respectively. The novel preservation technologies emitted less greenhouse gas emissions than the current method of freezing and the fresh imported strawberries (Fig. 1). The energy required for the novel technologies at the processing step accounted for 60-80% of the total GWP post-farm (Fig. 1). The results also demonstrate the importance of taking the full chain into account when evaluating new process technologies. An increased impact in one step may still lead to an overall improvement of the chain under given conditions. This study demonstrates how LCA can be applied to support the development of new technologies for food processing in an early stage by identifying potential hot spots early the development phase and by pinpointing suitable applications in relation to choice of inputs and production site.

References

Agterof, W.G. M., Bhatia R., Hofland G. W., 2005.Dehydration method. Feyecon Development & Implementation December 2005: WO 2005/122776

Hospido, A., Davis, J., Berlin, J. & Sonesson, U. 2010, LCA of novel products – Application to food processing, Int. J. Life Cycle Assess. 15, 44–52

Phoon, P.Y., Gomez Galindo, F., Vicente, A., Dejmek, P., 2008. Pulsed electric field in combination with vacuum impregnation with trehalose improves the freezing tolerance of spinach leaves. J. Food Eng. 88, 144-148

Sonesson, U., Novel Processing and Sustainability – a perfect match or not? New Food, Issue 5, 2010

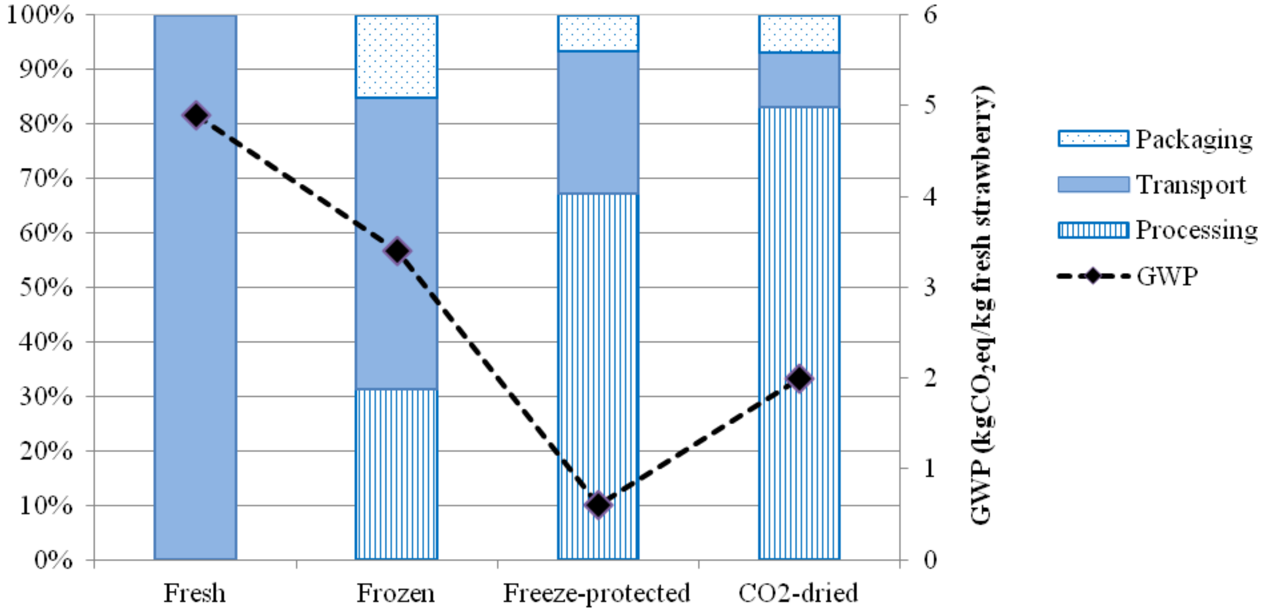


Figure 1. The GWP (secondary y-axis) and the contribution of packaging, transport and processing in percentage to the GWP (primary y-axis) of post-farm fresh, frozen, freeze protected and CO_2-dried strawberries.

159. Ecodesign of plant-based building materials using LCA: crop production and primary transformation of hemp

Andrianandraina[1,*], Hayo van der Werf[2,3], Anne Ventura[4], Rachida Idir[5]

[1] Université Nantes-Angers-Le Mans (LUNAM), Agence de l'Environnement et de la Maîtrise de l'Energie (ADEME), Institut Français Sciences et Technologies Transport Aménagement Réseaux (IFSTTAR), Département Matériaux, F-44341 Bouguenais, France, [2] INRA, UMR1069 Sol, Agro and hydroSystem, F-35000 Rennes, France, [3]Agrocampus Ouest, F-35000 Rennes, France, [4] Université Nantes-Angers-Le Mans (LUNAM), Institut Français Sciences et Technologies Transport Aménagement Réseaux (IFSTTAR), Département Matériaux, F-44341 Bouguenais, France, [5] Laboratoire Eco-Matériaux, CETE Ile de France, Paris, France, [*] Corresponding author. E-mail: *andy.andrianandraina@ifsttar.fr*

France has ambitious environmental goals regarding climate change and energy use, in particular since the so-called Grenelle de l'Environnement laws were passed. The building and housing sector is a major contributor to these impacts. New building materials, in particular thermal insulators, using plants like hemp as a raw material as presented by Korjenic et al. (2011) and Kymalainen and Sjoberg (2008), have appeared in the building materials market in recent years. These new products are often qualified as "green" low-impact materials because of their renewable origin.

A full life cycle assessment (LCA) of the material will be made for the following stages: crop production, industrial transformations, use and end-of-life. The environmental assessment of these materials using LCA is however complex, due to their multi functionality in buildings, and the lack of data on major life cycle stages, such as their behaviour in the use phase.

One of the main scientific challenges of this work is to develop and implement an LCA approach combined with a sensitivity analysis, in order to quantify the contribution and variability of technological, environmental and methodological factors to life cycle impacts (Fig. 1). This information will then be used to eco-design scenarios for the production, use, and end-of-life stages of plant-based building materials. The aim is to highlight the environmental hotspots of the life cycle of these materials and to reinforce the robustness of results of the LCA results by testing several scenarios.

The present article is focused on the production step of hemp fibres and shives: the agricultural system and primary transformation. Several scenarios are compared. Various sources of variability are taken into account: (i) technological variables such as the hemp cultivar, the various crop production modes (tillage intensity, harvest of seed and fibre or fibre only), primary industrial processes; (ii) environmental variables such as the soil type and the climate, (iii) as well as LCA methodological variables such as models (emission factors vs. dynamic simulation models) used to estimate direct emissions (e.g. nitrate) during crop production and allocation methods (based on mass, economic value or plant physiological mechanisms).

The repercussions of the variability of these input parameters are examined at the output i.e. the life cycle impact assessment. Using these results, eco-design scenarios for hemp fibres and shives production including crop production and primary transformation are proposed using processes which present interesting environmental potentials, and future improvements for each process are proposed.

Acknowledgments: The authors acknowledge the financial support of the French Agency for Environment and Energy Management (ADEME) and the Pays de la Loire region.

References

Korjenic, A., Petranek, V., Zach, J., Hroudova, J., 2011. Development and performance evaluation of natural thermal-insulation materials composed of renewable resources. Energy and Buildings 43:2518-2523.

Kymalainen, H.R., Sjoberg, A.M., 2008. Flax and hemp fibres as raw materials for thermal insulations. Building and Environment 43:1261-1269.

Life cycle stages	*Technological variables*	*Environmental variables*	*Methodological variables*
hemp seeds Crop production Stems Seeds	➢Cultivar (fédora, futura...) ➢Crop production mode (sowing, harvesting ...)	➢Climate ➢Soil type	➢Models for direct emissions ➢Lca methods (allocation, ...)
Primary transformation Fibre Shives	Processes ➢Primary industrial processes (retting, scutching ,...)	-	➢Data ➢Lca methods(allocation, ...)
Industrial transformation Thermal insulator	➢Chemical treatment ➢Composition of the material (binder, additive ....)		
Use	➢The material property (thickness, ...) ➢The behavior scenario (durability) ➢Heating type ➢Construction type (new, renovation)	➢Comfort zone ➢Building localization (Climate)	➢Data ➢Lca methods
End of life	➢Land filling ➢Composting ➢Incineration	-	➢Data ➢Lca methods

Figure 1. The life cycle of hemp-based building materials and major technological, environmental and methodological variables affecting life cycle impacts. This paper focuses on crop production and primary transformation.

160. Comparison of two milk protein separation processes: chromatography vs. filtration

Sophie Omont[1], Daniel Froelich[2,*], François Thueux[3], Murielle Rabiller-Baudry[4], Didier Beudon[5], Lionel Tregret[6], Christian Buson[7], David Auffret[7], Geneviève Gésan-Guiziou[8,9]

[1] ARTS, laboratoire MAPIE Chambéry, France, [2] Arts et Métiers ParisTech, laboratoire MAPIE, Chambéry, France, [3] ECOBILAN SA, Neuilly-sur-Seine, France, [4] Université Rennes 1, UMR 6226 «Institut des sciences chimiques de Rennes» CNRS; Rennes, France, [5]SOREDAB SA, Boissière, France, [6] Novasep Process, Saint Maurice de Beynost, France, [7]GES, Noyal sur Vilaine, France, [8] INRA, UMR1253 STLO, Rennes, France, [9]Agrocampus Ouest, UMR1253 STLO Rennes, France, [*] Corresponding author. E-mail: *daniel.froelich@ensam.eu*

Chromatography is the reference technology for protein fractionation at a large scale when high purity and targeted functionality are required. Fractionation of proteins by membrane operations such as micro- and ultrafiltration has however deserved a high attention over the last years, mainly due to their acceptable investment and operating costs. The aim of this study is to compare the environmental performances of two ways of fractionating whey proteins (chromatography and membrane filtration) into the two following added value products: powder of α-lactalbumin with a 70% purity and powder of β-lactoglobulin with high purity (> 95%) and high foaming properties. The study consists in a comparative attributional LCA, conducted within the context of the ECOPROM project (Eco-design of membrane processes). Project carried out with the financial support of the French national Research agency (ANR) under the Programme National de Recherche en Alimentation et Nutrition Humaine (Project ANR-06-PNRA-015").

The whey comes from processes commonly performed in a classical dairy: it corresponds to the aqueous phase of milk, obtained after microfiltration of skimmed milk (Omont et al., 2010). The whey is concentrated before its transportation into the upgrading plant where proteins will be purified and dehydrated. The system studied includes the entire process implemented, from the entry of the whey into the upgrading plant to the production of the two dehydrated fractions of purified proteins. The system takes into account all the processing operations, the cleaning phases and the associated equipment. It excludes the facilities (buildings, lighting, etc.). Its geographic scope is France. In this country, electricity is mainly produced by nuclear power and has a low CO_2 impact. As membrane processes are high electricity consumers (Notarnicola et al. 2008, European Commission 2006, Omont et al. 2010), a sensitivity analysis has been tested between French and European electricity mixes. The inventory of the foreground system was carried out with specific data given by the industrial partners of the project. Generic data derived from the ecoinvent V2.2 data base. The impact assessment was calculated by the IMPACT 2002+ method using the Simapro 7.2 software. A water flow indicator was defined for the first level processes. The "chromatography system" and the "membrane filtration system" are described on the Table 1. The fractions outgoing from the chromatography are more diluted. In order to generate the same concentrated α-lactalbumin and β-lactoglobulin fractions before drying, the ultrafiltration concentrations following chromatography were resized.

Drying steps consume the same amount of natural gas for the two processes. The environmental load of the "chromatography separation process" is mainly attributed to the ultrafiltration operations which consume more electricity due to the resize. The Chemical Oxygen Demand contained in the non-regenerated brine and in the column cleaning wastewater contributes to this load too, because their wastewater treatment consumes electricity. It is noticeable that sodium chloride is not decomposed by the wastewater treatment plant; its discharge into the water is then not assessed by the method IMPACT 2002+, which results in an underestimation of the environmental load of the chromatography process. The environmental impact of the "membrane filtration process" is mainly linked to the heating, the microfiltration and the ultrafiltration which consume natural gas, electricity and water due to diafiltration. As shown in Figures 1 & 2, the environmental load of chromatography tends to be higher than the membrane filtration. But the highest difference does not exceed 15%. The water consumption directly linked to the processes is 25% higher in case of chromatography.

References

European Commission, 2006. Intergrated Pollution Prevention and Control, reference Document on Best Available Techniques in the Food, Drink and Milk Industries

Notarnicola B., Tassielli G., Renzulli P., Settanni E., 2008. Life cycle assessment of a pilot plant for the must enrichment by reverse osmosis, Proc. of the 6th Int. Conf. on LCA in the Agri-Food Sector, Zurich, Nov. 12–14, 2008, pp.115-122.

Omont S., Froelich D., Osset P., Thueux F., Rabiller-Baudry M., Beudon D., Tregret L., Buson C., Auffret D., Gésan-Guiziou G., Eco-design of cascade membrane processes for the preparation of milk protein fractions: Approach and LCA results, Proc. of the 7th Int. Conf. on LCA in the Agri-Food Sector, Bari, Sept. 22-24, 2010, pp.197-202.

Table 1. The two systems studied.

Processes	Chromatography system	Membrane filtration system
1- Separation of the two proteins	Heating to 10°C, Chromatography separation, Brine recycling (Nano filtration and reverse osmosis)	Heating to 50°C, Acidification, Dilution, Microfiltration
2- Concentration of α-lactalbumin	Ultrafiltration of the α-lactalbumin fraction	Cooling, Re-solubilisation, Ultrafiltration, Cooling, Ultrafiltration
3- Concentration of β-lactoglobulin	Ultrafiltration of the β-lactoglobulin fraction	n/a
4- Powder formation	Drying of the α-lactalbumin; Drying of the β lactoglobulin	n/a

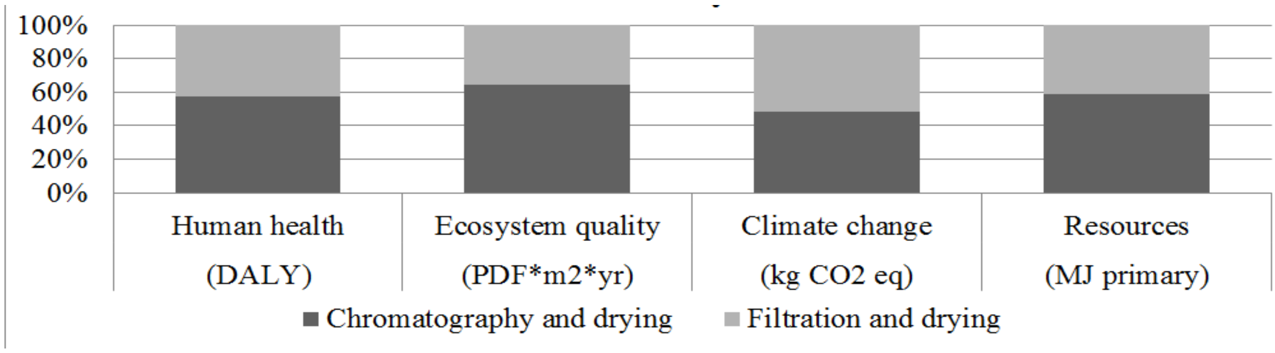


Figure 1. Chromatography and drying vs. Filtration and drying: LCIA with the French electricity mix, impact 2002+ method, SimaPro 7.2.

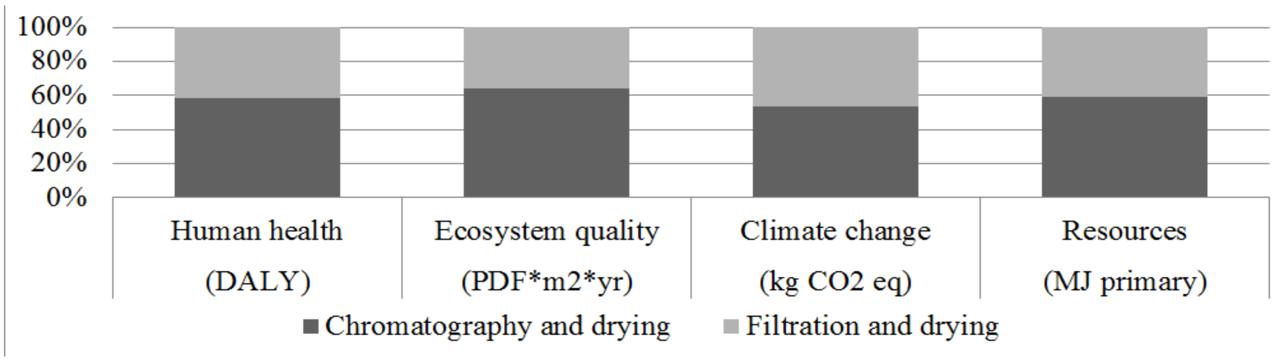


Figure 2. Chromatography and drying vs. Filtration and drying: LCIA with the European electricity mix, impact 2002+ method, SimaPro 7.2.

161. Design of a model for estimating LCI data by means of the Farm Accountancy Data Network. A citrus case study

J. Ribal[1,*], N. Sanjuan[2], G. Clemente[2], M.L. Fenollosa[1]

[1] Departamento de Economía y Ciencias Sociales. Universitat Politecnica de Valencia. Camí de Vera s/n 46022 València. Spain, [2] ASPA Group. Food Technology Department. Universitat Politecnica de Valencia. Camí de Vera s/n 46022 València. Spain, [*] Corresponding author. E-mail: *frarisan@esp.upv.es*

To carry out a Life Cycle Assessment means collecting a large amount of data. In agricultural production systems gathering of LCI data is specific for each study by surveying farmers or building scenarios after interviews with experts. According to Roches et al. (2010) different strategies are used to overcome the lack of relevant LCI and LCIA data: use of proxy data and generalisation, streamlined LCA and adaptation/extrapolation of inventories. In this work we have used a proxy methodology in order to obtain a straightforward procedure so as to estimate the order of magnitude of inventory data in agricultural systems
The main source of the model is the Farm Accountancy Data Network (FADN). FADN is an instrument for evaluating the income of agricultural holdings and the impacts of the Common Agricultural Policy. It gathers accountancy data from farms, mainly production costs and revenues. In each country a structured cost sheet is available for each family crop, region and economic size. Each cost sheet represents the average farm in each group. Additionally our model uses specific data of each crop and geographical area. Fig. 1 shows the outline of the model.
Thus the objective of the model is to estimate the LCI from the data of the FADN and then to carry out an impact assessment. To test and contrast the model a case study in orange citrus has been built comparing the results with Ribal et al. (2010).
This model would allow obtaining environmental impacts at a macro level in order to quantify eco-efficiency of crop production or to measure the value added integrating environmental impacts.

References

Ribal, J.; Sanjuan, N.; Clemente, G.; Fenollosa, M.L. 2009. Medición de la eco-eficiencia en procesos productivos en el sector agrario. Caso de estudio sobre producción de cítricos. Economía agraria y recursos naturales. 9, 125-148.

Roches, A.; Nemececk, T.; Gaillard, G.;Plassman, K.; Sim, S.; King, H.; Milà i Canals, L. 2010. MEX-ALCA: a modular method for the extrapolation of crop LCA. International Journal of Life Cycle Assessment. 15,842-854.

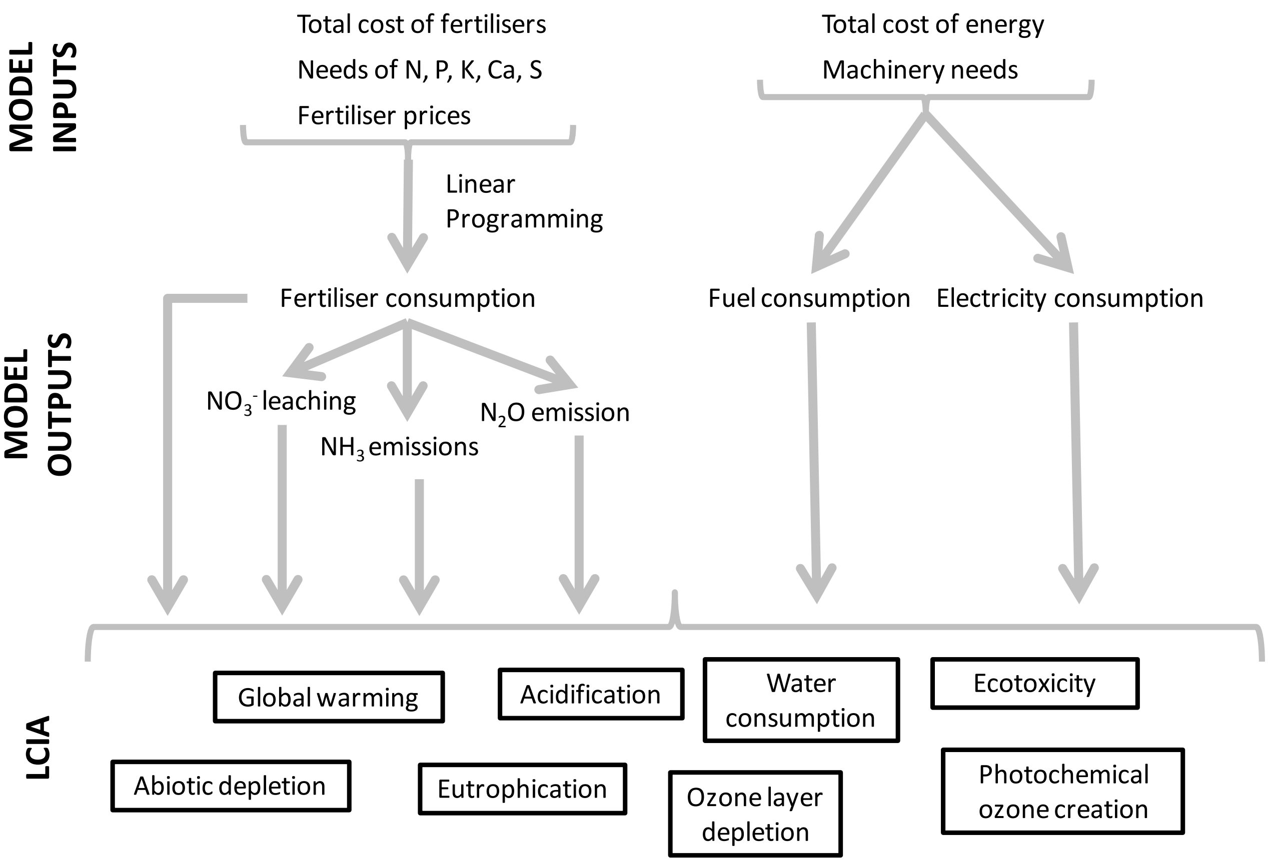


Figure 1. Outline of the LCI-FADN proxy model.

162. A simple model to assess nitrate leaching from annual crops for life cycle assessment at different spatial scales

Aurélie Tailleur*, Jean-Pierre Cohan, François Laurent, Afsaneh Lellahi

ARVALIS – Institut du végétal, Service Agronomie Economie Environnement, Boigneville, F-91720, France, * Corresponding author. E-mail: *a.tailleur@arvalisinstitutduvegetal.fr*

Life cycle assessment of food products show that nitrate leaching during crop production contributes significantly to their eutrophication potential (Xue X, Landis AE, 2010). Different dynamic field-scale models enable LCA-practitioners to estimate nitrate leaching taking into account detailed data about agricultural practices, soil type and climatic conditions. In the meantime, their use at a higher spatial scale (production area, region) requires collecting a high number of data on a representative sample of fields. In this study, we estimate average potential nitrate emissions for twenty annual French crops, using an easy to perform method based on the COMIFER (French Committee for the Development of Rational Fertilisation) methodology. This methodology was previously established from experimental results to diagnose nitrate leaching risk at different scales (watershed, small agricultural region, production area) and allows to classify the risk considering agricultural practices (from a very low to a very high level in function of duration without N absorption, absorption capacity of the following crops, slurry and droppings spreading in autumn, N input from crop residues, Fig. 1) and some important site-specific related soil parameters (water retention capacity, volume of drained water and soil organic matter content). The nitrate leaching amounts were attributed to each risk level (Fig. 2), based on results from the experimental database of the institute. Statistical data from the French Ministry of Agriculture about agricultural practices and our soil database were used to assess, at the region scale, percentages of crop area corresponding to each risk level and then to estimate an average regional nitrate leaching amount. Results will be compared with experimental results on particular sites and model results on watershed scale provided by bibliography.
This work shows that we can easily use this methodology in the framework of LCA to provide potential nitrate emission estimations at a regional scale. While these estimations may be less precise at smaller scale than those from dynamic field-scale models as the comparison with experimental results could show it, it requires collecting much less data and it takes into account the most impacting parameters to differentiate contrasting situations.
Different emissions (N-gases, plant protection products) from crop production are very variable due to conditions and practices. To estimate precisely these emissions, LCA-practitioners have often to face with an important need for data. In the meantime, our work suggests that it is possible to establish structured and simplified methods that can quickly and pertinently discriminate situations.
This work contributes to the Agri-BALYSE project which is aimed at providing a Life Cycle Inventory (LCI) database for agricultural products. Average LCI representative of the French production will be supplied for around twenty annual crops.

References

COMIFER, 2001. Lessivage des nitrates en systèmes de cultures annuelles. Diagnostic du risque et proposition de gestion de l'interculture. COMIFER Ed. 41 p.
CORPEN, 1991. Interculture. CORPEN Ed. 40 p.
Cohan J.P., Laurent F., Champolivier L., Duval R., 2011. Diagnostic du risque de lixiviation du nitrate et leviers d'actions. In « Cultures intermédiaires: enjeux et conduite ». Ed. ARVALIS-CETIOM-ITB-ITL ; pp. 128-136.
Xue X, Landis AE, 2010. Eutrophication Potential of Food Consumption Patterns. Environmental Science and Technology 2010, 44, 6450-6456.

Risk level regarding absorption capacity after harvest

Capacity absorption of the following crop	Number of months between harvest and seeding of the following crop			
	< 1 month	1-3 months	3-6 months	> 6 months
No cover crop*	2	3	4	5
Cover crop or oilseed rape as following crop	1	1	2	2

*** Risk regarding fertilisation**

Add + 1 to the risk in the absence of cover crop when slurry or droppings are spread on the following crop in autumn or when N inputs exceed crop needs.

Risk level regarding available N quantity from crop residues

Crop	Straw exported ?	Risk level
Cereals	No	5
	Yes	1

Risk inherent to agricultural practices

Figure 1. System of classifying risk inherent to agricultural practices established on the basis of COMIFER's method and the database of the institute (1: very low risk, 5: very high risk), example of cereals.

		Risk inherent to agricultural practices				
		1	2	3	4	5
Risk inherent to the site	1	5	10	20	25	30
	2	10	15	25	30	40
	3	15	20	30	40	50
	4	20	30	40	55	60
	5	30	40	40	60	80

Figure 2. Nitrate leaching amount (kg $N\text{-}NO_3^-$/ha) regarding risks (1: very low risk, 5: very high risk) inherent to practices (Fig. 1) and to soil parameters (water retention capacity, volume of drained water and soil organic matter content).

163. Life cycle sustainability assessment of fertilising options

Julia Martínez-Blanco[1,*], Annekatrin Lehmann[2,3], Pere Muñoz[4], Assumpció Antón[5], Marzia Traverso[2], Joan Rieradevall[1,6], Matthias Finkbeiner[2]

[1]Institute of Environmental Science and Technology (ICTA), Universitat Autònoma de Barcelona (UAB), 08193 Bellaterra, Barcelona, Spain, [2] Department of Environmental Technology, Technische Universität Berlin, Office Z1, 10623 Berlin, Germany, [3] Institute for Technology Assessment and Systems Analysis, Karlsruhe Institute for Technology, 76021 Karlsruhe, Germany, [4] Institute of Research and Technology in Agrifood Sector (IRTA), Environmental Horticulture, 08348 Cabrils, Barcelona, Spain, [5]Departament d'Enginyeria Química, Universitat Rovira i Virgili (URV), 43007 Tarragona, Spain, [6]Chemical Engineering Department, Universitat Autònoma de Barcelona (UAB), 08193 Bellaterra, Barcelona, Spain, [*] Corresponding author. E-mail: *julia.martinez@uab.cat*

Sustainability describes a development that meets the needs of the present without compromising the ability of future generations to meet their own needs (Brutland report, 1987). It comprises three "pillars" – environment, economy and society– which are addressed by Life Cycle Sustainability Assessment, LCSA, (Finkbeiner et al., 2010). The study aims to overview the three dimensions related to mineral fertilisers and compost, from a life cycle perspective and considering a real case study –the tomato production in the Mediterranean region (Martínez-Blanco et al., 2011).
The unit of available nitrogen in the short-medium term (1kg N) from the applied fertilisers to the tomato crops is the functional unit considered here. Fig. 1 shows the system boundaries of the two systems compared: compost and two mineral fertilisers (nitric acid and potassium nitrate). For the present study, fertiliser production and transport are taken into account as well as those stages of the cultivation being different between the two fertilising options, i.e. fertirrigation emissions and application works.
Data sources for the environmental and the economic assessment are mainly referred in previous works from the authors, whereas social evaluation uses sector data from Social Hotspot DataBase, Gabi 5, governmental and non-governmental organisations (such as ILO and OECD), corporate websites, sustainability reports, national statistics and literature.
Environmental Life Cycle Assessment (LCA) is following the obligatory classification and characterisation phases defined by the ISO 14044. Ten mid-point impact categories are considered as well as an energy flow indicator. For Life Cycle costing (LCC) we select three internal costs, which are involving all the chain process costs. Regarding Social Life Cycle Assessment (SLCA), upstream processes and mainstream processes of the production chain are taken into account. For the former, the indicators are referred to the stakeholder "worker", "local community", "society" and "consumer" – as proposed in the S-LCA guidelines (UNEP, 2009) – as well as a specific stakeholder related to the citizens collecting the waste. Social impacts related to upstream processes have so far not been considered in the few existing SLCA case studies. In our study some social indicators addressing the stakeholder "worker" are included for the upstream processes in the life cycle of compost and mineral fertiliser production (such as transport, energy, water, etc.). The number of working time which is spent on each unit process is used to score the relevance of each process in the product chain.
Life Cycle Sustainability Dashboard (LCSD) – an adaptation of the Dashboard of Sustainability, developed to assess environmental, economic and social life cycle impacts of a product (Finkbeiner et al., 2010) - was here selected to present and to compare the sustainability performance of the case studies. LCSD is using software that allows represent a certain number of indicators for LCA, LCC and SLCA, and their values, in order to be able to interpret the results.

References

Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards life cycle sustainability assessment. Sustainability 2, 3309-3322.
Martínez-Blanco, J., Muñoz, P., Antón, A., Rieradevall, R., 2011. Assessment of tomato Mediterranean production in open-field and in standard multi-tunnel greenhouse with compost or mineral fertilisers from an agricultural and environmental standpoint. J. Clean. Prod. 19, 985-997.
UNEP, 2009. Guidelines for Social Life Cycle Assessment of Products, UNEP-SETAC Life-Cycle Initiative, Paris.

17

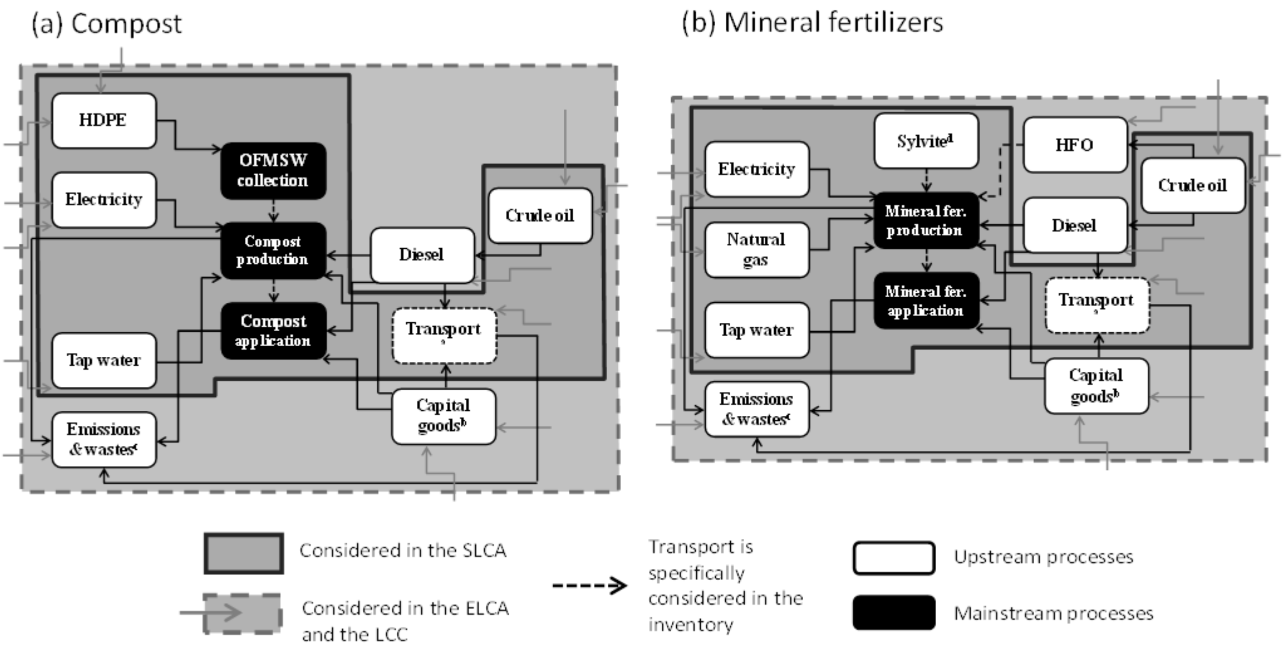


Figure 1. Compost and mineral fertiliser system boundaries for the three LCA approaches.

[a]Dotted arrows are involving transport processes/companies/costs.

[b] Capital goods include machinery, buildings and infrastructures.

[c] For fertiliser production, emissions are included in costs as technologies to reduce emissions, while no real money flows are occurring for emissions produced during fertiliser application.

[d] Sylvite is only included for KNO_3 production. HFO, Heavy fuel oil, HDPE, high density polyethylene.

164. Get SET, an innovative holistic approach to implement more sustainable nutrition

Georg Schöner*, Christoph Günther, Cristian Barcan, Kristina Gräper

Global SET Initiative Nutrition & Health, BASF, G-ENL/MS, * Corresponding author. E-mail: *georg.schoener@basf.com*

Feeding the world 2050 and beyond is a challenge particularly, when it comes to sustainable nutrition. It is now time for the industry to optimise the sustainability of their products step by step. To succeed, it is necessary to have a holistic, "cradle-to-grave" approach which involves stakeholders throughout the entire value chain while focusing on consumer goods. This paper shows how applied science and value chain management through BASF's SET Initiative meets those needs.

In principle, today SET is built on three pillars:

1. *Hot Spot Analysis*, a qualitative assessment tool, which helps identify major concerns and perceptions related to the sustainability of a product based on structured stakeholder interviews and relevant publications dealing with the entire value chain.
2. *Eco-Efficiency Analysis*, a life cycle assessment approach for measuring a product's environmental impact from cradle to grave. It includes at least 11 impact categories, such as greenhouse gas emissions, resource consumption, photochemical ozone formation or land use and transformation. Trade-offs between different impact categories can only be overcome by assessing all possible parameters, not only one aspect such as global warming potential, reflected with a product's carbon footprint. The identified impact is then contrasted with the final product's economic value and reveals the parts of the value chain that respond most sensitively to greater sustainability.
3. A *whole chain traceability program* that helps companies to make their supply chain more transparent – a prerequisite for managing sustainability. Traceability not only helps trace all of the components that lead up to a final product through the value chain, but also – and more importantly – makes it possible to follow a tailored plan of action and track the progress made over time.

Multiple projects have been carried out in the food- and feed industry, which show how current and future sustainability opportunities can be leveraged for everybody's benefit. Examples for the three pillars mentioned above are presented from studies on the sustainability optimisation in feed/food value chains. All stages of a life cycle are considered, for example for the production of pork: all aspects of the feed production phase, the animal breeding/fattening, meat/carcass processing, distribution, retail display, as well as consumption and disposal. Using BASF's SET Initiative, the dynamic and different perceptions of several supply chain actors are understood. The involvement of suppliers and customers up and down the value chain enables the producers to counter current hot spots and to drive towards more sustainable nutrition products. The value chain partners and customers can benefit through a proof of sustainable practices, sustainability optimisation and product or brand repositioning and differentiation in the market. SET also identifies potential for product innovation and generates additional brand equity. Along the way, this value adding partnership program enables the customers to gain a higher reputation in the market as a sustainable acting business.

165. Subcategory assessment method: stakeholder workers

Paola Karina Sánchez Ramirez[1,*], Luigia Petti[1], Cássia Maria Lie Ugaya[2]

[1] Department of Economic Studies, Università degli Studi "G. D'Annunzio", Pescara, Italy, [2] Technological Federal University of Parana, Curitiba, Brazil, [*] Corresponding author. E-mail: *p.sanchez@unich.it*

Social Life Cycle Assessment (S-LCA) is a technique to evaluate potential positive and negative impacts along the life cycle of a product (UNEP and SETAC 2009). S-LCA follows four phases, similarly to an (environmental) LCA: goal and scope definition, inventory analysis, impact assessment and interpretation; nevertheless it requires adaptations (Grießhammer, Benoît et al. 2006). Going from data to impact assessment in S-LCA is still a challenge. UNEP and SETAC (2009) have presented a contribution by providing a list of 31 subcategories related to 5 stakeholders (workers, consumers, local community, society and value chain actors) that can be either aggregated to impact categories or modelled, through pathways, to endpoints. Thereafter, methodological sheets for each of the subcategories were elaborated, including definition, contribution for the sustainable development, unit and even possible data sources (UNEP and SETAC 2010). Some Life Cycle Impact Assessment methods, to subcategory level, were also proposed (Dreyer, Hauschild and Schierbeck (2006); Ciroth and Franze (2011)). The first one is limited to worker stakeholder and the second is not clear how to evaluate from data to the subcategory and how to aggregate subcategories into impact categories. The aim of this study is to propose a Subcategory Assessment Method (SAM) to decrease the variability of the evaluation of subcategories in S-LCA studies. A proposal for stakeholder worker and a case study to test it are presented. SAM includes 8 subcategories (freedom of association and collective bargaining, child labor, fair salary, working hours, forced labour, equal opportunities/discrimination, health and safety, social benefits/social security) from the stakeholder worker of UNEP and SETAC (2009). SAM enables the analysis of the organisation in four classes (A, B, C and D) for each subcategory. Fulfilling Class A means that the organisation shows a proactive behaviour compared to the basic requirement. The basic requirement is defined for each subcategory, based on International Agreements. Class B means that the organisation follows the basic requirement. Classes C and D identifies the organisation which does not meet the basic requirement and are differentiated due to generic data which provides background information concerning the possibility of the environment to have a positive outlook to social issues. SAM was applied in a small winery to evaluate stakeholder worker. The case study showed that it was possible to collect data and evaluate a company using SAM. Results show that S-LCA is as time and work demanding as (environmental) LCA, since the issues are specific and change from company to company, sector to sector, and region to region. SAM may also be implemented for the whole product life cycle. Future development of SAM will include the other subcategories, adapting the basic requirement for each one.

References

Ciroth, A., Franze, J., 2011. LCA of an Ecolabeled Notebook – Consideration of Social and Environmental Impacts along the entire Life Cycle, Berlin.

Dreyer, L.C, Hauschild, M., Schierbeck ,J., 2006. A framework for social life cycle impact assessment. Int J Life Cycle Assess. 11, 2, 88–97.

Grießhammer, R., C. Benoît, et al., 2006. Feasibility study: integration of social aspects into LCA. Available from http://www.saiplatform.org/uploads/Library/UNEP-SETACLifeCycleInitiativeTFonSocialIssues-FeasibilityStudy.pdf Last visited on 13/01/2012.

UNEP and SETAC, 2009. Guidelines for social life cycle assessment of products, United Nations Environment Program, Paris SETAC Life Cycle Initiative United Nations Environment Programme.

UNEP and SETAC, 2010. Social Life Cycle Assessment: Methodological Sheets of Sub-Categories of Impact for Social LCA. Available from http://fr1.estis.net/builder/includes/page.asp?site=lcinit&page_id=D45A9A8F-10FA-4501-BF93-FB9AEAE4D163 Last visited on 23/01/2012

166. Supply-demand balances of food and energy in mountainous rural areas to help ensure regional sustainability

Kazutoshi Tsuda[1,*], Keishiro Hara[2], Michinori Uwasu[2]

[1] Creative Design Studio on Technology, Osaka University, Osaka, Japan, [2] Center for Environmental Innovation Design for Sustainability, Osaka University, Osaka, Japan, [*] Corresponding author. E-mail: *tsudakazutoshi@lce.mech.eng.osaka-u.ac.jp*

Regional sustainability with resilience is becoming an important concept. In particular, securing potential of self supply for food and energy within local communities, together with local independence, is one of the essential conditions for realising regional sustainability. In order to discuss such potential of self supply for food and energy, re-valuing local natural resources and enhancing a proper balance between supply and demand of renewable resources within a region is of critical importance. However, studies which address this aspect are very scant. In addition, in an attempt to achieving regional sustainability, it is essential to provide information useful and practical enough for making action plans or policy making for local governments. For this purpose, it is required to understand the causal relations among sustainability indicators and materials and energy flow within a certain boundary, in addition to the construction of database.

In Japan, hilly and mountainous areas are approximately three-quarters of the total land areas, and cover the forty percent of the total cultivated areas. These areas hold the key to regional sustainability in Japan. Shinjo village is located in the mountainous area in western Japan. The village is a unique municipality trying to enhance self-reliance in terms of food, energy and financial conditions, aiming to avoid a possible merger with other bigger cities. Unique visions and proper measures are needed for the village to keep the self-reliance viable and to pursue regional sustainability especially at a time when the labor force is shrinking due to the aging population and resultant declining local economy.

In this study, we aim to discuss the outlooks of self-reliance level for Shinjo village from the viewpoint of supply and demand of food and renewable energy, especially biomass energy such as firewood and wood/rice husk charcoal. We first looked into geographical data and examined ecological conditions and local landscapes, which serve as the basis for evaluating local natural resources. We then developed inventory data of food and renewable energy available within the region by applying material flow analysis and life cycle assessment methodology. Based upon the information we demonstrated the assessment of the regional sustainability from the viewpoint of demand and supply balance for food and energy as well as self-reliance of economic conditions.

Figure 1 illustrates the natural resources flow in Shinjo village. The flow chart consists of forestry, agriculture, livelihood, and composting process. We found that Shinjo village has sufficient food production while it heavily relies on energy inflow for fossil fuels. On the other hand, the village has large supply potentials for renewable materials and energy. We proposed institutional and technical options as well as policy measures, which utilise these rich resources, to further enhance regional sustainability of the village.

We conclude this paper by proposing the following as key research questions for this regional sustainability: conducting assessment of ecosystem services in a rigorous and comprehensive way, which includes a completion of a material flow chart for the entire region, and exploring the ways to utilise those ecosystem services along with the establishment of systems to mobilise human resources, money, and materials between the rural and urban areas.

References

Tsuda, K., Nakakubo, T., Umeda, Y., Morioka, T., 2011. Potential accounting of regional urban-rural partnership for constructing resource-circulating systems, in: Morioka, T., Hanaki, K., Moriguchi, Y. (Eds), Establishing a resource-circulating society in Asia: Challenges and opportunities. United Nations University Press, Tokyo, pp. 184-200.

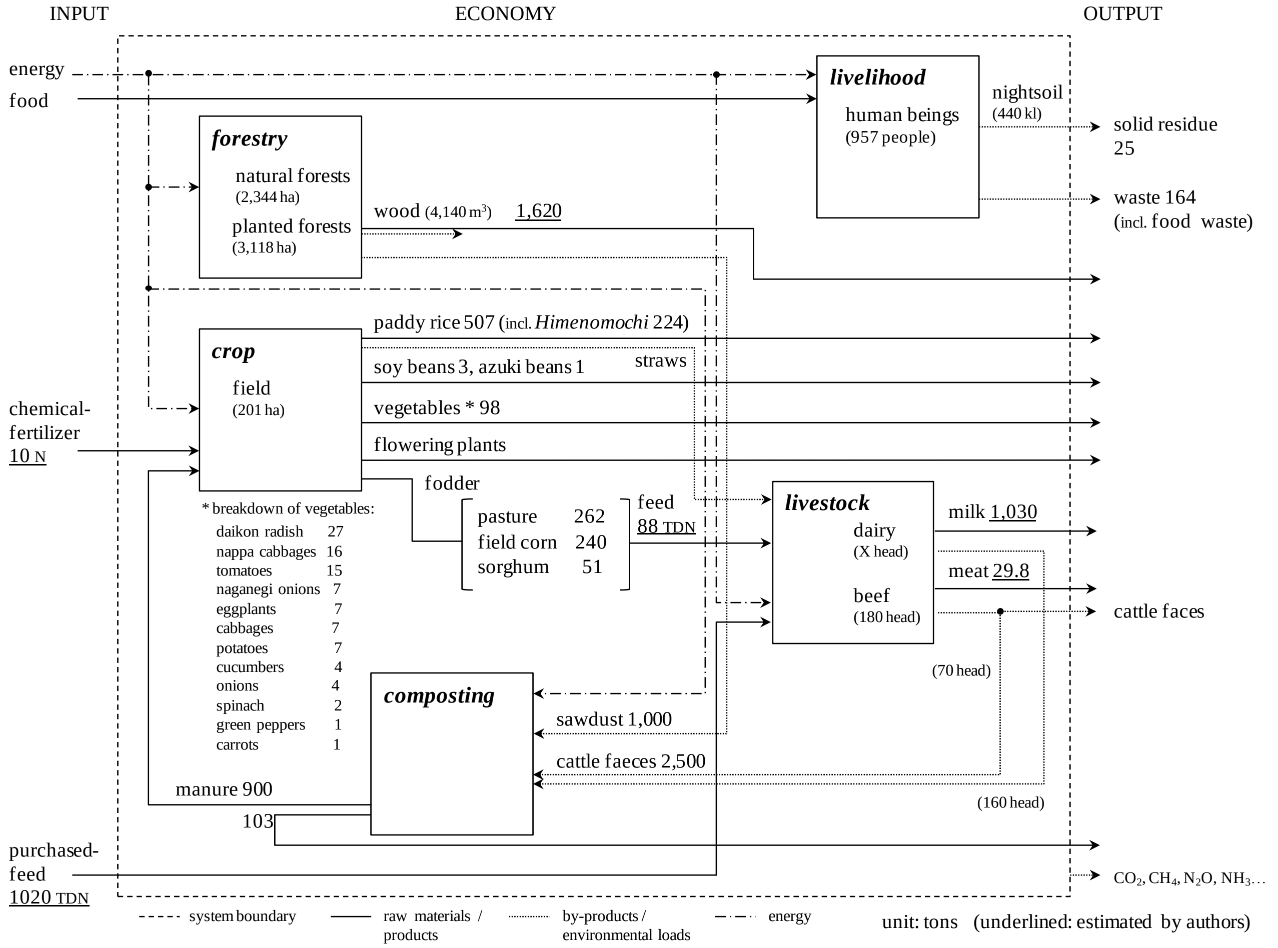


Figure 1. Natural-resource flows of Shinjo village, Okayama, Japan.

167. AgBalance – holistic sustainability assessment of agricultural production

Jan Schöneboom[1,*], Peter Saling[1], Richard van Gelder[1], Markus Frank[2], Dirk Voeste[2], Martijn Gipmans[3]

[1] Sustainability Centre, BASF SE, GU/NE – Z7, 67056 Ludwigshafen, Germany, [2] Crop Protection Division, BASF SE, Limburgerhof, Germany, [3] BASF Plant Science Tegeler Weg 33, 10589, Berlin, Germany, [*] Corresponding author. E-mail: *jan.schoeneboom@basf.com*

Life Cycle Assessment (LCA) and approaches based thereon, e.g., Eco-Efficiency Analysis and SEEBALANCE, have proven useful tools for quantitative sustainability assessment along value chains and across industry sectors, particularly for industrial manufacturing and consumption. While these approaches are commonly also applied to assess the sustainability of agricultural products or production systems, there is a need for further development in order to adequately evaluate specific impacts of agricultural activity, particularly relating to biodiversity and soil quality.

Herein, we present a new method for sustainability assessment in agriculture, named AgBalance. Based on the environmental impacts and life cycle costs assessed in BASF's Eco-Efficiency Analysis, and the social impact indicators in SEEBALANCE, AgBalance additionally serves to record and evaluate a range of specific agricultural management indicators. AgBalance thus combines environmental LCA, social LCA and life cycle costs with ecological, economic and social sustainability indicators related to agricultural production, which are generalised to varying spatial scales. Its flexibility allows consideration of variable upstream and downstream processes, includes all three dimensions of sustainability in an integrated approach, and is suitable for application in different regions.

AgBalance comprises up to 70 indicators, based on a significantly larger number of input data and parameters. The indicators show different impacts and are grouped into 16 categories. They are then aggregated in the three dimensions environment, society and economy. Specific impacts on biodiversity in agricultural areas are assessed by a set of indicators: biodiversity state/endangered species, protected area coverage, agri-environmental schemes, pesticide eco-toxicity potential, nitrogen surplus, intermixing potential, crop rotation elements, and farming intensity. Impacts on soil quality in agricultural areas evaluate the indicators nutrient balances, soil organic matter balance, compaction potential and erosion potential. Social indicators include, in addition to those covered in SEEBALANCE, societal representation of agriculture, observation of food law requirements, access to land, trade balances and fair trade benefits for producers. Economic indicators cover farm profitability and productivity, in addition to life-cycle costs.

Both, detailed in-depth results of individual impact indicators, as well as aggregated results and a single sustainability evaluation score are output of AgBalance. Sensitivity analyses can be employed to study the robustness of the model results, and to investigate trade-offs or the response to external influences. Scenario Analyses can model different situations by simulating new sets of inputs followed by an assessment of improvement potentials of the analysed system. AgBalance is useful for assessing and managing sustainable development in agriculture for farmers, business in the food value chain, decision making and policy making and for public communication.

References

Saling P., Kicherer A., Dittrich-Kraemer B., Wittlinger R., Zombik W., Schmidt I., Schrott W., Schmidt S., 2002. Eco-efficiency Analysis by BASF: The Method. Int. J. Life Cyle Assess. 7, 203-218.

Kölsch, D., Saling P., Kicherer A., Grosse-Sommer A., Schmidt I, 2008. How to measure social impacts? A socio-eco-efficiency analysis by the SEEBALANCE® method. Int. J. Sustain. Dev. 11, 1 – 23.

Schöneboom J.C., Saling P., van Gelder R. 2012. AgBalance – holistic assessment of sustainability in agriculture. Submitted to Int. J. Life Cycle Assess.

168. Social life cycle assessment of milk production in Canada

Jean-Michel Couture[1,*], Catherine Brodeur[1], Julie Parent[2], Jean-Pierre Revéret[1,2], Mia Lafontaine[3]

[1] Groupe AGECO, 2014 Cyrille-Duquet suite 307, Quebec, QC, G1N 4N6, Canada, [2] UQAM, Department of Strategy, Social and Environmental Responsibility, Suite R-3555, École des sciences de la gestion, QC, H2X 3X2, Canada, and Groupe AGECO, [3] Quantis, 395 Laurier Ouest, Montreal, QC, H2V 2K3, Canada, [*] Corresponding author. E-mail: *jean-michel.couture@groupeageco.ca*

The agricultural sector in general and the livestock and dairy sectors in particular have been over the years increasingly blamed for their environmental impacts, especially in regards to their greenhouse gases emissions (Steinfeld et al. 2006). In the same time, there has been a growing awareness that farming activities equally induce significant social and economic impacts over a wide range of stakeholders (Van Calker et al. 2005). As new challenges arise from this context, the sector needs to improve his sustainability level to respond to critics.

In this regards, the Life Cycle Assessment (LCA) approach, which addresses the environmental impacts throughout a product's life cycle, has become a widely used tool to identify hotspots and to foster actions to improve a product' environmental sustainability. With the recent development of the Socio-economic Life Cycle Assessment (S-LCA) methodology, this approach now also enables to assess, in an integrated way, for the social and economic aspects of sustainability. In order to document the Canadian milk sector' sustainability, this integrated LCA framework has thus been used in this project to comprehensively define the environmental and socio-economic impacts of the sector's whole life cycle.

However, whereas E-LCA now relies on a well-defined methodology, S-LCA is a new technique whose methodological underpinnings have only been recently specified in the UNEP's Guidelines for Social Assessment of Products released in June 2009 (UNEP 2009). Based on those Guidelines, this project has developed a fully operational assessment methodology adjusted to the Canadian milk sector' specificities. More specifically, two assessment frameworks have been used to assess the social and economic impacts induced all along the Canadian milk sector's life cycle. For enterprises found within dairy farms' sphere of influence, a Specific Analysis has been performed by using a set of indicators related to the sector' specific issues of concern and stakeholders categories. A Potential Hotspot Analysis has been used instead to assess, outside the dairy farms' sphere of influence, the potential misbehaviours of enterprises in regards to acknowledged social norms.

In addition of offering a detailed set of socio-economic indicators reflecting the particular social issues of concern found within the agricultural sector, this project contributed to the advancement of the S-LCA methodology by clarifying the steps to conduct such an assessment in practice, in particular in regards to the two unique and complementary assessment frameworks. It proposes furthermore to integrate to the social and economic dimensions of sustainability with the environmental one in one broad assessment methodology.

References

UNEP/SETAC, 2009. Guidelines for Social Life Cycle Assessment of Products. United Nations Environment Programme. Paris.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., De Haan, C., 2006. Livestock's long shadow: Environmental issues and options. FAO [Food and Agriculture Organization of the United Nations], Rome, 391 p.

Van Calker, K.J., Berentsen, P.B.M., Giesen, G.W.J., Huirne, R.B.M., 2005. Identifying and ranking attributes that determine sustainability in Dutch dairy farming. J. Agri., Food, and Human Values Society 22, 53-63

169. Environmental impact and social attributes of small- and large-scale dairy farms

Luciana Bava, Matteo Guerci*, Anna Sandrucci, Alberto Tamburini, Maddalena Zucali

Dipartimento di Scienze Agrarie e Ambientali, Università degli Studi di Milano, via Celoria 2 , 20133 Milano, Italy, * Corresponding author. E-mail: *matteo.guerci@unimi.it*

In recent years a growing interest was observed from public opinion in the concept of "sustainability" of farming systems (Van Calker et al., 2005). A common perception is that a dairy farm based on pasture, with low-input and low number of cows is more respectful from the environmental point of view than an intensive and large dairy farm (Capper et al., 2009). The aim of this work is to study the environmental impact and the social attributes of intensive dairy farms characterised by different scale in terms of number of lactating cows. We selected 22 dairy farms located in the Po valley in the North of Italy. All the farms were members of the same cooperative feed industry and belonged to one of the two groups: ≤ 70 or ≥ 150 lactating cows. The environmental impact of each dairy farm was calculated with a detailed ''cradle-to-farm-gate'' LCA. All the processes related to the farm activity (i.e. forage and crop production, energy use, fuel consumption, manure and livestock management), and all external factors or inputs (i.e. production of fertilisers, pesticides, feed, energy and fuels, litter materials, replacing animals) were considered as part of the system. The functional unit chosen was 1 kg fat and protein corrected milk (FPCM, 4.0% of fat and 3.3% of protein content). LCA was carried out with SimaPro 7.3.2 (PRé Consultants bv., 2011). Gross margin, i.e. revenues minus direct production costs, excluding labour cost (€/t FPCM) was used as economic indicator. The social attributes of the farming systems were studied using an on-line questionnaire sent to a large sample of stakeholders with different age (18 to more than 60). Daily milk production, stocking density and feed self-sufficiency were not significantly different between the two group of farms; also the production efficiency and economic performance, expressed as dairy efficiency and gross margin, were similar (Table 1). Large scale farms had higher percentage of farm land sown with maize for silage, lower percentage of grassland in comparison with the other group. Nitrogen and phosphorus balances at farm level did not show any significant difference among farms. Climate change and acidification potentials per kg FPCM showed significantly lower value in the large scale farms ($P < 0.05$) compared with smaller ones (Table 2). The results in terms of climate change potential were in agreement with previous studies of Rotz et al. (2010); this could be due to the reduction of methane emission determined by the higher intake of maize silage and high moisture maize silage of cows in large scale farms in comparison with the other group (9.7 vs 7.7 kg DM) (Cedeberg and Flysiö, 2004). Energy use was higher in small dairy farms compared to large ones. The results of the survey (n=479) showed that common perception of some aspects of farming systems sustainability is frequently far from our data, in particular for climate change, eutrophication potential and energy use most of the people considered that large farm impact more than small ones. The results showed that intensive dairy farms with a high number of lactating cows fed with maize-based diet reduced the environmental impact of milk production particularly for greenhouse emission, energy use and land occupation compared with similar intensive farms but with lower number of cows. The study suggests that ecological sustainability is not compromised by increasing farm scale.

References

Capper, J.L., Cady, R.A., Bauman, D.E., 2009. The environmental impact of dairy production: 1944 compared with 2007. J. Anim. Sci. 87, 216-217.

Cederberg C., Flysjö A., 2004. Life cycle inventory of 23 dairy farms in South-Western Sweden. SIK report no. 728.

Rotz, C.A., Montes, F., Chianese, D.S., 2010. The carbon footprint of dairy production systems through partial life cycle assessment. J. Dairy Sci. 93, 1266-1282.

Van Calker K.J., Beretsen, P.B.M., Giesen, G.W.J., Huirne R.B.M., 2005. Identifying and ranking attributes that determine sustainability in Dutch dairy farming. Agr. Human Values 22, 53-63.

Table 1. Characteristics of two group of farms (least squares means)

	≤ 70 milking cows	≥ 150 milking cows	SE	*P*
N observations	11	11		
Daily milk production, kg FPCM/cow	28.0	29.2	1.02	0.413
Stocking density, LU/ha	4.40	5.83	0.68	0.150
Farm land, ha	25.1	78.0	7.5	< 0.001
Feed self-sufficiency,%	63.6	61.8	4.72	0.790
Production intensity, GJoule/ha	98.4	120	8.87	0.105
Gross margin, €/t FPCM	209	175	19.0	0.224
Dairy efficiency, kg milk/kg DMI	1.29	1.34	0.04	0.433
Lucerne,% farm land	16.9	18.0	3.66	0.833
Grass,% farm land	23.1	8.5	6.89	0.149
Maize for silage,% farm land	15.7	25.6	3.98	0.097
N balance, kg/ha	431	547	62.4	0.206
P balance kg/ha	54.3	54.9	10.40	0.968
N farm efficiency,%	27.3	28.0	0.78	0.555
P farm efficiency,%	28.3	31.2	0.95	0.044

Table 2. Environmental impact of the two groups of farms, expressed for kg of fat-and-protein-corrected milk (least squares means)

	≤ 70 milking cows	≥ 150 milking cows	SE	*P*
No. observations	11	11		
Climate change, kg Co_2-eq.	1.35	1.18	0.050	0.036
Acidification, g SO_2-eq. per kg	21.10	17.90	1.05	0.042
Eutrophication, g PO^{-3}_4-eq. per kg	9.74	8.15	0.590	0.072
Energy use, MJ	6.52	5.23	0.300	0.006
Land occupation, m^2	1.56	1.35	0.070	0.045

170. Guidelines for addressing environmental and social LCA within Quebec's food processing industry – Part 1: environmental aspects

Julie-Anne Chayer[1,*], Rosie Saad[1], Catherine Brodeur[2], Jean-Pierre Revérêt[2]

[1] Quantis, 395 Laurier Ouest, Montreal, QC, H2V 2K3, Canada, [2] Groupe AGéco, [*] Corresponding author. E-mail: *julie-anne.chayer@quantis-intl.com*

Many food industries are now developing responsible purchasing policies and challenging their suppliers by asking for guarantees with regards to their environmental and social practices. In response to such product and company-specific needs, life cycle assessment (LCA) is becoming a fundamental method widely used to support informed decisions within a broader sustainable management. Based on a holistic approach, LCA is a decision-support tool used for the assessment of potential environmental impacts of a product over its entire life cycle. Such a meaningful assessment helps to identify environmental hotspots where actions can be taken to reduce inefficiencies and energy cost while developing greener and socially acceptable products.

In an effort to provide a beneficial and competitive advantage for the food processing sector industries located in Québec (Canada), a framework is being developed addressing both environmental and social aspects. On one level, the outcome of this project encourages and facilitates the implementation and the achievement of a company and/or product LCAs from SME's to larger companies. Additionally, it can help position both product and company in a green market influenced by the increasing demands of consumers.

However, since LCA results are conditioned by several choices to be taken from the very beginning of a project, it is necessary to refine the scope and boundaries. This aspect is quite unique in this approach as only few studies have considered both aspects of sustainable development, up until now. In addition, the framework provided establishes a series of guidelines and rules that are built according to the ISO standards (14040-14044), the guidelines for social life cycle assessment of products (UNEP, 2009) and Product Category Rules (PCRs) development.

Such guidelines allow for an integrated and standardised approach for implementing a comprehensive LCA, in order to ensure quality, consistency and comparability among studies for various food processing sectors. In addition to providing information on how to define the functional unit, data quality requirements and other requirements of the LCA, the guidelines help to find the best data available for a specific sector in a specific region. The latter include sectors such as dairy products, poultry, juice and drinks, food packaging and many others.

The process of developing such an initiative for the sector as well as the guidelines' environmental requirements will be presented. This includes a list of stakeholders who participated in the guidelines' development and the integration process of the socio-economic and environmental issues. Gains made by food processors will also be shared.

References

UNEP/SETAC (2009). Guidelines for Social Life Cycle Assessment of Products. United Nations Environment Programme. Paris.

171. Guidelines for addressing environmental and social LCA within Quebec's food processing industry – Part 2: socio-economic aspects

Jean-Michel Couture[1,*], Catherine Brodeur[1], Jean-Pierre Revéret[1,2], Julie-Anne Chayer[3], Rosie Saad[3]

[1] Groupe AGECO, 2014 Cyrille-Duquet suite 307, Quebec, QC, G1N 4N6, Canada, [2] UQAM, Department of Strategy, Social and Environmental Responsibility, Suite R-3555, École des sciences de la gestion, QC, H2X 3X2, Canada, and Groupe AGECO, [3] Quantis, 395 Laurier Ouest, Montreal, QC, H2V 2K3, Canada, [*] Corresponding author E-mail: *jean-michel.couture@groupeageco.ca*

The growing demand of consumers for environmentally friendly products and socially responsible practices is becoming a major issue for the food processing industry. As new regulations are enacted and responsible procurement practices policies become widespread among large retailers, processors have to adapt and provide credible information on the environmental and social impacts induced by their products and activities. In order to support the Quebec's (Canada) food processing industry facing these new challenges, guidelines have been produced to allow processors operating in one of the five participating sectors (dairy, poultry, juice and drinks, processed vegetables and animal feed) to conduct Life Cycle Assessment (LCA) over their enterprise.

Although the LCA approach has become to be known as one of the most effective and rigorous method to assess the impacts of a product over its entire life-cycle, it traditionally focuses on environmental impacts, thus failing to provide a complete assessment of the product' sustainability. Doing so would require taking also into account it social and economic impacts. Based on recent theoretical developments, these guidelines propose a Sustainability-LCA framework to assess the social and economic impacts as well as the environmental ones.

Drawn from the UNEP's Guidelines for social life cycle assessment of products (UNEP 2009), the Social-LCA part of these Guidelines provides a fully operational framework in which impact sub-categories have been related to a set of readily measurable socio-economic indicators. Although the framework relies on a common and standardised assessment methodology, it also allows for adaptations to take place in order to take into account sectors' specificities. More fundamentally, by clarifying the S-LCA's terminology, steps of realisation and the indicators' selection criteria, these Guidelines enhanced this tool's methodological foundations to facilitate its use in other contexts and for other subjects.

In sum, in addition of being one of the first attempts to develop an integrated assessment framework to perform a Social and Environmental LCA, these Guidelines propose a practical, simplified and sector-oriented S-LCA methodology that will facilitate the conduct of such an assessment in the participating food processing sectors and beyond.

References

UNEP/SETAC, 2009. Guidelines for Social Life Cycle Assessment of Products. United Nations Environment Programme. Paris.

172. Aggregated indicators of sustainable agricultural production and food security in the Central Maghreb

J. Soler-Rovira[1,*], J.M. Arroyo-Sanz[1], F. González-Torres[1], C. Rojo-Hernández[1], A. Marquina Barrio[2].

[1] Escuela Universitaria de Ingeniería Técnica Agrícola, Universidad Politécnica de Madrid, Ciudad Universitaria, s/n 28040 Madrid, Spain, [2] Facultad de Ciencias Políticas y Sociología, Universidad Complutense de Madrid, Campus de Somosaguas, 28223 Pozuelo de Alarcón, Spain, [*] Corresponding author. E-mail: *jose.soler@upm.es*

A sustainable food production and supply for a growing human population is a major challenge for the next years and decades. Food security, including supply and demand factors as physical and economic access and wealth and assimilation of nutrients, must be performed by a sustainable agricultural production that ensures the long term agricultural productivity and ecosystem services. The region called the Central Maghreb includes Algeria, Morocco and Tunisia in the North of Africa. In these countries, despite the great economic differences, the agro-ecological characteristics are homogenous and are limiting productivity: arid and semi-arid climates, with low rainfall values along the year and concentrated in the sea-side land stripe, where are located also the most suitable farming soils. These ecological constraints are aggravated with political and socioeconomic ones that have resulted in a dual agricultural structure: high yield export-oriented vegetable and fruit farming systems and low-yield inner-consumption-oriented cereal farming systems. In these low input systems the technological use is scarce and there is an important yield gap between actual yields and potential ones. The production and inner supply of basic products as cereals or milk is mainly supplied by external imports from international markets, whereas vegetables and fruits are exported to foreign countries. In both cases, Spain and the EU are key actors in trade, agriculture, food, health and development policies.

This multidimensional challenge must be studied with a holistic approach, where sustainability and food security may be assessed by a number of indicators that reveal the strengths and weaknesses, as well barriers and drivers operating in each country in a multinational scenario and a multilevel assessment. This assortment of indicators should be synthesised into an appropriate unique indicator that in spite of containing much information, is easy to understand by the end-users (policy-makers, scientific, technicians, etc.). Aggregated indicators help to communicate the information succinctly and to make easier to distinguish patterns in the data by formalising the aggregation process that is often done implicitly, subjectively and intuitively. Indicator sets may be built up within a framework according to two conceptions of sustainability: goal or property oriented. The latter is based on systemic properties, such as existence or effectiveness, etc. The aim of this work is to assess the sustainability of the agricultural production and food security in the Central Maghreb making operational the systemic framework and incorporating multivariate statistical tools.

The hierarchical structure of the framework was based upon three subsystems: human (food security and social dimension), natural (environmental dimension) and support (production and economic dimension). Indicators were organised according to Bossel´s seven basic orientors (Table 1). The methodology was an iterative process consisting on several steps: system definition, selection of indicators and aggregation of indicators. This analysis was multinational, including Mediterranean countries, Middle East countries and others with ecological, sociological or economic similarities, such as South Africa, Norway and Iran. These countries were selected as the observation set. For each country, 21 indicators were computed (7 orientors x 3 subsystems). The data were obtained from FAO, United Nations, Worldwatch Institute, World Resources Institute and other international organisations. LCA was performed with some of the basic data.

Synthetic indicators were calculated by principal components analysis, using STATGRAPHICS software. The aggregation of data into single indices was done using coordinates of the countries with the principal components and the eigenvalues from the analysis. The indicator set considered for the selected countries is shown in Table 1. The selected indicators explained with good agreement the differences in sustainability and food security between the different countries and the synthetic indices ranked them all. The specific indexes that characterised the countries from the Central Maghreb were analysed in order to evaluate present and short term strengths and weaknesses to propose a development strategy.

Table 1. Indicator set for each selected country.

Basic orientor	Natural system	Human system	Support system
Existence	Impact of agricultural production per ha	%energy per capita intake by minimum requirements	Production per capita
Effectiveness	Eco-efficiency of agriculture	Agricultural production vs rural population	Cereal yield^{-1}
Freedom of action	Water footprint vs rainfall	Food consumption diversity	Net food imports per capita
Security	Anthropogenic nitrogen inputs	Basic food supply	Cereal yield stability
Adaptability	Rate of EF trend	Minimum food requirements	Slope of cereal yield trend over time
Coexistence	Share of basic and technical energy use	Undernourished population	Cereal yield gap
Psychological needs	EF vs. biocapacity	Life satisfaction	Input productivity

173. Consistent inclusion of deforestation in food life cycle assessment

Sebastien Humbert*, Vincent Rossi

Quantis, Parc scientifique EPFL, Bât. D, 1015 Lausanne, Switzerland, * Corresponding author. E-mail: *sebastien.humbert@quantis-intl.com*

Deforestation is recognised as being one of the major cause of greenhouse gas (GHG) emissions and destruction of ecosystems. It is also recognised that the majority of deforestation of natural ecosystems is associated with agriculture and agroforestry. On a world scale, GHG emissions from deforestation associated with agriculture and agroforestry are of the same order of magnitude as all other GHG emissions associated with agriculture and agroforestry production systems. The sector of food and beverage is rapidly incorporating the method of life cycle assessment (LCA) to address issues such as labelling, "food eco-design" but also to inform consumers and non-governmental organisations asking for more transparency on the environmental performance of food and beverage products.

However, among major limitations in doing LCAs on food and beverage products is the lack of consistent consideration of impacts associated with deforestation in inventory database and impact assessment results. Therefore, there is a need to develop and incorporate transparent and reliable data on deforestation in inventory databases and impact assessment results in order to increase accuracy of food and beverage LCAs.

Different approaches exist to address deforestation in LCA: the most common approach is to actually neglect this issue; the GHG protocol suggests to allocate deforestation to the cultures that have grown in the country where deforestation occurs; another approach is to allocate deforestation equally to all land cultivated in a specific area (normally the country); finally, another approach is to allocate deforestation to the culture on the boarder of the forest being deforested, assuming that it is this culture that causes deforestation.

In any cases, deforestation is most of the time not considered, which can be a significant bias for products produced in countries experiencing significant deforestation such as those in the tropics. In this context, at Quantis, we are evaluating the influence of incorporating deforestation consistently in inventory databases and impact assessment results, using the different approaches as sensitivity studies.

The presentation will show the contribution of deforestation in overall food and beverage LCA studies, using different allocation systems.

Results show that neglecting deforestation can cause a major underestimation of GHG emissions and other ecosystem impacts associated with products produced in tropical countries, such as palm oil, coffee, sugar cane, soybean or beef. In some cases, deforestation can double the GHG and ecosystem impacts as compared to when deforestation is neglected. For example, if considering the average annual Brazilian deforestation rate of 0.8% (in ha deforested/ha used for farming), the GHG emissions associated with green coffee production can double.

In addition - and this is something even more neglected in most LCA food studies - to be consistent, impacts of deforestation should also be considered in studies indirectly using such products, as for example, potential impacts from deforestation in LCAs of European milk production where part of the dairy cows fodder is based on soybean produced in regions where deforestation occurs.

This presentation will highlight the cases when deforestation should be considered with care to evaluate the actually potential environmental impacts of food and beverage products.

174. Comparison of current guidelines to calculate the carbon footprint of carrots

Svanes, Erik

Ostfold Research, Fredrikstad Norway, E-mail: *erik.svanes@yahoo.no*

The Carbon Footprint of carrots grown in South-Eastern Norway was calculated using a calculation method following the principles of ISO 14067 and harmonised with the PAS 2050 and GHG Protocol Products Guideline. No calculation method for the CFP of carrots exists but some guidelines for vegetables do: The “Fruit and Vegetables” PCR from the Japanese CFP Pilot Programme, the “Vegetables” PCR from the International EPD system and the guideline for calculating CFP of Horticultural Products from Productschap Tuinbow. In addition the sector guidance for horticultural products PAS 2050-2 was evaluated. These guidelines are not identical.
The Carbon Footprint was found to be 0.38 kg CO_2-eq./kg carrot sold to consumer. The product system stopped at retail but included the waste handling for the materials wasted in the consumption stage. The total Cradle-to-Grave carbon Footprint was 0.55 kg CO_2-eq./kg carrot, but there are large uncertainties in the calculations of emissions from the consumer stage.
The effect of applying different methodological choices in the calculation of Carbon Footprints was examined using the abovementioned PCR and Horticultural guideline as example. All guidelines excluded Capital Goods on the basis that the impact is low but for carrots the effect was found to be >1% of the cradle-to-gate CFP. Some issues are not adequately addressed in the guidelines. Emissions from electricity can be very different depending on whether the national grid or some multinational is considered, and whether or not green electricity schemes such as the European Guarantee of Emissions scheme is being accounted for. In some cases the guidelines give different recommendations. The allocation in recycling and recovery is a prime example.
This study shows that there is a need for more harmonisation between CFP and EPD schemes around the world. It also shows that it might be necessary with more detailed guidance than guidelines covering a whole sector or a whole range of products (“Horticulture products”, “Vegetables” or “Vegetables and fruits”). The study has resulted in recommendations for a standard method for carrots and similar products.

References

Blonk, H, Kool A, Luske B, Ponsioen T and Scholten J, 2010. Methodology for assessing carbon footprints of horticultural products. A study of methodological issues and solutions for the development of the Dutch carbon footprint protocol for horticultural products. Blonk Milieuadvies. http://www.blonkmilieuadvies.nl/en/pdf/engels%20rapport%20pt%202010.pdf

ISO DIS 14067. International Standards Organisation 2012. http://www.iso.org/iso/iso_catalogue/catalogue_tc/catalogue_detail.htm?csnumber=59521

BSI (British Standards Institute) (2011). Publicly Available Specification 2050. http://www.bsigroup.com/en/Standards-and-Publications/How-we-can-help-you/Professional-Standards-Service/PAS-2050/PAS-2050/

WBSCD (World Business Council for Sustainable Development) and WRI (World Resources Institute) (2011): Greenhouse Gas Protocol. Product Life Cycle Accounting and Reporting Standard. http://www.ghgprotocol.org/standards/product-standard

The International EPD system (2011): Product Category Rules for CPC Group 012: Vegetables. http://www.environdec.com/en/Product-Category-Rules/Detail/?Pcr=8074

The Japanese Carbon Footprint Pilot Project. Product Category Rules for Vegetables and Fruits (approved PCR ID:PA-BF-02). http://pcr-library.edf.org.tw/data/japan/Vegetables%20and%20Fruits.pdf

175. Establishing a reference carbon calculator and policy options to promote low carbon farming practices in the EU

Hanna L. Tuomisto*, Vincenzo Angileri, Camillo De Camillis, Adrian Leip, Nathan Pelletier, Fabien Ramos, Palle Haastrup

European Commission, Joint Research Centre (JRC), Institute for Environment and Sustainability, Via Enrico Fermi 2749, 21027 Ispra, Italy, * Corresponding author. E-mail: *hanna.tuomisto@jrc.ec.europa.eu*

The direct emissions of greenhouse gases (GHG) from agriculture accounted for around 10% of total European Union (EU) emissions in 2010. It has been estimated that in the United Kingdom the agri-food sector as whole contributes around 19% of GHG emissions when life cycle emissions are taken into account. To reduce the farming-related GHG emissions appropriate policy instruments and supporting tools that promote low carbon farming practices must be developed. This paper describes an on-going project that aims at assessing the policy options to promote low carbon farming practices in the EU. The project includes: i) a review of existing climate-related certification and labelling schemes in agri-food sector, ii) the development of a user friendly open-source carbon calculator suitable for assessing the life cycle GHG emissions from different types of farming systems across the whole EU, and iii) the design/assessment of policy options for promoting low carbon farming practices. The carbon calculator quantifies direct and indirect GHG emissions according to the general-level and sector-specific international standards and guidelines on Life Cycle Assessment and carbon footprint. In addition to the GHG emission quantification, the tool also proposes mitigation options and sequestration actions suitable for single farms. The recommendations of the specific farming practices are based on emission reduction potential, potential leakage effects, inherent costs of implementation, and impact on other environmental issues. The practicality and acceptability of the carbon calculator has been tested on around a hundred farms across the EU. Finally, a range of options for making widespread use of the carbon calculator will be outlined, including e.g. public or private certification schemes, incentive payments to farmers, and legal obligations for farmers to reduce GHG emissions.

References

European Environment Agency, Approximate EU GHG inventory, Early estimates for 2010, in EEA Technical report No 11/2011.

Garnett, T., Cooking up a storm – Food, greenhouse gas emissions and our changing climate. 2008. Food Climate Research Network, Centre for Environmental Strategy, University of Surrey. p. 155.

European Commission, Joint Research Centre, International Reference Life Cycle Data System (ILCD) Handbook – General guide for Life Cycle Assessment – Detailed guidance (2010) Luxembourg: Publication Office of the European Union.

176. Surveying tools and methods for LCA in the agri-food sector

Ricardo Teixeira*, Lori Gustavus, Anne Himeno, Sara Pax

Bluehorse Associates, 12 rue Soyer, 92200 Neuilly-sur-Seine, France, * Corresponding author. E-mail: *ricardo.teixeira@bluehorseassociates.com*

The objective of the present paper is to understand how Life Cycle Assessment (LCA) practitioners make choices between different tools, depending on their objectives. In order to do so, we divided our work in two stages:

1. The first state was making a review of how many tools are in the LCA market, and which are their main characteristics. We reviewed 63 tools out of more than 100 that are available, 11 of which we tested to some extent.
2. The second part was conducting a survey on LCA practitioners, trying to understand how those tools are used and to what end. The basis for this work was an adaptation of the questions by Cooper and Fava (2006). The survey was announced in the PRe Consultants LCA discussion list, and sent by e-mail to the Bluehorse Associates (BHA) mailing list. A total of 117 LCA practitioners answered at least one question in the survey. Since BHA is a sustainability metrics company specialised in the food industry, there was a high share of replies by LCA practitioners in agriculture and food. This inherent bias was intended for this study. Our objective is to understand, from the standpoint of an informed LCA practitioner, which solutions are available, what differentiates them, and how they adapt to each specific objective of the studies.

Our first finding was that the frontier between “full LCA” (ISO compliant) tools and simplified, non-standardised tools (simplified LCA) is now much fuzzier. Simplified tools are becoming more accurate, while full LCA tool developers are coming up with their own simplified tool versions. Simplification is today a synonym with user-friendliness and practicality, not necessarily lack of rigor.

Part of the explanation for this has to do with political context. There is now a need for more practical, business-oriented tools that respond to the high demand created by the generalisation of product LCA. Other part of the explanation was found during the survey (Teixeira and Pax, 2011). Even though most respondents claim to follow some kind of standard, they do not always submit their studies to peer review (Table 1). Research and development, innovation and eco-design are the most mentioned objectives of LCA studies today, and all of these are internal to companies. In fact, almost all tool providers organise seminars, webinars or some other forum to communicate with users. Learning and knowledge transmission from developers to users is now a key concern, as many companies do not have in-house LCA expertise, but LCA is progressively done in-house. Since the focus is no longer on communication, simplification has gained in importance against standardisation.

Still, practitioners quote data availability as their main challenge. Simplified methods, for example those based on large quantities of secondary data, address this concern. The tool review confirmed that the number of data providers is still very low, and data availability is a fair concern.

Another interesting finding was that most tools do not easily display trial versions or disclose much information about the tool and databases included. In many cases pricing models are either very complex or absent from public display. So, the task of a practitioner selecting the best-suited tool for the project’s objectives is difficult, due to the disappearing frontier between full LCA and simplified LCA, the many similar options available, and the lack of transparency in price and use. Unless the practitioner has previous pointers or well-defined targets to start with, it is very difficult to make an informed choice without spending time and resources surveying the market for a long period of time. In the future, it is highly recommendable that tools become transparent about what they can deliver to clients and how they are different from their competitors.

References

Cooper, J.S., Fava, J., 2006. Life-Cycle Assessment Practitioner Survey Summary of Results. Journal of Industrial Ecology, 10, 12-14.

Teixeira, R., Pax, S., 2011. A Survey of Life Cycle Assessment Practitioners with a Focus on the Agri-Food Sector. Journal of Industrial Ecology, 15, 817-820.

Table 1. Answer to the multiple-choice question "Are your LCAs peer-reviewed?".

Are your LCAs peer-reviewed?	Number of practitioners
Yes, always	6
Yes, sometimes	35
Yes, occasionally	24
No	26
Total	91

177. PISC'n'TOOL: an operational tool for assessing sustainability in aquaculture systems

Aurélie Wilfart[1,2,*], Thomas Merle, Syndhia Mathé[3], Joël Aubin[1,2]

[1] INRA, UMR1069 Sol Agro et hydrosystème Spatialisation, F-35000 Rennes, France, [2] AGROCAMPUS OUEST, F-35000 Rennes, France, [3] University of Montpellier 1 (UM1), Faculté d'Economie, Av. Raymond DUGRAND C.S. 79606 F- 34960 Montpellier, France, [*] Corresponding author. E-mail: *aurelie.wilfart@rennes.inra.fr*

To meet challenges of producing more while lowering impacts on ecosystems, new fish-farming systems have to be designed. The first step before any changes in farming systems is to make a diagnosis on environmental impacts, economic viability and social aspects of the activity. The assessment step has to be as complete as possible, adapted to the system and scientifically robust, but also convenient for the user. To fulfil this complex combination, we designed an operational tool for fish-farm systems assessment based on LCA conceptual framework and Emergy accounting: PISC'n'TOOL. This tool was created in the PISCEnLit project (Ecologically Intensive PISciculture project) funded by the French National Research Agency.

PISC'n'TOOL aims to analyse environmental farm scale impacts of a fish farm according to LCA attributional approach, to determine the contribution of the farm's components and environmental intervention to its impacts and also to analyse farm's performances (zootechnical, economical, and social). The targeted users are researchers and agricultural advisors for fish farming in France, Indonesia and Brazil. PISC'n'TOOL applies a cradle to farm gate analysis; the farming system evaluated focuses on the fish farms and its main inputs (Fig. 1). For fish farms associated to livestock (i.e. integrated pig -fish farming system), the terrestrial production are outside of the system boundaries, meaning manure/slurry used for fish production is considered as an input with specific allocation rules. Temporal coverage of PISC'n'TOOL is a period of 1 year or one production cycle in order to be adapted to the evaluated system. According to the multiple functions of a farm, PISC'n'TOOL defines 5 functional units: one tonne of fish produced, one m^3 of water used, one on-farm hectare, one human labour unit, and 1000 $ of farm income, calculation depending on specific user's data or incremented data base. The farm's environmental inventory is based on tables of energy carriers, infrastructures, equipments, vehicles, chemicals and veterinary products, water consumption, feeds (up to 10 different feeds with 15 ingredients are allowed) and fry/fingerlings. These data stem from previous aquaculture LCA studies or new data collected during PISCEnLIT project. Farm emissions (N, P, solids emissions) are calculated using mass balance modelling (Cho et al., 1990). Emission and consumption data are aggregated into midpoint indicators (included also Net Primary Production Use) according to CML 2 baseline 2001 (version 2.04) method and endpoint indicators (human health, Ecosystem and resources) according to Recipe endpoint H Eur H/A method.

PISC'n'TOOL allows also to provide Emergy indicators (based on LCA data), but also zootechnical, economic efficiency level and social indicators (Fig. 2). Results are systematically given in tables for each functional unit or in graphical form. The resultants ensure an easy comparison between different running scenarios for one farm or for different systems as well. Despite a large interne database, this tool requires additional data from the users to perform the LCA.

PISC'n'TOOL is a practical tool for the multidimensional evaluation of farming system. The tool allows the identification of the environmental, economical and social hotspots of fish farm, and thus can help to define improved strategies and scenarios for farm evolution.

References

Cho CY, Kaushik SJ. Nutritional energetics in fish: energy and protein utilization in rainbow trout (*Salmo gairdneri*). World Rev Nutr Diet. 1990;61:132-72.

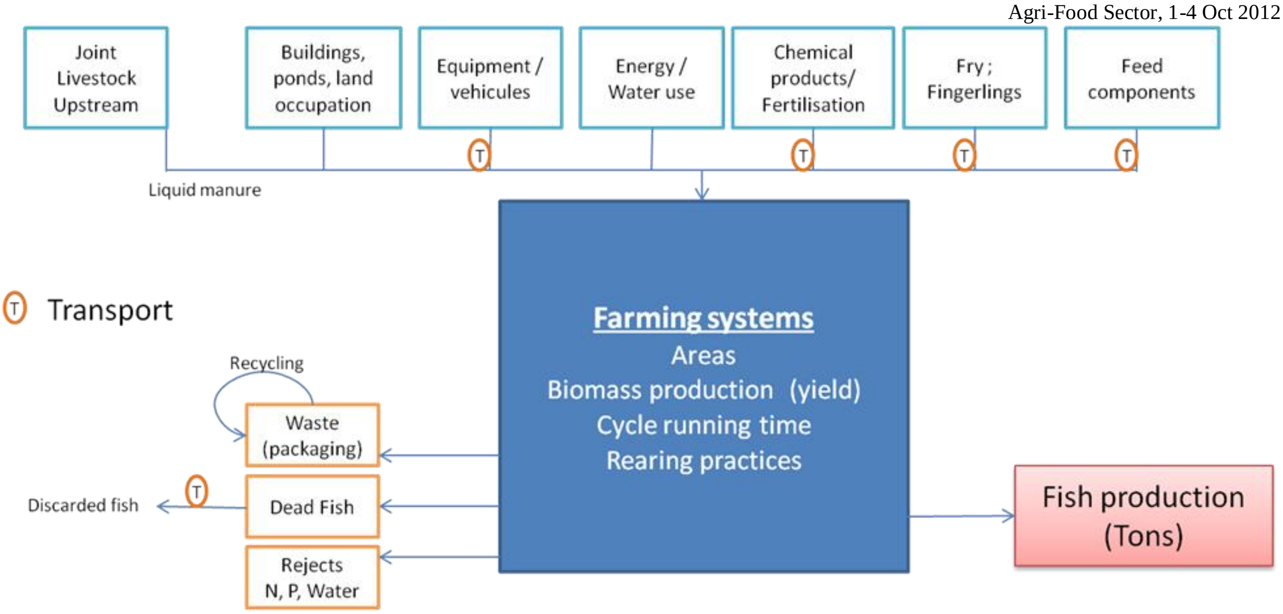


Figure 1. Fish farm system boundaries

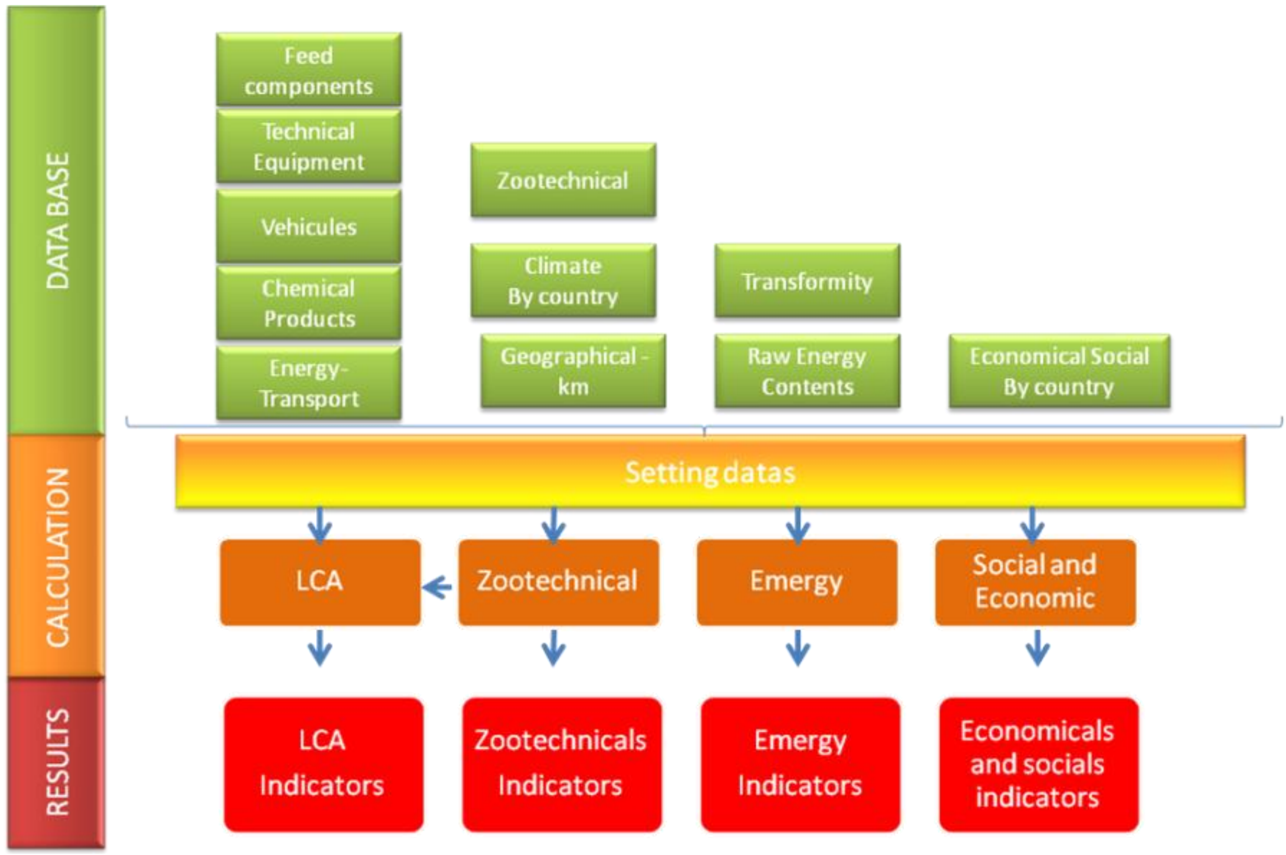


Figure 2. Simplified representation of PISC'n'TOOL framework

178. Food and climate change: FOODprint as a tool to support GHG reduction in the Thai agri-food sector

Shabbir H. Gheewala[1,2,*], Rattanawan Mungkung[3]

[1] The Joint Graduate School of Energy and Environment (JGSEE), King Mongkut's University of Technology Thonburi, Bangkok, Thailand, [2] Center for Energy Technology and Environment, Ministry of Education, Thailand, [3] Centre of Excellence on enVironmental strategy for GREEN business (VGREEN) and Department of Environmental Science, Faculty of Science, Kasetsart University (KU), Bangkok, Thailand, [*] Corresponding author. E-mail: *shabbir_g@jgsee.kmutt.ac.th*

As a leading food producer and a major exporter, Carbon Footprinting (CF) and Labelling are seen as useful tools in Thailand to quantify GHG emissions and identify the priority areas for GHG reduction for climate change mitigation and to stimulate the development of innovative technology/management to produce lower-carbon food products. CF was initiated through academic-industrial collaborative projects and national pilot projects leading to the development of the national CF guideline as well as carbon footprint labelling scheme. Though it has been well taken by the Thai agri-food industry for a few years already, it is still very difficult for them to identify the required data, data sources and collection methods particularly for background and secondary data. More importantly, they echoed the need for quick CF calculation for effective business decisions which was limited by their lack of understanding of the underlying scientific background and methodological issues (Mungkung et al., 2010). These issues are critical especially for small and medium enterprises (SMEs) who do not always have the necessary competence. This has led to the initiative in developing a carbon footprint calculation tool so called "FOODprint", for the Thai agri-food industry, which will serve as a means for capacity building. The development of FOODprint is being carried out by VGREEN-KU, JGSEE and the Federation of Thai Industries together with 40 Thai food companies; the studied products covering different sectors: agriculture, livestock, fisheries and aquaculture. FOODprint is based on spreadsheets, with the flexibility to add on new sets of databases specific to a supply chain. The data requirements and calculation methods are based on PAS 2050: 2011 (BSI, 2011). Specific templates for data input and for CF calculation for plant/animal-based production systems, processing, packaging, transport of inputs/distribution, sales, pre-consumption, and post-consumption waste management are included. A user guideline of FOODprint provides the principles, methodology and practical approach for data collection, transformation and input. The CF results are illustrated in tables and graphs showing contribution analysis, including comparison with similar products from previous studies for benchmarking. Possible strategies to reduce GHG emissions are suggested for further analysis and the high-level analysis of each option is calculated and compared for management decisions. This also fits well for developing Nationally Appropriate Mitigation Actions (NAMA), supporting wider application of carbon footprinting and labelling, as well as to anticipate the market requirement of carbon labelling both for domestic and export products, which will contribute to climate change mitigation and promote low-carbon trade between Thailand and EU. This will also lead to a synergistic effect of this project with the EU's existing Integrated Product Policy (IPP) which aims to improve the environmental performance of products along their life cycle. As Thailand is the very first country in ASEAN taking initiative on carbon footprinting and labelling, the knowledge and experiences gained from this project can be shared with other ASEAN countries through the collaborative framework of "ASEAN Climate Change Initiative: ACCI" for joint response and efforts to combat climate change.

References

Mungkung R., Gheewala, S., Bonnet, S., Hospido, A., Kanyarushoki, C., van der Werf, H. and Poovarodom, N. 2010. Carbon Footprinting and Labelling: Experiences from the Thai food industry Available online: http://www.thaicarbonlabel.sci.ku.ac.th/

BSI, 2011. PAS 2050:2011. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution

179. Determining greenhouse gas emissions profiles for Australian agricultural products at a regional scale: methodological opportunities and obstacles

Philippa M Brock[1,*], Phillip Graham[1], David F Herridge[2], Patrick Madden[1], Graeme D Schwenke[1]

[1] Department of Primary Industries, New South Wales, Australia, [2] University of New England, New South Wales, Australia, [*] Corresponding author. E-mail: *pip.brock@industry.nsw.gov.au*

Life Cycle Assessments (LCAs) of greenhouse gas emissions associated with agriculture have commonly been conducted at a farm scale (case study) or emissions are calculated at a national scale. However, until recently, there has tended to be a scarcity of regionally-applicable assessments. This may be partly due to difficulties in obtaining representative data that account for variability in agricultural production systems.

However, for national policies to be regionally applicable and for landholders to identify opportunities for practice change, it is essential that inter- and intra-regional differences be better understood. We contend that by accommodating variability, highly robust regional LCAs can be built and that this broader analysis will improve our understanding about the representativeness of existing case-studies.

We have found the greatest opportunity for emissions reduction in cropping systems to be replacing synthetic nitrogenous fertilisers with biologically fixed N, with emissions from a legume-based system found to be 33% of those from a non-legume system (Fig. 1; Table 1). Other factors which greatly influence calculated emissions per unit of product are yield variation across seasons and the choice of direct nitrous oxide emissions factor (Brock et al. accepted; Herridge et al. 2011; Schwenke et al. 2011). We have also found variability in emissions from sheep enterprises, ranging from 39% reduction for change in enterprise emphasis from wool to meat production (Table 2), to approximately 28% variability for change in wool price or calculation method, to 23% variability for change in fibre diameter, to 10% variability for change in fleece weight (Brock et al. in preparation.; Graham et al. 2010).

To account for this variability, we are:

1. obtaining regional-level data, including measures of variability, from research and extension staff
2. providing detailed documentation about the variables and discussing other environmental effects
3. testing the sensitivity of the emissions profile by changing the parameter values for formulas constructed in SimaPro and checked against FarmGAS
4. applying data from instrumented regional field trials
5. testing the effect of changes in economic allocation in animal production systems
6. using data from modelling packages, such as GrassGro, to compare long and short term variability.

The variability associated with many LCA inputs and lack of enterprise-level seasonally-specific data makes farm-scale (case-study) assessment problematic. It seems highly desirable for there to be greater focus on regional-scale LCA, to support both testing of national policies and on-farm emissions reduction.

References

Brock, P., Madden, P., Schwenke, G., Herridge, D. Greenhouse gas emissions profile for 1 tonne of wheat produced in Central Zone (East) New South Wales: A Life Cycle Assessment approach. Crop and Pasture Science, accepted.

Brock, P., Graham, P., Madden, P. Greenhouse gas emissions profile for 1 kg of wool produced in the Yass Region, New South Wales: A Life Cycle Assessment approach. Animal Production Science, in prep.

Graham, P. 2010. Bookham Grazing Demonstration Results. Department of Primary Industries, Yass NSW.

Herridge, D., Schwenke, G., Brock, P. 2011. Nitrogen-fixing chickpea underpin productive, sustainable farming systems in Australia's Northern Grains Region. Proceedings of the 25th Latin American Conference on Rhizobiology, 9 Sep 2011, Uruguay.

Schwenke, G., Brock, P., Herridge, D. 2011. Nitrogen-fixing legumes in farming systems reduce greenhouse gas emissions. Proceedings of the 17th International Congress on Nitrogen Fixation, 27 Nov - 1 Dec, Fremantle, Western Australia.

Table 1. Total greenhouse gas emissions for canola-wheat and chickpea-wheat production systems with different levels of fertiliser N, i.e. zero (0N) or 80 kg N/ha (80N)

Rotation[A]	Pre-farm and on-farm emissions year 1 (kg CO_2-e/ha)	Pre-farm and on-farm emissions year 2 (kg CO_2-e/ha)	Total emissions per ha over 2 years (kg CO_2-e)
Canola (80N)–wheat (80N)	896	908	1804
Chickpea (0N)–wheat (80N)	306	848	1154
Chickpea (0N)–wheat (0N)	306	297	603[B]

[A]The canola and chickpea crops yielded 1.8 t/ha; the three wheat crops yielded 3.0 t/ha; [B]Total emissions from chickpea (0N)–wheat (0N) are 33% of those from canola (80N)–wheat (80N); using ecoinvent and the Australasian LCI database updated May 2012.

Table 2. Sensitivity of the emissions profile for sheep enterprises to changes in enterprise from wool dominance to the production of first-cross lambs

Enterprise type	Value of wool (%)	Value of mutton (%)	Value of lamb/live animal (%)	Enteric methane (kg /ha)	Total emissions (kg CO_2-e/kg greasy wool)
19-micron wool production	56	32	12	99.1[A]	25.6
19-micron ewes joined to Dorset rams for meat production	30	11	60	94.0 + emissions from production of feed and replacement ewes	15.5[B,C]

[A]Based on daily modelling for the 51-year long-term average period; [B]Includes emissions from the production of wheat (0.157 kg CO_2-e) and replacement ewes (2.29 kg CO_2-e); [C]Emissions decreased by 39% due to change in enterprise emphasis.

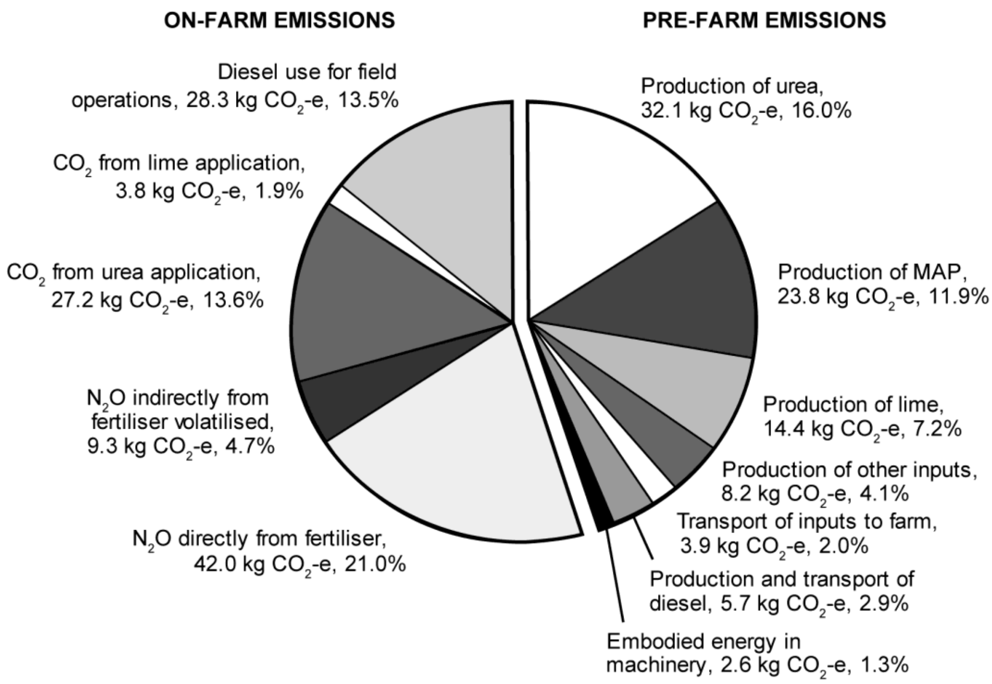


Figure 1. Greenhouse gas emissions (kg CO_2-e) from the production of 1 tonne of wheat in Central Zone (East) NSW, Australia.

180. Energy analysis of agricultural systems: uncertainty associated with energy coefficients non-adapted to local conditions

Mathieu Vigne*, Philippe Faverdin, Jean-Louis Peyraud

[1] INRA, UMR PEGASE, Domaine de la Prise, 35590 Saint-Gilles, France, * Corresponding author. E-mail: *mathieu.vigne@rennes.inra.fr*

Among studied impacts in Life Cycle Analysis, fossil energy use has been widely considered. But choice of energy coefficients from the literature and their ability to express accurately local conditions is questioned for territories where references for inputs life-cycle are lacking. This study measured fossil energy use in dairy farms and assessed uncertainty associated to energy coefficients in order to improve energy analysis methodology of agricultural systems.

Fossil energy use for forty two dairy farms from Poitou-Charentes (PC) and thirty from Reunion Island (RI) have been analysed using PLANETE for PC (Bochu, 2002) and PLANETE MASCAREIGNES for RI (Thevenot et al., 2010). Uncertainty analysis and sensitivity analysis has been conducted through the SIM-LAB tool (Saltelli et al., 2004). Uncertainty analysis consisted in a Monte-Carlo methodology: 30,000 sets of energy coefficients have been randomly drawn from a uniform law between minimum and maximum values found in the literature for each energy coefficients. Uncertainty is expressed by 95% confidence interval of average fossil energy use in megajoule per litre of milk produced ($MJ.l^{-1}$). Estimation of sensitivity of energy coefficients is based on similar drawn and has been studied through the calculation of the Standardised Regression Coefficient (SRC).

Estimated probability distribution is reported in Fig. 1. Minimum and maximum values for 95% confidence interval are respectively 3.6 and $5.0MJ.l^{-1}$ for PC and 5.8 to $8.2MJ.l^{-1}$ for Run. The corresponding variabilities from mean were ±16% and ±17% respectively for PC and RI. Whereas they could appear low, these values question comparison of systems from different territories. Among the set of coefficients chosen, difference between the territories could appear large or conversely equal when considering higher values for PC and lower values for RI. This results highlights need for a common methodology for calculation of energy coefficients. This could enable to calculate energy coefficients adapted to local conditions and to produce accurate values of energy use of agricultural systems. Such method should concern clear definition of system boundaries in indirect energy assessment and promote precise investigation of the technology used in the different processes.

SRCs obtained for the different energy coefficients (Table 1) showed that the most sensitive energy coefficients are not the same in the two territories. Energy coefficient for concentrate feeds is mainly responsible of this uncertainty for RI farms whereas it is a combination of several energy coefficients for PC farms (electricity, concentrate feeds, animal buildings, fuel, N fertiliser). Calculation of adapted energy coefficients could be associated to a preliminary sensitivity analysis through minimum and maximum values of energy coefficients found in the literature in order to focus on the most influential energy coefficients and to fit an appropriate value for them. This will avoid adapting all energy coefficients which could be time-consumer.

Nonetheless, energy coefficients do not represent the only source of uncertainty in energy analysis. Uncertainty related to inputs data could be decrease as done in our study with large surveys of real farms and individual economic follow-up surveys based on representative years. Uncertainty related to the methodology could be decrease, in addition to common methodology for energy coefficients, by common choice of allocation method.

References

Bochu, J.L., 2002. PLANETE: Méthode pour l'analyse énergétique des exploitations agricoles et l'évaluation des émissions de gaz à effet de serre. Colloque national: Quels diagnostics pour quelles actions agroenvironnementales? Toulouse, France.

Saltelli, A. Tarantola, S., Campolongo, F., Ratto, M., 2004. Sensitivity Analysis in Practice. A Guide to Assessing Scientific Models. Probability and Statistics Series, John Wiley & Sons Publishers.

Thévenot A., Vigne M., Vayssières J., 2011. Référentiel pour l'analyse énergétique et l'analyse du pouvoir de réchauffement global des exploitations d'élevage à La Réunion. CIRAD-FRCA, Saint-Pierre, France.

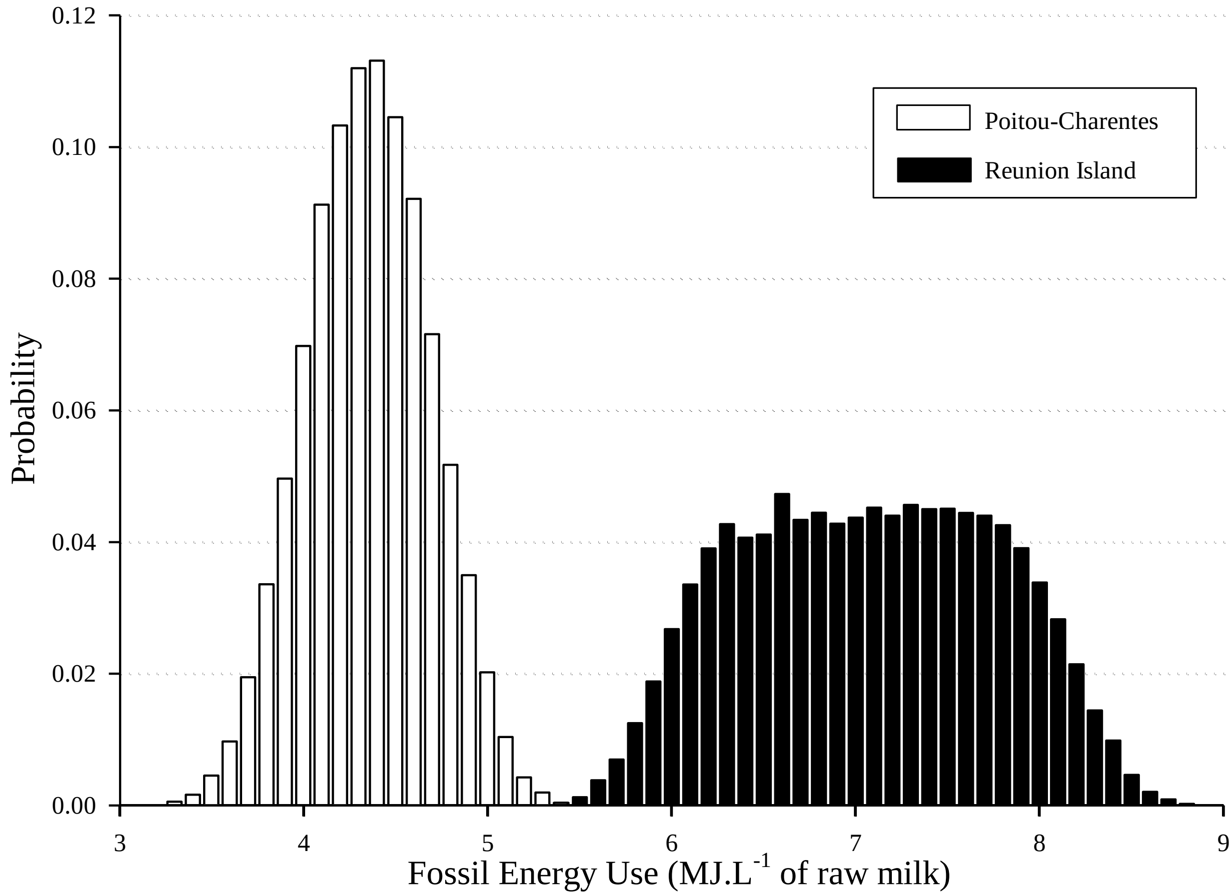


Figure 1. Probability distribution of energy use for dairy farms from (a) Poitou-Charentes and (b) Reunion Island calculated with the 30,000 sets of energy coefficients

Table 1. Standardised Regression Coefficients (SRC) of the five most influential energy coefficients for Poitou-Charentes and Reunion Island dairy farms

Poitou-Charentes		Reunion Island	
Energy coefficients	SRC	Energy coefficients	SRC
Electricity	0.53	Concentrate feeds	0.91
Concentrate feeds	0.51	Tractor	0.25
Animal buildings	0.47	Fuel	0.17
Fuel	0.38	Electricity	0.15
N fertiliser	0.27	Animal buildings	0.14

181. Uncertainty analysis in a comparative LCA between organic and conventional farming of soybean and barley

Monia Niero*, Alessandro Manzardo, Sara Toniolo, Filippo Zuliani, Antonio Scipioni

CESQA (Quality and Environmental Research Centre), University of Padova, Department of Industrial Engineering, Via Marzolo 9, 35131 Padova, Italy, * Corresponding author. E-mail: *monia.niero@unipd.it*

There are several methods based on different approaches to quantify and analyse uncertainty (Lloyd and Ries, 2007). One of the main advantages of the uncertainty analysis method used within Ecoinvent database (Frischknecht et al., 2005) is to determine a correlation between data quality and the uncertainty of LCIA results (Cooper and Kahn, 2012).

The objective of this research is to test the effectiveness of the uncertainty analysis methodology developed by Scipioni et al. (2009) in the case of a comparative Life Cycle Assessment. The uncertainties on the LCA input come from a qualitative assessment by data quality indicators based on the pedigree matrix. The research considered two different cultivation techniques: organic (system A) and conventional (system B) farming of a 3-year crop cycle for the production of soybean in the first and third year and barley in the second year of the triennial crop. The LCA study was conducted in accordance with the ISO 14040 standards (ISO 2006a,b), using the ReCiPe 2008 methodology for the LCIA step (Goedkoop et al., 2008). The functional unit was 1 kg of seeds, composed respectively by 2/3 kg of soybean from first and third years of the 3-year cycle and 1/3 kg of barley from second production year. The results of the comparison between the two farming systems at damage category level (Fig. 1.)

Concerning the damage category resources, conventional farming presents higher impacts than organic, because of the resources (oil and gas) used in the production of triple superphosphate and urea fertilisers. On the other hand the damage to ecosystems is higher for organic farming, because of the lower crop yields. Within human health end-point category results, it is controversial to determine which is the best option, because of the minor differences among the two farming systems. The first step of the uncertainty analysis allowed the selection of the main parameters contributing to the uncertainty for both the systems under study, through a contribution analysis at the damage assessment level, with 1% cut-off and the assignment of a probability distribution. The most influencing input data for the human health category are shown in Table 1. The second step included the quantitative uncertainty analysis through Monte Carlo simulation (10^3 iterations), considering the number of comparison runs in which organic farming (A) is larger than conventional farming (B) (Fig. 2).

The methods developed by Scipioni et al. (2009) showed its effectiveness when applied to comparative LCA. The results confirmed that for human health there are no significant differences among the two farming systems. Finally, the application of the two step methodology for the quantification of uncertainty connected with the results allowed to define to which extent the LCIA results at damage level are reliable.

References

Cooper, J.S., Kahn E., 2012. Commentary on issues in data quality analysis in life cycle assessment. Int J Life Cycle Assess doi10.1007/s11367-011-0371-x.

Frischknecht, R., Jungbluth, N., Althaus, H.J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischer, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2005. The ecoinvent database: overview and methodological framework. Int J Life Cycle Assess 10 (1), 3–9.

Goedkoop, M., Heijungs, R., Huijbregts M, De Schryver, A., Struje, J., van Zelm, R., 2009. ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and then endpoint level. First edition, Report I: Characterisation.

ISO, 2006a. Environmental management. Life cycle assessment. Principle and Framework. ISO 14040:2006. International Organization for Standardisation, Geneva, CH.

ISO, 2006b. Environmental management. Life cycle assessment. Requirements and Guidelines. ISO 14044:2006. International Organization for Standardisation (ISO), Geneva, CH.

Lloyd, S.M., Ries, R., 2007. Characterizing, Propagating, and Analyzing Uncertainty in Life-Cycle Assessment. A Survey of Quantitative Approaches. J of Ind Ecol 11(1), 161-179.

Scipioni, A., Mazzi, A., Niero, M., Boatto, T., 2009. LCA to choose among alternative design solutions: The case study of a new Italian incineration line. Waste Manege 29, 2462–2474.

Table 1. Contribution analysis for the damage category Human health.

Inventory data	**Organic (A)**	**Conventional (B)**
Emission from soil management (N_2O, NO_x) - soybean	63.2%	43.9%
Emission from soil management (N_2O, NO_x) - barley	21.5%	22.0%
Diesel consumption	8.8%	6.3%
Organic compost	2.9%	-
Triple superphosphate	-	20.0%
Soybean seed	1.7	2.4%
Urea	-	3.3%
Other processes	2.0%	1.5%

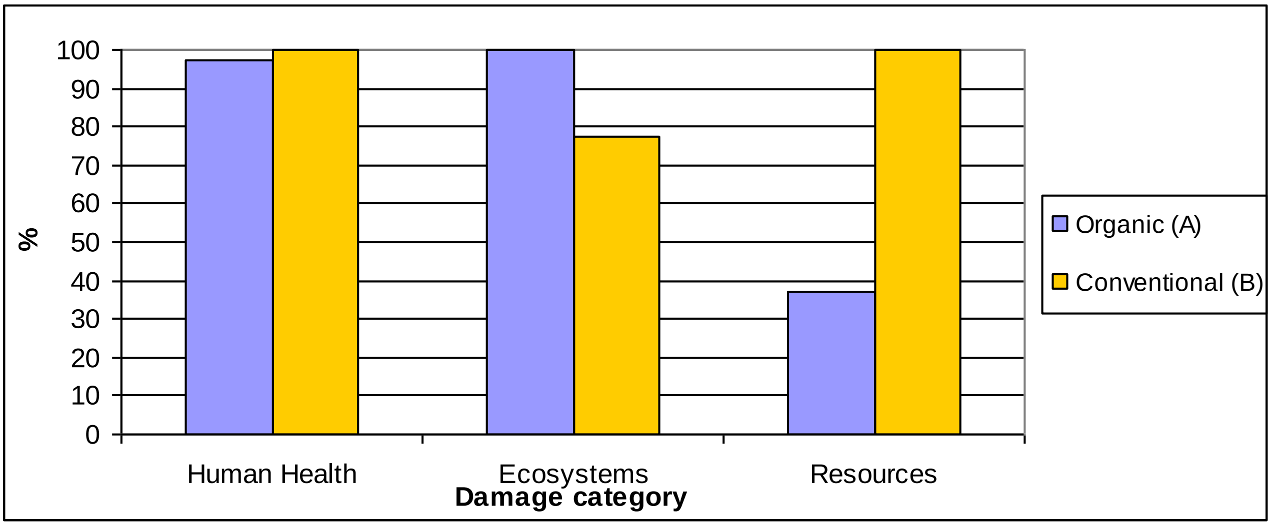


Figure 1. LCIA results from the comparative LCA between organic and conventional farming of soybean and barley

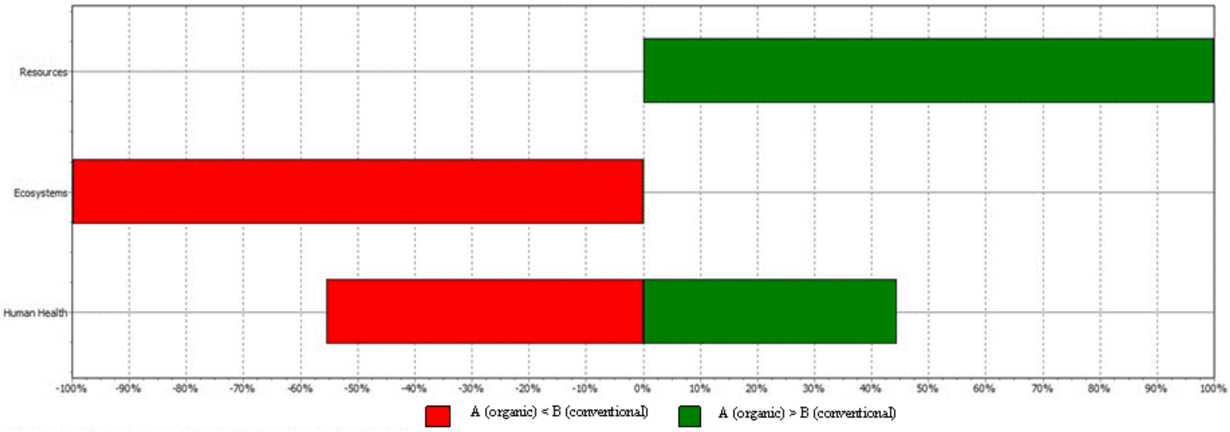


Figure 2. Monte Carlo results of the comparison between organic (left) and conventional (right) farming of soybean and barley

182. Analysis and propagation of uncertainty in agricultural LCA

Xiaobo Chen[1,2,*], Michael Corson[1,2]

[1]INRA, UMR1069 Sol Agro et hydrosystème Spatialisation, F-35000 Rennes, France, [2]Agrocampus Ouest, F-35000 Rennes, France, Corresponding author. E-mail: *xiaobo.chen@rennes.inra.fr*

The confidence in LCA results depends primarily on the quality of source data and their pertinence for the system studied. However, the need for large amounts of data leads to much uncertainty in impact estimates due to the data themselves: measurement, use in calculations, and final transformation into impact estimates. The main sources of uncertainty in the data chain include not only statistical uncertainty (mean and standard deviation) of the data, but also methodological choices in LCA, such as hypotheses made to represent the system of interest, data representativeness, impact assessment methods, and allocation of impacts between co-products. Therefore, consideration of uncertainty in LCA would provide more scientific information for decision making. This topic is the focus of doctoral research recently begun at INRA that aims to (1) identify sources of uncertainty in agricultural-production systems, (2) analyse their propagation, and (3) estimate their relative contributions to the overall uncertainty in calculated impacts.

Although some texts describe uncertainty generally as a lack of knowledge, its definition may change depending on the LCA steps in which it occurs (Huijbregts, 1998). Therefore, uncertainty is often classified according to its nature and source (e.g., "natural" (i.e., variability) vs. "epistemic"; Van Asselt and Rotmans, 2002). Epistemic uncertainty has been subdivided into three sources: scenario, model and parameter (Fig. 1). This first step allows uncertainties from each source to be evaluated by corresponding approaches.

Monte-Carlo analysis is used most frequently to evaluate uncertainty in LCA (Basset-Mens, 2009). With it one can estimate the influence of uncertainties in input variables (using their probability distributions) on predicted potential impacts. However, this approach suffers some methodological bias due to correlations between variables and poorly-known response rules. For example, the selection of appropriate distributions is usually based on literature, expert judgment, or empirical studies in other systems, which may increase the complexity of model and parameter uncertainty. Moreover, Monte-Carlo simulation is commonly used for assessing the influence of uncertainty in emission factors in LCIA, but it may not be appropriate for uncertainty in other steps, such as the definition of scope or functional unit or the interpretation of results that consist of both subjective and objective uncertainty (Fig. 2). Although it is not necessary or possible to consider all uncertainties in LCA, subjective uncertainty should not be overlooked. More complex approaches (e.g., fuzzy logic) exist, but their use remains marginal. Currently, the methods used to describe uncertainty propagation in LCA have not tried to differentiate the various types of uncertainty but rather to aggregate them. Thus, more research is necessary to overcome barriers to analysing uncertainty in LCA.

Thesis work will begin by identifying and classifying uncertainty in each LCA step, especially uncertainties frequently encountered when assessing agricultural systems. With case studies, the research will identify the most important uncertainties, develop methods to categorise them, and work to estimate the contribution of each source of uncertainty to the overall uncertainty in output. By considering uncertainty in agricultural LCA, more complete information about environmental impacts can be given to decision-makers, who should consider uncertainty as an important part of decision analysis.

References

Basset-Mens, C., Kelliher, F.M., Ledgard, S., Cox, N., 2009. Uncertainty of global warming potential for milk production on a New Zealand farm and implications for decision making. Int. J.LCA 14, 630-638.

IPCS, 2008. Uncertainty and Data Quality in Exposure Assessment. Geneva, World Health Organization, International Programme on Chemical Safety (Harmonization Project Document No. 6).

Huijbregts, M.A.J., 1998. Application of uncertainty and variability in LCA. Part I: a general framework for the analysis of uncertainty and variability in life cycle assessment. Int. J. LCA 3, 273-280.

Van Asselt, M.B.A., Rotmans, J., 2002. Uncertainty in integrated assessment modelling. Int. J. LCA 54, 75-105.

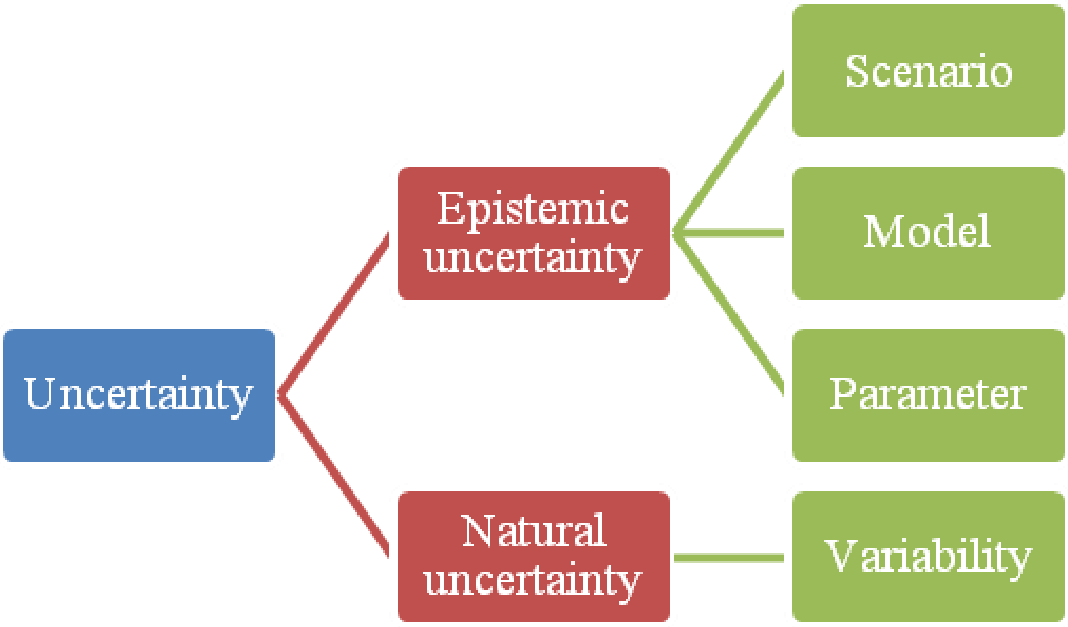


Figure 1. Classification of uncertainty types (Van Asselt and Rotmans, 2002; IPCS, 2008).

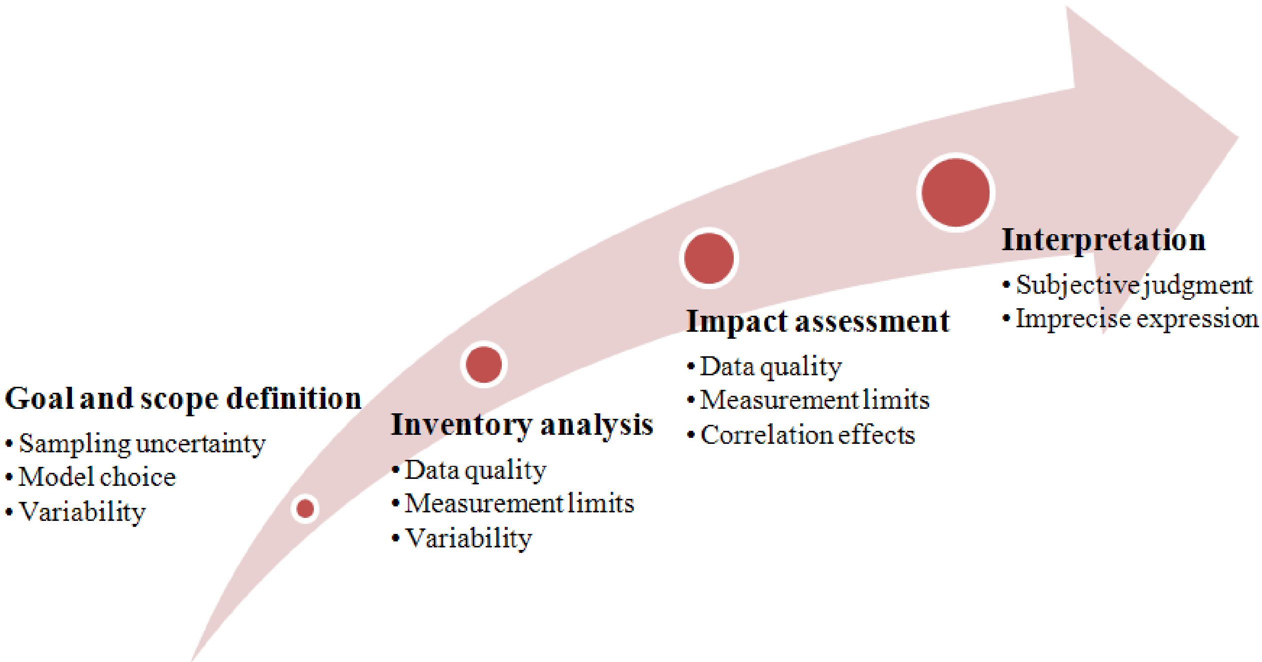


Figure 2. Uncertainties in different steps of LCA.

183. Quantifying the impact of in-data variability and uncertainty on the life cycle assessment of dairy products

Jacob Fels

University of Witten/Herdecke, Center for Sustainable Leadership, Alfred-Herrhausen-Str. 50, 58448 Witten, Germany, E-mail: *jacob.fels@uni-wh.de*

Calculating the Carbon Footprint (CF) of consumer goods is of increasing importance. Many "cradle-to-retail" CFs are dominated by the climate impact resulting from the production of raw materials especially animal based raw materials. In many cases processing, transportation and production processes are playing minor roles. Within this context, the ratio of animal raw materials to the overall recipe weight is of particular relevance. Given the current degree of standardisation in CF calculations and the accuracy of existing Product Category Rules (PCR), differences, variability and uncertainty of CFs is highly contingent on assertions, methodological choices, different assumptions, geographical and temporal scopes, data selection and data aggregation. For instance, as Henriksson et al. (2011) have shown, the CF of milk varied between -17% and +17% from the mean due to management differences between Swedish dairy farms. The present study contributes to the quantification and analysis of the uncertainty in CFs. The study's findings concerning the causes, magnitude and sources of variability and uncertainty in CFs can provide the basis for the definition of PCRs and contribute to the overall understanding of CFs.

This study examines the influence of different assumptions pertaining to the dairy sector on the results of a CF for 1kg energy-corrected milk (ECM). This study concentrates on in-data variability and uncertainty concerning the pre-farm-gate phase. Besides focusing on in-data variability a parameter relating to methodological choices is included by analysing the influence of economic allocation. (For parameters see Table 1.) To quantify the uncertainty in the CF and to identify a realistic range of results, empirical boundary values are identified and assigned to each parameter. The subsequent multi-scenario-analysis examines the impact of these parameters on the uncertainty in the CF.

While keeping the output data like GHG emissions per kWh electricity etc. on a constant level the CF for 1 kg ECM is 1.11kg CO_2e without allocation. From all 6 examined parameters, the assumed average milk yield has the greatest influence on the end-result CF. While using an empirically supported upper and lower threshold for the milk yield, the CF of the 1 kg ECM varies between -17% and +15% (Table 1).

The findings from this study can contribute to the definition of PCRs in the dairy sector. Relatedly, the study emphasises that PCRs should be evaluated with regard to their potential for reducing the volatility of CFs. The study identifies the crucial influencing parameters for this purpose. Particular emphasis is given to determining the impact of the assumptions concerning the average milk yield on the CF.

References

Belitz, Hans-Dieter, Hrsg. (2010) Calculations of emissions from German agriculture 2010/2008. Braunschweig: vTI, Johann-Heinrich-von-Thünen-Institut.

Ciroth, A., Fleischer, G., Steinbach, J. (2004). Uncertainty calculation in life cycle assessments. Int J LCA, 9(4).

Dorfner, Gerhard, und Guido Hofmann. (2011). Milchreport Bayern 2010 Ergebnisse der Betriebszweigabrechnung Milchproduktion 2009/10. Bayerische Landesanstalt für Landwirtschaft.

Flachowsky, G., Brade, W. (2007). Potenziale zur Reduzierung der Methan-Emissionen bei Wiederkäuern. Züchtungskunde, 79(6), 417–465.

Hirschfeld, Jesko, Weiß, J., Preidl, M., Korbun, T. (2008). Klimawirkungen der Landwirtschaft in Deutschland. Berlin: IÖW.

Huijbregts, M. A. J., Gilijamse, W., Ragas, A. M. J., & Reijnders, L. (2003). Evaluating Uncertainty in Environmental Life-Cycle Assessment. Environmental Science & Technology, 37(11), 2600-2608.

Henriksson, M., A. Flysjö, C. Cederberg, und C. Swensson. (2011). Variation in carbon footprint of milk due to management differences between Swedish dairy farms. animal 5, 1474-1484.

Statistisches Bundesamt (2009), Milcherzeugung und –verwendung, Fachserie 3 Reihe 4.2.2.

Table 1. Parameters, boundary values, and change in carbon footprint (CF).

Parameter	Boundary values			Change of CF in%	
Milk yield (kg ECM/cow and year)	6072	6977	8446	15%	-17%
Feed DMI (kg DMI(cow and year)	5653	6242	6830	-6%	6%
Share of concentrated feed in DMI (% per kg DMI)	0,18	0,32	0,46	-2%	2%
Lifespan (years)	3,10	3,60	4,10	3%	-2%
Quantities of enteric fermentation (kg CH4 per kg DMI)	0,02	0,02	0,03	-7%	3%
Allocation% to milk (based on economic values)	100%		85%	0%	-12%